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Uncertainty in critical load assessment models

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Author(s):
[1]Richard Skeffington, [2]Jane Hall, [2]Liz Heywood, [2]Richard Wadsworth, [1]Paul Whitehead, [3]Brian Reynolds, [4]John Abbott and [4]Keith Vincent

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Research Contractor:
[4]AEA Technology (Netcen), B551 Harwell, Didcot, Oxfordshire, OX11 0QJ. Tel: 0870 190 6590

Environment Agency's Project Manager:
Bernard Fisher, Kings Meadow House, Reading

Collaborator(s):
[1]Skeffington Consultants Ltd
[2]CEH Monks Wood
[3]CEH Bangor

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Steve Killeen Head of Science

Executive Summary

This is the final report of a project aimed at quantifying the uncertainties in critical load assessment models. The work forms part of Environment Agency R&D Project No. p4-120/4: *Acidification and Annual Audits*. The aim of the work described in this report was to examine the uncertainties in critical load assessments and develop a practical methodology for such assessments within the Environment Agency's regulatory role. This work built on an earlier Environment Agency R&D project which developed methods for assessing the uncertainty in critical loads due to uncertainty in the input parameters to critical load models; the methods were then applied to a forested site in southern England.

In this report, a programme of testing uncertainties in the acidity critical load model used for terrestrial ecosystems was carried out at the same site. The sensitivities to choice of input parameters, their statistical distribution type and their intercorrelation were calculated. It proved important to include correlations between deposition parameters. The methods were then applied to other coniferous forest sites. It was hoped that this would identify parameters which could be selectively targeted for research to narrow output uncertainties, but almost any input parameter proved potentially important. The methods were also applied to heathland, unmanaged woodland and critical loads for nutrient nitrogen, covering the entire range of terrestrial critical load methods used in the UK.

An uncertainty analysis of acidity critical loads for freshwaters was carried out using similar methods, and applied to the 22 sites of the UK Acid Waters Monitoring Network. Here, one parameter - the current base cation concentration in waters - dominated the contribution of input parameters to total uncertainty. For both aquatic and terrestrial ecosystems, output parameter uncertainty was generally considerably less than input parameter uncertainty, thought to be due to a 'compensation of errors' mechanism. The methodology could also be used to calculate probability of fish population reduction or extinction instead of a critical load.

The methodology was then applied on a national scale using managed coniferous forests as an example. Similar patterns of sensitivity were observed, but also variability between regions. At a national scale other uncertainties were identified, such as uncertainty in mapping habitats. The problems of estimating uncertainties at designated sites are discussed, with examples, in this report. A comparison of critical loads calculated using site-specific and national parameters identified a number of differences. It was concluded that it is inadvisable to attempt to interpret national-scale critical load maps at a local or site-specific scale.

A method for modelling critical load exceedance was developed and tested at a regional scale. It was concluded that use of bivariate normal distributions for both deposition and critical loads was the most appropriate method, but that the critical load distribution needed to take into account site conditions. Various presentation methods for critical loads and exceedance are discussed.

A case study at the Liphook site concluded that reducing the emissions from power stations and refineries alone would not significantly reduce the chance of exceeding the acidity critical load. Reducing the emissions from all Environment Agency-regulated sources more generally would have some effect, comparable with that associated with reducing local ammonia emissions from agriculture. This sector analysis is likely to be site-specific, but could be applied elsewhere.

Finally, a set of frameworks to allow the Environment Agency to incorporate critical load uncertainty in its regulatory role is proposed here. It would be useful to be able to trial these in a practical context. The advantages and disadvantages of incorporating uncertainty into critical load assessments are discussed, and a number of recommendations for further work are made.

Contents

Introduction

1.1 Overall aims of the project

This project is part of a larger Environment Agency umbrella project, *Development of methods to assess and manage the impacts of major industries on air quality*, which addresses a broad range of issues affecting the assessment of air quality.

The aim of the umbrella project is "*to establish better methods with which the Environment Agency can assess changes to acid deposition exceedance and to airborne concentration, from changes in the emissions it regulates, or on which it is asked to comment during consultations*."

Two specific objectives for the overall project were identified, namely:

1. To examine the uncertainties in critical load assessments and develop a practical methodology for such assessments within the Environment Agency's regulatory role (Task 1);

2. To develop an appropriate methodology for use by the Environment Agency to audit, on an annual basis, the industrial source contributions to priority air quality concentrations (Task 2).

This report addresses the first of these objectives (Task 1).

Methods used by the Environment Agency must be practical, reliable and defensible whilst using up-to-date modelling techniques and available data sets. It is also important that the methods developed can be applied to any Environment Agency-regulated industrial source within the UK, to ensure consistency within the regulatory process. The complexity and variability of the natural environment makes developing such methods a difficult task. This report summarises our investigations into various aspects of critical load uncertainty in the UK, and makes some recommendations for a practical methodology for critical load assessment within the Environment Agency's role. It also outlines aspects which need further investigation.

1.2 Breakdown of work

1.2.1 Introduction

This work builds on a previous Environment Agency contract (Abbott *et al.,* 2003) during which an assessment was made of the uncertainties of critical loads at Liphook, a coniferous forested site in southern England (Skeffington *et al.*, 2006). The purpose of the present investigation was to extend this earlier work to a wider range of sites and ecosystem types, and to investigate other aspects of critical load uncertainty relevant to the Environment Agency's role. To this end, the work was organised into six subtasks:

Subtask 1.1: Determine the uncertainty in critical loads for other terrestrial and freshwater ecosystem types;
Subtask 1.2: Draw conclusions on likely dependence of uncertainty in critical loads on habitat size and location;
Subtask 1.3: Clarify uncertainty at a site-specific scale;

Subtask 1.4: Develop and recommend a practical methodology for critical load assessment within the Environment Agency's regulatory role;
Subtask 1.5: Recommend ways to present critical load exceedances;
Subtask 1.6: Make recommendations about source sectors requiring the greatest attention in a critical load assessment.

These are discussed in more detail below. The interactions between the subtasks were such that it was decided not to use the subtask titles as section headings, but to write the report in a more coherent fashion. Section 1.2 explains where the results from each subtask are to be found.

1.2.2 Uncertainty in critical loads for different ecosystem types

Subtask 1.2 required the uncertainty in critical loads to be determined at as many terrestrial and freshwater ecosystem types as possible, using the most up-to-date critical load assessments. Critical loads for the UK are based on empirical methods and simple mass balance (SMB) equations. Acidity critical loads for non-woodland terrestrial habitats are based on the empirical acidity critical loads for soils. The steady state mass balance (SSMB) equation is applied to both managed and unmanaged woodland habitats. In each case, habitat-specific inputs are used to calculate critical loads. Mass balance critical loads of nutrient nitrogen ($CL_{nut}N$) are applied to the managed woodland habitats and empirical critical loads to the remaining terrestrial habitats (Hall *et al.,* 2003a). Freshwaters are assessed by the First Order Acid Balance (FAB) model. The aim of the subtask was assess the uncertainty attached to a selection of these methods at a selection of sites. Uncertainty would also be calculated at the national scale for at least one of these habitat types.

During the course of the work, it became clear that a number of assumptions related to the input parameters needed to be investigated in a systematic way. It was most convenient to do this at coniferous forest sites, as these are the sites for which most data are available. The method developed was then applied to other coniferous forest sites, and then to heathland and unmanaged woodland sites. A comparison with the empirical method for heathland is included. Finally, uncertainty in the critical load for nutrient nitrogen at two coniferous forest sites was calculated. All this is described in Section 2.1. Section 2.2 covers critical loads for freshwaters. Uncertainty in the model used was calculated at 22 sites of the UK Acid Waters Monitoring Network. The work therefore covers at least one example of every critical load method used in the UK.

Because it was thought best to separate the site-specific and national applications in the report, the application of critical load uncertainty at the national scale is discussed in Section 3.1. Section 3.2 considers the relative importance of two major input parameters (calcium weathering and deposition) at a national scale.

1.2.3 Dependence on habitat size and location

Subtask 1.2 was to draw conclusions on the likely dependence of uncertainty in critical load on habitat size and location. This is discussed in Sections 3.3 to 3.6.

1.2.4 Uncertainty on a site-specific scale

Subtask 1.3 involved clarifying uncertainty at a site-specific scale. A comparison of critical loads and their uncertainties based on either national or site-specific data at a managed coniferous woodland site is made in Section 4. Comparison of acidity critical loads based on (i) dominant soil type; (ii) least-sensitive soil type; (iii) most sensitive soil type; (iv) area-weighted mean by soil type is given in Section 3.3, which also considers other aspects of critical loads in relation to soil type. The suitability of criteria/models used at the national scale for site-level assessment is discussed in Section 4.3. Uncertainties in applying critical load methodology to designated sites are discussed in Section 3.4, and in mapping habitats in Section 3.5.

1.2.5 A practical methodology for critical load assessment

Subtask 1.4 was to develop and recommend a practical methodology for critical load assessment within the Environment Agency's regulatory role. A hierarchical practical approach is suggested in Section 6.

1.2.6 Methods of presenting critical load exceedance

Subtask 1.5 was to recommend ways of displaying results in terms of mapping critical load exceedance. Several possible approaches are described in Section 7. A methodology for performing an uncertainty analysis of critical load exceedance, and the presentation of results for the South East of England, is described in Section 5.

1.2.7 Source sector case study

Subtask 1.6 was to make recommendations on source sectors requiring the greatest attention in a critical load assessment. This is, of course, likely to be site-specific: a case study of one site is described in Section 8.

2 Site-specific uncertainty

Summary

- This section describes the calculation of uncertainty in critical loads at a number of individual sites, both terrestrial and aquatic.
- Estimates were made of the range and statistical distribution of all the input parameters to the critical load models at given sites, and the ranges and distributions of the resulting critical loads and exceedances were estimated using Monte Carlo methods. The sensitivity of these outputs to the uncertainty in each input parameter was also calculated.
- The effects of various assumptions about input parameters were tested at Liphook, a coniferous forest site in southern England used in previous work for the Environment Agency. The model used was the Steady State Mass Balance model, as used in national critical load assessments.
- Using largely national defaults rather than measured data to parameterise the Liphook site led to an increase in uncertainty of the critical load for acidity, the coefficient of variation (CV) increasing from 37 to 60 per cent. With the measured parameters, uncertainty in weathering rates was most important, whereas with the national defaults uncertainty in calcium deposition dominated.
- The above point illustrates that uncertainty in base cation deposition can be a major contributor to terrestrial critical loads as well as to exceedance.
- Correlations between input parameters should be taken into account in Monte Carlo analysis. Changing the correlation between the two weathering rate parameters had little effect on critical load uncertainty, but incorporating correlations between deposition parameters reduced the uncertainty of exceedance and some critical load values at Liphook. Critical loads and exceedance also became less sensitive to uncertainty in deposition if correlations were included. Including correlations is recommended for future work.
- Use of different statistical distributions for terrestrial input parameters had only a small effect at Liphook. Lognormal or truncated normal distributions seemed best for deposition parameters.
- Liphook was compared with two contrasting coniferous forest sites (Thetford and Aber) using the same methodology. Different input parameters were significant in determining output parameter uncertainty at different sites. Thus, it is not easy to target research towards narrowing uncertainties except on a site-specific basis.
- However, on-site measurements clearly reduced uncertainty in critical loads and exceedances at several sites.
- The uncertainties in critical loads for heathland and unmanaged woodland were calculated for a single site in North Wales. The empirical method used to generate critical loads for heathland produced lower critical loads and greater uncertainty than the SSMB method used for unmanaged woodland. At this site, uncertainty in deposition was much more important than uncertainty in the terrestrial parameters in influencing exceedance uncertainty.
- The critical load for nutrient N was calculated by the mass balance method at Liphook and Aber. The CVs were quite low, and the method at these sites was most sensitive to the definition of acceptable nitrogen leaching. Deposition parameters dominated the uncertainty in exceedance.
- Uncertainty in critical loads for freshwater ecosystems was calculated using the First-Order Acid Balance (FAB) model on data from the UK Acid Waters Monitoring Network (AWMN).

- Uncertainty in a single parameter, the present-day non-marine base cation concentration, dominated the uncertainty in critical loads calculated by FAB. Uncertainty in N deposition was the most important influence on uncertainty in exceedance at most sites.
- The FAB model allows predictions of fish population health as well as critical loads to be used in a probabilistic fashion for assessment purposes. This is demonstrated with two Lake District sites.
- Uncertainty in N deposition dominated uncertainty in critical load exceedance for most of the aquatic sites, the remainder being sensitive to the present non-marine base cation concentration.
- If 90% confidence of non-exceedance is required, all 18 GB sites were exceeded with 1998-2000 deposition. For 50% confidence, there were 15 sites, and for 10% confidence, 12 sites. The choice of exceedance probability is arbitrary, but will have a very large effect on perceptions of damage.
- A meta-analysis showed that the order of uncertainty of critical loads for both terrestrial and aquatic systems, from most to least, was: $CL(A) > CL_{max}S > CL_{max}N > CL_{min}N$. Coefficients of variation were surprisingly low, generally 20-45%, due to 'compensation of errors'.
- At no site was there 100% confidence of exceedance or non-exceedance.

2.1 Critical loads for terrestrial ecosystems

2.1.1 Introduction

Previous work for the Environment Agency in Project P4-083(5) included a Monte Carlo analysis of the critical load for acid deposition at Liphook, a coniferous forest site in southern England. It was shown how estimates of deposition and critical loads at this site could be combined into a probabilistic estimate of exceedance. This section describes how this work was extended. Firstly, the influence of various assumptions about the input parameters to the critical load model was tested at the same site. These were: the effects of correlations between input parameters; the effects of changing assumptions about the amount of uncertainty in input parameters; and the effects of different assumptions about distribution types. Having developed a reasonable method for parameterisation of the Liphook site, the work was extended to other coniferous forest sites. These were research sites with contrasting attributes, and the results of calculation of critical load uncertainties at these sites were used to test the hypothesis that the relative sensitivity of the critical loads to different input parameters would remain the same, thus enabling research to narrow these uncertainties to be appropriately targeted. These sites were also used in a comparison of the uncertainty in critical loads obtained using national data with that obtained using site-specific data. This is described in Section 3.4. The Steady-State Mass Balance (SSMB) model used in these assessments was then applied to a heathland site, and compared to the uncertainty in the empirical critical load method which is actually used in the UK for heathland. Finally, the SSMB was used to investigate uncertainty in the critical load for nutrient nitrogen at two sites in the UK.

2.1.2 Base case

It was decided to initiate the contract by re-running the Monte Carlo analysis previously performed (Abbott *et al.,* 2003). There were two reasons for this: more information and data had been accumulated on appropriate uncertainty values to use for the input parameters (see Heywood *et al.,* 2006a; Skeffington, 2006); and it was desired to compare the site-specific values used before with data obtained from national data sets. The site-specific parameters were calculated for a coniferous forested site near Liphook in Hampshire (Lat. 51º 04' N.; Long. 0º 47' W.; OS Grid Reference 4857 1299). National and default parameters for the same site were calculated as explained in a review by Heywood *et al.* (2006a).

For the Monte Carlo analyses presented in this report, the input parameters are described first in the form of a table which lists the parameters used and their characteristics. These consist of the distribution type (such as rectangular, normal) and appropriate descriptive statistics. For rectangular distributions these are the mean and upper and lower limits; for normal and lognormal distributions the mean and standard deviation. Also noted is whether any correlations between input variables were included in the form of a correlation coefficient (*r*). These input tables will mostly be found in the Appendices to this report, but Tables 2.1.1 and 2.1.2 below are provided as examples. Outputs from the analyses are presented in various ways depending on the question being addressed, but typically they will include descriptive statistics for one or more response variables. These will include the mean, standard deviation and if appropriate the coefficient of variation (CV: the standard deviation divided by the mean and expressed as a percentage). Also typically included in the outputs are the results of a sensitivity analysis. This calculates the percentage contribution of variation in each input parameter to uncertainty in the given output parameter, described in Appendix A1.1.

The input parameters used for the initial runs are shown in Table 2.1.1, and those for the original runs in Table 2.1.2. These differed considerably, differences being highlighted in Table 2.1.2. The uncertainty ranges are generally larger in the present study, and some distribution types have changed. Though the correlation structure appears to be different, the use of the Ca_{corr} parameter (the proportion of the ANC_w which is Ca weathering, where ANC is the acid neutralising capacity) is an alternative method of estimating correlation between ANC_w and Ca_w, as Ca_w is varied within the Monte Carlo analysis by multiplying ANC_w by Ca_{corr}. This ensures that Ca_w is always less than ANC_w. Bootstrapping was used to estimate the effective correlation coefficient as about 0.42. Otherwise the changes are largely due to the use of available site-specific parameters for the previous work (Table 2.1.2).

Table 2.1.1: [1]SSMB parameters used for the initial runs, Liphook site

Parameter	Units	Mean	Lower	Upper	SD	Distribution	Correlations
ANC_w	eq $ha^{-1}yr^{-1}$	100	0	200		Rectangular	None
BC_u	eq $ha^{-1}yr^{-1}$	250	125	375		Rectangular	None
Ca_{dep}	eq $ha^{-1}yr^{-1}$	430			215	Normal	None
Ca_{corr}	unitless	0.1	0	0.2		Rectangular	None
Ca_w	eq $ha^{-1}yr^{-1}$	10				Calculated	None
Ca_u	eq $ha^{-1}yr^{-1}$	160			43.2	Normal	None
Q	m^3 $ha^{-1}yr^{-1}$	4,100			943	Normal	None
$[BC_l]$	µeq L^{-1}	2	2	2		None	None
Ca/Al_{crit}	mol mol^{-1}	1	0.5	1.5		Rectangular	None
K_{Gibb}	m^6 eq^{-2}	950	760	1,140		Rectangular	None

[1]Abbreviations are expanded in the *List of Abbreviations* at the end of the report, and defined if necessary in the Appendices.

The critical load for acidity (*CL(A))* was used as the response variable. It is defined in Equation A4 in Appendix A. Perhaps not surprisingly, the results are different from those previously obtained. The deterministic critical load value for the site is 701 eq $ha^{-1}yr^{-1}$ using the default values in Table 2.1.1 and 516 eq $ha^{-1}yr^{-1}$ using the default values in Table 2.1.2. Some statistics for the distribution of CL(A) are shown in Table 2.1.3. A thousand runs were used to produce the original data, and 5,000 runs for the present study. More runs were required to reach a stable output in the present study due to the larger uncertainty ranges, and this number of runs was retained as standard in subsequent work.

Table 2.1.2: SSMB parameters used for original study, Liphook site

Parameter	Units	Mean	Lower	Upper	SD	Distribution	Correlations
ANC_w	eq $ha^{-1}yr^{-1}$	150	50	250		Rectangular	0.9 Ca_w
BC_u	eq $ha^{-1}yr^{-1}$	250	125	375		Rectangular	None
Ca_{dep}	eq $ha^{-1}yr^{-1}$	175	0		33	Lognormal	None
Ca_{corr}	unitless					Not used	
Ca_w	eq $ha^{-1}yr^{-1}$	105	35	175		Rectangular	0.9 ANC_w
Ca_u	eq $ha^{-1}yr^{-1}$	125	62.5	187.5		Rectangular	None
Q	m^3 $ha^{-1}yr^{-1}$	3,469	3,125	3,817		Triangular	None
$[BC_l]$	μeq L^{-1}	1	0	2		Rectangular	None
Ca/Al_{crit}	mol mol^{-1}	1	0.5	1.5		Rectangular	None
K_{Gibb}	m^6 eq^{-2}	1,025	0		410	Lognormal	None

Values which differ from Table 2.1.1 are shaded.

Table 2.1.3: Statistics of Monte Carlo estimates of CL(A) at Liphook, original and new parameters

Variable	Original	New
Mean	0.555	0.736
Standard deviation	0.205	0.444
Coefficient of variation (%)	37	60
Median	0.538	0.696
Maximum	1.397	2.933
Minimum	0.112	0.0002

Values are in keq $ha^{-1}yr^{-1}$.

It is clear from Table 2.1.3 that not only is the critical load greater using the revised estimates, but the range, standard deviation and coefficient of variation are significantly larger, indicating greater uncertainty. This is unsurprising, as the uncertainty ranges of the input parameters are also larger. A sensitivity analysis on the results was also carried out using the methods described above. The parameters making most contribution to variability (Table 2.1.4) differed considerably between the runs. A few more parameters were considered for the original data, so the results are not strictly comparable, but the pattern is clear. In the new run, *CL(A)* is overwhelmingly most sensitive to calcium deposition, whereas previously the acid neutralising capacity (ANC) and Ca weathering rates had the most influence. The critical ratio, Ca/Al_{crit}, makes about the same contribution to both runs, but other parameters have little influence in the new run, whereas originally there were minor contributions from base cation and calcium uptake.

Table 2.1.4: Percentage sensitivity of CL(A) to each parameter

Parameter	Units	Original	New
ANC_w	eq $ha^{-1}yr^{-1}$	25.5	2.5
BC_u	eq $ha^{-1}yr^{-1}$	6.2	<0.1
Ca_{dep}	eq $ha^{-1}yr^{-1}$	5.2	83.8
Ca_{corr}	unitless	-	<0.1
Ca_w	eq $ha^{-1}yr^{-1}$	26.2	-
Ca_u	eq $ha^{-1}yr^{-1}$	7.1	2.3

Q	$m^3\ ha^{-1} yr^{-1}$	<0.1	0.2
$[BC_l]$	µeq L^{-1}	<0.1	-
Ca/Al_{crit}	mol mol^{-1}	10.6	11.1
K_{Gibb}	$m^6\ eq^{-2}$	0.5	0.1

This piece of work indicated that two equally reasonable input parameter sets can lead to very different conclusions about the relative importance of input parameters to uncertainty in output variables, and the amount of uncertainty to be attributed to the variables themselves. It highlighted the need to develop an appropriate method for estimating uncertainty in the input parameters. This was explored as described in the next sections.

2.1.3 Effects of correlation between weathering parameters

In order to explore the reasons for the differences between new and old runs, it was decided to test the hypothesis that there was some relation with the correlation structure of the input parameters. The only relevant correlation was that between ANC_w and Ca_w, which was 0.90 in the old runs (Table 2.1.2), and about 0.42 in the new (see earlier discussion in connection with Table 2.1.1). A series of Monte Carlo runs was performed with different degrees of correlation between ANC_w and Ca_w, using the parameters in Table 2.1.1, except that Ca_w was modelled as a rectangular distribution with limits 0 and 20 eq ha^{-1} yr^{-1} and Ca_{corr} therefore not used. This is essentially equivalent to the formulation in Table 2.1.1. The correlation coefficients between ANC_w and Ca_w used in the four runs were 0.0, 0.01, 0.42 and 0.99.

Table 2.1.5: Effect of correlation between ANC_w and Ca_w on CL(A)

Correlation coefficient (r)	0.0	0.01	0.42	0.99
Mean	0.7351	0.7399	0.7496	0.7393
SD	0.4528	0.4554	0.4519	0.4527
CV %	61.6	61.5	60.3	61.2

Values are in keq $ha^{-1}yr^{-1}$.

As Table 2.1.5 shows, the choice of correlation has no significant effect on the outcome, and is therefore not a reason for the differences between the runs. This may not of course be true for parameters to which the outcome is more sensitive. The influence of correlations between deposition parameters on critical load exceedance is investigated in Section 2.1.5 below.

2.1.4 Effects of changing input parameter uncertainty

Since CL(A) is apparently most sensitive to calcium deposition (Table 2.1.4), it was decided to try the effect of reducing the spread of the Ca_{dep} input parameter. The parameters were set up as in Table 2.1.1, except that the CV of the calcium deposition parameter was reduced from an initial 50% to 40%, 30%, 20% and 10% for successive runs. The results are shown in Table 2.1.6. It is apparent that the CV of CL(A) falls dramatically as the uncertainty in Ca deposition is reduced, as does the range, largely due to a reduction in the maximum. The mean and median remain unaffected within the limits of random errors in the Monte Carlo analysis.

Table 2.1.6: Effect of uncertainty in Ca deposition on uncertainty in CL(A)

CV of Ca_{dep}	**50%**	**40%**	**30%**	**20%**	**10%**
Mean	0.7509	0.7321	0.7460	0.7421	0.7460
Standard Deviation	0.4557	0.3977	0.3283	0.2606	0.2139
CV	**61**	**54**	**44**	**35**	**29**
Median	0.7113	0.6922	0.7036	0.7015	0.7086
Maximum	3.263	2.550	2.267	1.921	1.665
Minimum	0.0002	0.002	0.0001	0.079	0.191

Values are in keq $ha^{-1}yr^{-1}$.

As the uncertainty in base cation deposition decreases, then naturally other parameters start to assume greater importance in determining the overall uncertainty. A sensitivity analysis is shown in Table 2.1.7 below. Calcium deposition remains the biggest contributor to uncertainty until its CV falls below 20 per cent. Its place as dominant contributor is progressively taken by the critical ratio, Ca/Al_{crit}, and to a lesser extent rate of calcium uptake Ca_u, which remains more important than weathering, ANC_w.

Table 2.1.7: Effect of uncertainty in Ca deposition on percentage contributions of parameters to overall uncertainty in CL(A)

CV of Ca_{dep}	**Units**	**50%**	**40%**	**30%**	**20%**	**10%**
ANC_w	eq $ha^{-1}yr^{-1}$	2.8	3.4	2.9	6.2	8.8
BC_u	eq $ha^{-1}yr^{-1}$	<0.1	0.1	<0.1	<0.1	<0.1
Ca_{dep}	eq $ha^{-1}yr^{-1}$	81.8	78.4	64.9	42.8	14.7
Ca_u	eq $ha^{-1}yr^{-1}$	3.6	4.1	5.8	10.5	13.8
Q	m^3 $ha^{-1}yr^{-1}$	0.2	0.6	0.8	1.0	1.7
$[BC_l]$	μeq L^{-1}	<0.1	<0.1	<0.1	<0.1	<0.1
Ca/Al_{crit}	mol mol^{-1}	11.6	13.7	25.5	39.4	60.8
K_{Gibb}	m^6 eq^{-2}	<0.1	<0.1	<0.1	0.1	0.1

Although it is obvious that reducing the uncertainty of the most important input variable will reduce the overall uncertainty and increase the relative contribution of other parameters, this model experiment shows the importance of a realistic definition of uncertainty ranges for the key parameters. Even the 'optimistic' estimate of precision of deposition at a given site is a CV of 25 per cent (see Section 3.1), so Ca deposition is likely to be the key parameter at the Liphook site. In the SSMB model, the critical load is not exclusively a property of the ecosystem but depends on deposition. These results show that calcium and base cation deposition can be the most important components of uncertainty in the calculated critical loads. It would be interesting to attempt to define those ecosystems where this is true.

2.1.5 Effect of correlations between deposition parameters

Input data for this section is given in Tables A2 and A3 in Appendix A. Response variables include not only *CL(A)* but all the parameters of the critical load function (Appendix A1.2), that is, $CL_{max}S$, $CL_{max}N$, and $CL_{min}N$, and also exceedance. Exceedance is defined as deposition minus critical load (see Equation A5), and thus for the calculation of exceedance it is necessary to know the deposition of sulphur, oxidised nitrogen and reduced nitrogen. As seen in Section 2.1.4, calculation of the critical load itself requires an estimate of non-marine base cation deposition and calcium deposition. Estimates of the uncertainty in these quantities can be incorporated into the Monte Carlo analysis as explained in Appendix A. However, the values of the deposition parameters are clearly not independent of each other – sites with a high deposition of one parameter will tend to have high depositions of the others because of meteorological controls, and because a high proportion of base cation deposition in the UK is seasalt which will therefore correlate with chloride deposition.

To incorporate this correlation structure into the analysis, product-moment correlation coefficients between the modelled deposition of these parameters at a 5 x 5 km scale across the UK were calculated. The data were supplied by Ron Smith, CEH Bush, and the resulting correlations are given in Table A3. Since the forces causing spatial variation over the UK are the same as those causing variation at individual sites, this should give a reasonable approximation to an appropriate correlation structure for input parameters. Including these correlations makes a considerable difference to some calculated critical loads (Tables 2.1.8 and 2.1.9). As expected, including correlations makes no difference to the mean values calculated (within the limits of the random Monte Carlo process), but it reduces the spread of values of exceedance and of $CL_{max}S$ and $CL_{max}N$, whereas *CL(A)* and $CL_{min}N$ are unaffected. The effects on $CL_{max}S$ and $CL_{max}N$ presumably operate via the calcium and base cation deposition parameters which are correlated with deposition of the acidifying substances. The reduction in spread is considerable; the CV for $CL_{max}S$ reducing from 78 to 50 per cent, for instance. A cumulative distribution function (CDF) for exceedance is shown in Figure 2.1.1, illustrating that the spread of the calculated results becomes symmetrically narrower. The CDFs for $CL_{max}S$ and $CL_{max}N$ are very similar. The pattern of sensitivity to the input parameters also changes dramatically (Table 2.1.9). For $CL_{max}S$ and $CL_{max}N$, including the correlations almost removes the sensitivity to atmospheric parameters, including base cation deposition. For exceedance, the balance shifts from atmospheric to terrestrial parameters once deposition correlations are included, and the dependence on base cation deposition almost disappears. This work shows that intercorrelations between deposition parameters are important and should be included in analyses if the data are available to do so.

Table 2.1.8: Effect of including deposition correlations on critical loads

Param.	$CL_{max}S$		$CL_{min}N$		$CL_{max}N$		***CL(A)***		***Exceedance***	
[1]Correl.	**With**	**Without**	**With**	**Without**	**With**	**Without**	**With**	**Without**	**With**	**Without**
Mean	568	560	786	786	1353	1346	372	371	217	210
SD	285	435	88	87	265	421	193	192	340	459
CV %	50	78	11	11	20	31	52	52	-	-

[1]Values shown are from Monte Carlo analysis with or without correlations between deposition parameters. Units are eq $ha^{-1}yr^{-1}$.

Table 2.1.9: Effect of including deposition correlations on sensitivity

Param.	$CL_{max}S$		$CL_{min}N$		$CL_{max}N$		CL(A)		Exceedance	
[1]Correl.	**With**	**Without**	**With**	**Without**	**With**	**Without**	**With**	**Without**	**With**	**Without**
BC_w	7.6	4.5			10.6	5.4	25.1	25.6	6.0	4.7
BC_u	33.3	20.1	15.7	16.6	30.9	16.0	22.7	23.6	19.8	15.9
Bc_w										
Bc_u	33.3	19.7	12.9	13.9	32.6	16.3	29.3	29.5	20.1	16.0
Bc/Al_{crit}	0.9	0.5			1.4	0.7	2.4	2.1	0.9	0.6
K_{gibb}	0.5				0.8		1.2	0.8		
Q										
N_u	18.2	10.9	30.5	30.1	11.0	5.9	12.0	13.3	8.9	7.0
N_i			36.6	35.8	3.4	1.7			0.6	0.5
N_{de}			4.0	3.3	0.8					
[2]Catch	**93.8**	**55.7**	**99.7**	**99.7**	**91.5**	**46.0**	**92.7**	**94.9**	**56.3**	**44.7**
S_{dep}	1.8				2.7		1.3		8.0	9.2
NH_{4dep}									10.7	3.6
NO_{3dep}							0.5		14.1	1.3
$^*BC_{dep}$	1.6	24.2			2.1	29.3	0.6		2.1	24.8
Bc_{dep}	1.4	0.9			2.0	1.1	3.3	4.4	4.6	0.9
Cl_{dep}	0.9	18.7			1.1	23.1	0.7		3.4	15.1
[3]Dep	**5.7**	**43.8**	**0.0**	**0.0**	**7.9**	**53.5**	**6.4**	**4.4**	**42.9**	**54.9**

Units are percent contribution to uncertainty. [1]Values shown are from Monte Carlo analysis with or without correlations between deposition parameters. [2]Sum of catchment parameters. [3]Sum of deposition parameters. Values less than 0.5 are not shown – hence totals do not add exactly to 100%.

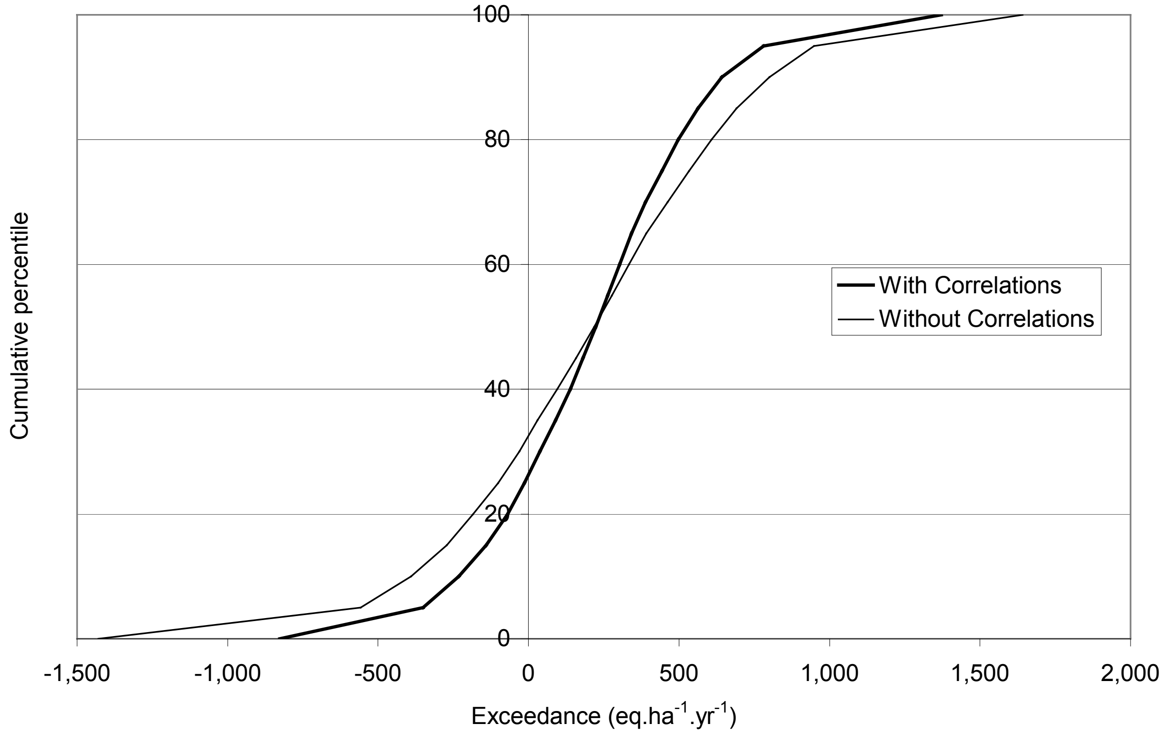


Figure 2.1.1: Cumulative distribution function for exceedance at the Liphook site, with and without correlations between deposition parameters

2.1.6 Deposition distributions

A related question when calculating exceedance is the appropriate distribution for deposition parameters. This project used both lognormal and normal distributions to represent the range of variation in deposition. The normal distribution has the disadvantage that it extends to infinity in both directions, thereby including negative values which are physically impossible. Though the lognormal distribution cannot become negative, it may generate values at the extremes which are physically close to impossible. Distributions truncated at ± 2 standard deviations have been used for both catchment and atmospheric parameters to avoid these problems, but it would be useful to have less arbitrary criteria for choosing distributions.

In previous work for the Environment Agency, J.A. Abbott performed a Monte Carlo analysis of the TRACK Model for a number of sites, and showed that for S and reduced N the model generated a distribution which was lognormal, and in which the 95th percentile was typically around 1.5 times the mean, and the mean in turn was about 1.5 times the 5th percentile. For oxidised N, the 95th percentile was typically around two times the mean, and the mean in turn was about two times the 5th percentile. A distribution with these properties can be set up for Monte Carlo analysis in Excel, and gives intuitively satisfying distributions which stay away from zero. This was done as follows:

If we call the mean of a distribution M, then for a normal distribution, the 95th percentile corresponds to M + 1.645*SD. The 5th percentile corresponds to M - 1.645*SD. If the 95th percentile is twice the mean and ln(x) is normally distributed, then

SD = ln(2)/1.645 = 0.4213.

Note that ln(2M) = ln(2) + ln(M) = 1.645*SD + ln(M). Hence, if the standard deviation of the lognormal distribution is 0.4213, the 95th percentile is always twice the mean irrespective of the mean value.

Similarly, if the 95th percentile is 1.5 times the mean, the appropriate standard deviation is ln(1.5)/1.645 = 0.2465.

Therefore, for modelled deposition, subsequent work used these distributions, that is lognormal distributions with a standard deviation of 0.2465 for S and reduced N, and lognormal distributions with a standard deviation of 0.4213 for oxidised N.

2.1.7 Terrestrial parameter distributions

The distributions for catchment parameters also need to be chosen. Usually the only basis for this is expert judgement (see Skeffington, 2006; Skeffington *et al.,* 2006), but it would be useful to investigate the sensitivity of the outcomes to different choices. A complete sensitivity analysis would obviously be impractical, so it was decided to run a Monte Carlo analysis with identical deposition where the distributions of the input parameters were: (a) all rectangular; (b) all triangular; (c) all normal; (d) all normal but truncated at two standard deviations. Means were identical, and the ranges were set to be the same for rectangular and triangular distributions and to correspond to two standard deviations for the normal and truncated normal distributions. Input parameters are shown in Tables A5 and A6, and the results in Tables 2.1.10 and 2.1.11.

Table 2.1.10: Effect of different distribution types on critical load and exceedance statistics

Param.	$CL_{max}S$				$CL(A)$			
[1]Distrib.	Rect	Triang	Norm	TNorm	Rect	Triang	Norm	TNorm
Mean	711	693	703	683	451	433	442	431
SD	241	181	216	191	187	135	165	144
CV %	34	26	31	28	41	31	37	33
10 %ile	403	460	428	441	214	259	236	246
90 %ile	1,024	929	979	938	697	610	647	618
Param.	$CL_{max}N$				$CL_{min}N$			
[1]Distrib.	Rect	Triang	Norm	Tnorm	Rect	Triang	Norm	Tnorm
Mean	1,496	1,477	1,487	1,475	785	784	785	785
SD	226	170	203	179	87	61	77	66
CV %	15	12	14	12	11	8	10	8
10 %ile	1,210	1,262	1,235	1,246	667	705	687	698
90 %ile	1,792	1,699	1,747	1,706	904	864	883	872
Param.	**Exceedance**							
[1]Distrib.	Rect	Triang	Norm	TNorm				
Mean	77	88	77	81				
SD	295	260	281	258				
10 %ile	-292	-227	-270	-236				
90 %ile	466	445	444	423				

Units are eq $ha^{-1}yr^{-1}$. [1]Distributions are rectangular, triangular, normal and normal truncated at two standard deviations.

Table 2.1.10 shows the effects of different distribution types on various critical load statistics. As would be expected, the rectangular distribution gives wider spreads (higher CVs) than the other distributions in which the central value is more probable than the extremes. For each of the critical loads, and for exceedance, the order measured by spread is rectangular > normal > truncated normal > triangular. This order applies also to the 10th and 90th percentiles. Only for the most extreme values does the normal distribution give a wider spread than the rectangular, again as would be expected since there is no constraint on the upper bound of a parameter given a normal distribution. The differences between the distributions are not, however, huge. Employing one distribution or another would not lead to qualitatively different conclusions.

The effect of different distribution types on input parameter sensitivity is shown in Table 2.1.11. Again the patterns of sensitivity are very similar between the distributions, though the order of importance changes in some cases. In particular, base cation weathering is relatively more important with rectangular distributions, and calcium uptake (Bc_u) relatively less important. For exceedance, terrestrial catchment parameters are slightly more important with the rectangular distribution than with the other distributions. Once again, the differences between different distribution types are not huge. The conclusion from this work is that although the most appropriate distribution type should be selected, the results are not very sensitive to the choice of type, at least if the extremes of the output distributions are avoided. Peaked distributions will give smaller spreads than rectangular ones.

Table 2.1.11: Effect of different distribution types on sensitivities to input parameters

	$CL_{max}S$				***CL(A)***				***$CL_{max}N$***			
[1]Distrib	**Rect**	**Trian**	**Nor**	**TNor**	**Rect**	**Trian**	**Nor**	**TNor**	**Rect**	**Trian**	**Nor**	**TNor**
BC_w	17.0	13.1	13.6	12.0	38.6	21.9	31.0	27.8	22.6	18.3	19.1	17.7
BC_u	19.8	23.4	25.1	23.6	5.9	13.5	14.8	13.9	11.8	17.3	18.8	17.3
Bc_w												
Bc_u	18.9	23.7	24.7	23.9	7.2	16.3	17.0	16.9	12.1	18.9	19.9	19.0
Bc/Al_{crit}	5.3	2.6	3.5	3.5	11.7	5.6	6.7	7.0	6.9	3.4	4.8	5.1
K_{gibb}	0.9		0.6		1.6	0.9	1.3	0.8	1.1	0.5	0.9	0.5
Q												
N_u	10.7	14.0	14.3	14.1	3.0	7.9	8.5	8.2	2.5	5.8	5.8	5.7
N_i									5.6	4.8	4.2	4.1
N_{de}									0.5		0.5	
[2]Catch	**72.6**	**76.8**	**81.8**	**77.1**	**68.0**	**66.1**	**79.3**	**74.6**	**63.1**	**69.0**	**74.0**	**69.4**
S_{dep}	10.1	8.6	6.9	8.0	7.7	6.7	5.1	6.0	13.6	11.4	9.5	10.4
NH_{4dep}					0.9			0.9				
NO_{3dep}					2.3	2.6	1.7	2.2				
$^*BC_{dep}$	6.0	4.6	3.7	4.7	3.5	2.3	1.9	2.6	7.9	6.2	5.4	6.5
Bc_{dep}	7.3	6.4	5.0	6.3	12.8	11.2	8.4	10.3	9.9	8.9	7.1	8.3
Cl_{dep}	3.4	2.6	2.1	3.0	3.6	2.5	1.9	2.9	4.5	3.5	3.1	4.1
[3]Dep	**26.8**	**22.2**	**17.7**	**22.0**	**30.8**	**25.3**	**19.0**	**24.9**	**35.9**	**30.0**	**25.1**	**29.3**

	$CL_{min}N$				**Exceedance**			
[1]Distrib	**Rect**	**Trian**	**Nor**	**TNor**	**Rect**	**Trian**	**Nor**	**TNor**
BC_w					10.1	6.0	7.6	6.9
BC_u	16.5	17.2	17.8	17.8	10.7	7.4	9.1	8.2
Bc_w								
Bc_u	13.4	13.8	14.3	14.0	12.0	7.5	10.0	9.0
Bc/Al_{crit}					2.3	1.5	1.8	1.8
K_{gibb}								
Q								
N_u	35.8	35.5	30.3	30.8	4.4	3.0	4.1	3.3
N_i	30.0	29.8	34.0	33.2	1.0	0.8	0.7	0.7
N_{de}	3.9	3.7	3.5	3.7				
[2]Catch	**99.6**	**100.0**	**99.9**	**99.5**	**40.5**	**26.2**	**33.3**	**29.9**
S_{dep}					10.2	12.3	10.6	11.7
NH_{4dep}					14.1	18.4	16.8	17.1
NO_{3dep}					18.5	24.3	22.9	23.0
$^*BC_{dep}$					3.9	4.0	3.6	4.0
Bc_{dep}					6.5	8.0	6.5	7.4
Cl_{dep}					5.4	6.2	5.7	6.1
[3]Dep	**0.0**	**0.0**	**0.0**	**0.0**	**58.6**	**73.2**	**66.1**	**69.3**

Units are percent contribution to uncertainty. [1]Values shown are from Monte Carlo analysis with varied distribution types. [2]Sum of catchment parameters. [3]Sum of deposition parameters. Values less than 0.5 are not shown – hence totals do not add exactly to 100%.

2.1.8 Comparisons of different sites

After extensive investigation of the Liphook site, work was extended to other coniferous forest sites: Thetford in Eastern England (Grid Ref. 5954 2382) and Aber in North Wales (Grid Ref. 2675 3505). These are research sites with contrasting attributes and deposition regimes. Monte

Carlo analysis was carried out as before. The catchment input parameters are shown in Table A5, and the deposition parameters in Table A6. As discussed above, the input parameters are crucial to the outcome. Both Barkman and Alveteg (2001) and Skeffington (2006) proposed that input parameters should have a 'pedigree' attached to them, to distinguish the qualities of input data. In these cases, the pedigree would be unusually good as these are research sites, but nevertheless some are expert judgement by the authors with no supporting data. This applies to N_{de} and N_i at Liphook and Thetford, BC_w at Liphook, and Ca_{corr} and N_u at Thetford. Some are based on expert judgement by several experts, such as Bc/Al_{crit}, which is effectively Ca/Al_{crit}, (Cronan and Grigal, 1995), and $K_{gibb,}$ which is based on the range for the relevant soil type in UBA (2004). Some are based on measurements, such as N_{de} and N_i at Aber, and BC_u and N_u at Liphook and Aber. Some are based on modelling supported by measurements, such as BC_w at Thetford and Aber which were derived from modelling with PROFILE (Warfvinge and Sverdrup, 1992). Modelling supported by measurement was also used to generate all the values of *Q* and deposition at Liphook and Aber. Deposition at Thetford was purely modelled, and the distribution used is described in Section 2.1.6. The choice of distribution type was mostly expert judgement, and correlations were derived from measurements at Liphook and used at the other sites. One hypothesis to be tested in this work was that the relative sensitivity of the critical loads to different input parameters would remain about the same. This would mean that research to narrow these uncertainties could be targeted at only a few quantities. Another was to compare the ranges of uncertainty in critical loads and exceedances at the three sites to determine whether the different conditions involved led to different conclusions. Uncertainty statistics are shown in Table 2.1.12. Cumulative distribution functions for $CL_{max}S$, *CL(A)*, and exceedance are shown in Figures 2.1.2 and 2.1.3.

Table 2.1.12: Uncertainty statistics from three coniferous forest sites

	Liphook			**Thetford**			**Aber**		
Param.	**Mean**	**SD**	**CV (%)**	**Mean**	**SD**	**CV (%)**	**Mean**	**SD**	**CV (%)**
$CL_{max}S$	568	285	50	11,580	4,054	35	1,883	518	27
$CL_{min}N$	786	87.6	11	353	67.1	19	431	22.5	5
$CL_{max}N$	1,353	265	20	11,930	4,053	34	2,315	515	22
CL(A)	372	193	52	11,660	4,052	35	1,804	474	26
Exceed	217	340	-	-8,409	4,138	-	1,284	612	-

Units are eq $ha^{-1}yr^{-1}$.

The lowest CVs at each site are for $CL_{min}N$, which requires only three parameters – they are particularly low at Aber where uncertainty in the input parameters is reduced by the availability of measurements. At Liphook and Aber, the CV of $CL_{max}N$ is lower than that of $CL_{max}S$ and *CL(A)* even though $CL_{max}N$ has more parameters. This is observed in most studies, and is probably due to a 'compensation of errors' mechanism. At Thetford, the uncertainty in the large weathering rate term dominates all other input uncertainties, and the uncertainty is the same for $CL_{max}N$, $CL_{max}S$ and *CL(A)*. At all sites, the uncertainty in the output terms is less than that in most of the input terms. The cumulative distribution functions for *CL(A)* and $CL_{max}S$, as seen in Figure 2.1.2, cover a wide range and have long tails. The range is greater for Aber than Liphook, though the absolute value is greater at Aber.

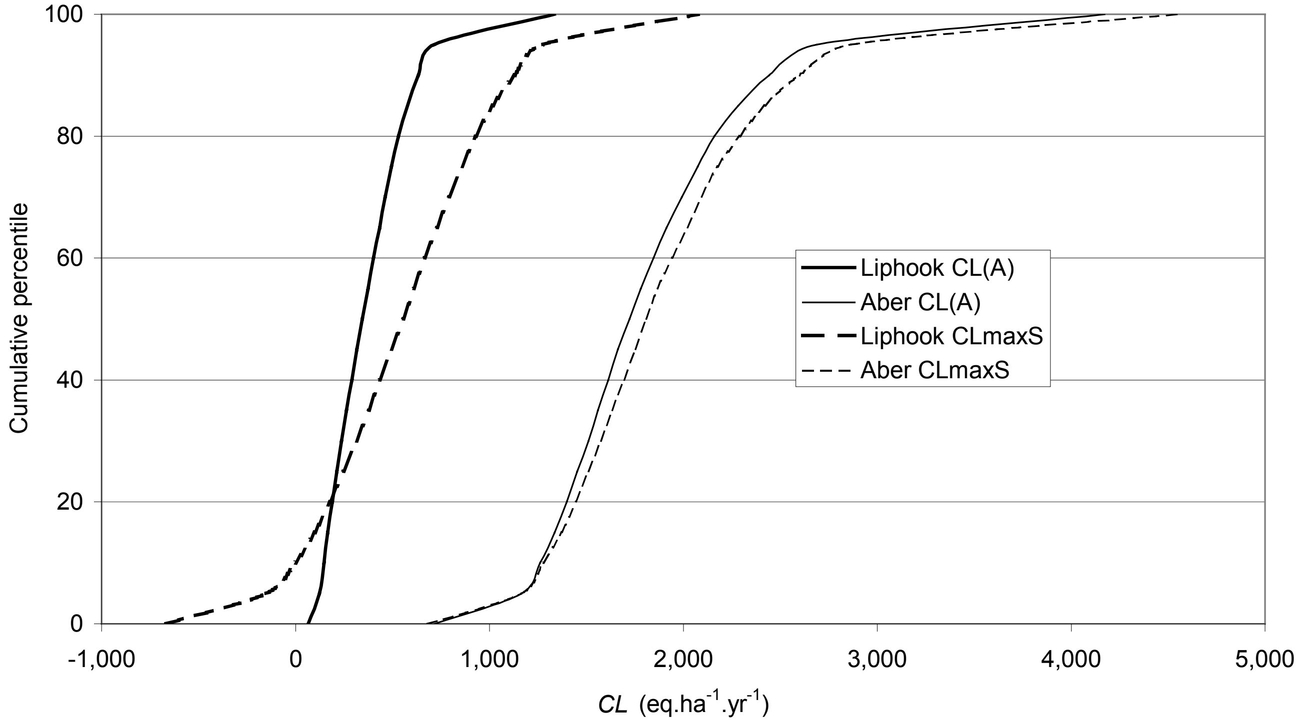


Figure 2.1.2: Cumulative distribution function for *CL(A)* and $CL_{max}S$ at Liphook and Aber

The cumulative distribution function for exceedance is shown in Figure 2.1.3.

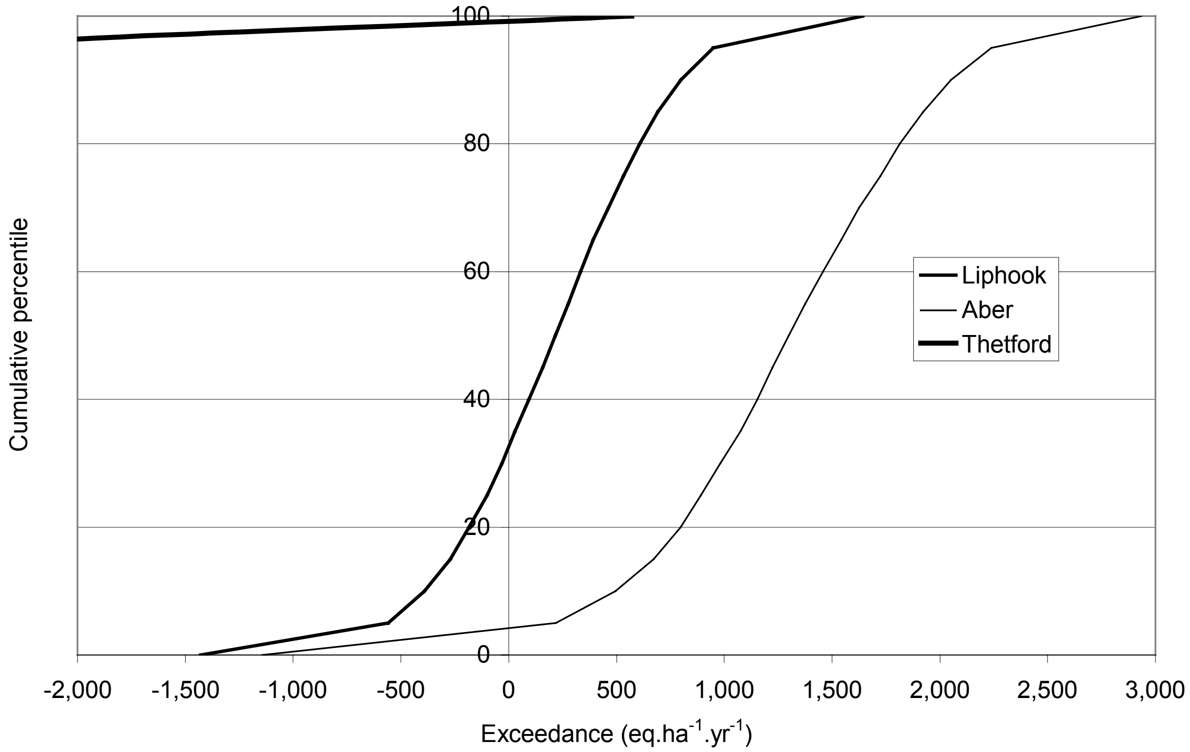


Figure 2.1.3: Cumulative distribution function for exceedance at three coniferous forest sites

There is a finite probability of exceedance at all three sites: about 95 per cent at Aber, 68 per cent at Liphook and two per cent at Thetford. It is noteworthy that a few Monte Carlo runs cause exceedance at Thetford even though the limestone geology produces a very large critical load. To demand very high probabilities for non-exceedance as a precautionary regulatory criterion would thus seem to be unwise. The long tails on these graphs imply that it would be expensive or difficult to achieve very high probabilities of protection, even just taking into account parameter uncertainty.

Table 2.1.13 shows the sensitivity of the output parameters to the input parameters at the three different sites. Different input parameters are important at different sites. At Thetford, a calcareous site, uncertainty in the weathering rate term dominates and only the critical ratio Bc/Al_{crit} is also important. At Liphook, base cation deposition and uptake have the most influence, with weathering in a secondary role. At Aber, Bc/Al_{crit} is most important, followed by base cation and calcium deposition and also K_{gibb}, which does not appear elsewhere. Total chloride deposition was measured only at Liphook, and is important there for $CL_{max}N$ and $CL_{max}S$, but not $CL(A)$. At least one output parameter is sensitive (> 0.5%) to every input parameter in Table 2.1.13 except Q. It is easy to imagine that where good estimates of run-off are not available, Q would feature too. Uncertainty in deposition is immaterial to all output parameters at Thetford, whereas at Aber it is as important as the terrestrial parameters to $CL_{max}S$ and $CL_{max}N$ and more important to exceedance. $CL_{min}N$ is sensitive to the three nitrogen sink variables, the sensitivity being more or less proportional to their means and ranges. The apparent sensitivity to base cation and calcium uptakes is an intercorrelation effect, as these parameters are correlated with N uptake. This also accounts for the apparent influence of N uptake on $CL_{max}S$ and $CL_{max}N$. Aber is a managed forest on an organo-mineral soil, and the UK critical load methodology (Hall *et al.*, 2004) requires the applications of ground rock phosphate typically received by such sites to be taken into account. This is shown as "fertiliser" in Table 2.1.13, and contributes about 50 per cent more uncertainty than base cation weathering, which is the other source of base cations at Aber.

Uncertainty analysis can be used to direct research toward narrowing uncertainties in the most influential input parameters, but these results indicate that the importance of input parameters varied with site, even with this restricted sample of three contrasting sites. Potentially, any parameter could have an important influence. It may be that there are classes of site in which there are similar patterns of importance, but to reveal this would require the analysis of more sites. It is clear that at some sites, perhaps those like Aber where base cation deposition is high compared to the weathering rate, a good estimate of base cation deposition is essential for accurate critical load estimation (quite apart from the estimate of exceedance). The issue of whether there is a class of ecosystems where critical loads are more likely to be sensitive to base cation deposition than weathering rates is explored further in Section 3.2. The same three sites were used for a comparison of national and site-specific input data (Section 4) and to estimate uncertainty distributions to be applied to coniferous forest sites in the whole of South East England (Section 5).

Table 2.1.13: Sensitivity to input parameters at three different sites

Param.	***$CL_{max}S$***			***$CL_{min}N$***			***$CL_{max}N$***			***CL(A)***			***Exceedance***		
Site	Lip.	Thet.	Aber	Lip.	Thet.	Aber	Lip.	Thet.	Aber	Lip.	Thet.	Aber	Lip.	Thet.	Aber
BC_w	7.6	73.3	0.8				10.6	73.3	0.8	25.1	73.3	1.2	6.0	71.0	0.8
Fertiliser			1.3						1.3			2.0			1.2
BC_u	33.3		5.5	15.7	25.2	14.9	30.9		4.9	22.7		6.3	19.8		4.0
Bc_w															
Bc_u	33.5		6.6	12.9	20.1	11.5	32.6		6.1	29.3		8.0	20.1		4.8
Bc/Al_{crit}	0.9	26.0	22.2				1.4	26.0	22.7	2.4	26.0	32.9	0.9	25.0	20.1
K_{gibb}	0.5		5.6				0.8		5.8	1.2		8.5			5.4
Q												0.5			
N_u	18.2		3.6	30.5	45.3	28.8	11.0		2.9	12.0		4.0	8.9		1.5
N_i				36.6	4.7	42.5	3.4						0.6		
N_{de}				4.0	4.7	2.3	0.8								
[1]Catch	**94.0**	**99.3**	**45.6**	**99.7**	**100.0**	**100.0**	**91.5**	**99.3**	**44.5**	**92.7**	**99.3**	**63.4**	**56.3**	**96.0**	**37.8**
S_{dep}	1.8		6.6				2.7		6.8	1.3		3.9	8.0		19.3
NH_{4dep}			3.0						3.1			1.8	10.7	1.0	21.1
NO_{3dep}			6.6						6.7	0.5		3.3	14.1	2.5	18.2
$*BC_{dep}$	1.6		16.9				2.1		17.3	0.6		10.3	2.1		3.3
Bc_{dep}	1.4		20.7				2.0		21.2	3.3		17.3	4.6		
Cl_{dep}	0.9						1.1			0.7			3.4		
[2]Dep	**5.7**	**0.0**	**53.8**	**0.0**	**0.0**	**0.0**	**7.9**	**0.0**	**55.1**	**6.4**	**0.0**	**36.6**	**42.9**	**3.5**	**61.9**

Units are percent contribution to uncertainty. [1]Sum of catchment parameters. [2]Sum of deposition parameters. Values less than 0.5 are not shown – hence totals do not add exactly to 100%.

2.1.9 Critical loads for heathland and unmanaged woodland

The SSMB model can also be used to calculate critical loads for ecosystems other than coniferous forest. Managed deciduous forest is calculated in the same way, with different values for base cation and nitrogen uptake, and hence is unlikely to yield different results. Unmanaged forest is calculated using the SSMB, but without the uptake terms. Heathland and grassland could be calculated in the same way, but in the UK an empirical method is used in which a critical load is ascribed depending on the weathering rate of the dominant soil type, modified by various factors (see Hall *et al.*, 2003a, 2004a for methods). It was therefore decided to apply both the modified SSMB (without uptake terms) and the empirical method to a heathland site. This could serve as a comparison of methods, but also the SSMB could stand as surrogate for an unmanaged forest. The site chosen was the CEH climate change research site (Climoor) near Corwen in North Wales (Grid Ref. 3015 3515). Input parameters were as shown in Tables A7 and A8, and were based on CEH data. The crucial BC_w parameter is based on the range of measurements of weathering rates of similar soil types in Wales using Ti/Zr methods, and of calculations using PROFILE (Warfvinge and Sverdrup, 1992) on the same soils. Critical load statistics are shown in Table 2.1.14, and a sensitivity analysis in Table 2.1.15.

Table 2.1.14: Uncertainty statistics for a heathland site at Climoor by two different methods

	SSMB method			**Empirical method**		
Param.	**Mean**	**SD**	**CV (%)**	**Mean**	**SD**	**CV (%)**
$CL_{max}S$	1,378	379	28	690	254	30
$CL_{min}N$	284	66	23	284	66	23
$CL_{max}N$	1,662	386	23	974	262	27
CL(A)	1,239	368	30	550	252	46
Exceed	1,006	705	-	1,690	723	-

Units are eq $ha^{-1}yr^{-1}$.

The mean critical loads using the empirical method are lower than the SSMB Method, because each critical load is aimed essentially at protecting different aspects of the ecosystem (base saturation for the empirical method, Ca/Al ratio for the SSMB method). Exceedance is correspondingly higher for the empirical method. The standard deviation of the empirical method is lower than the SSMB, because the uncertainty of the method depends on uncertainty in only one parameter (BC_w) for *CL(A)* and two (BC_w and BC_{dep}) for $CL_{max}S$ and $CL_{max}N$. The CVs tend, however, to be slightly higher because of the lower means of the empirical method.

The empirical method is applied somewhat differently to the SSMB, in that the dominant soil within each kilometre square is allocated to one of five critical load classes, based on the mineralogy and weathering rate. Each class represents a range of critical load values (< 0.2, 0.2-0.5, 0.5-1.0, 1.0-2.0, > 2.0 keq $ha^{-1}yr^{-1}$). This method is applied to all squares dominated by mineral or organo-mineral soils. For the calculation of exceedances, the midpoint critical load value of each class is used. The Monte Carlo analysis can thus be used to ask what the probability is of a site being allocated to one or other of these classes. For Climoor, 10% of the *CL(A)* values would lie in the lowest class and be given a critical load of 100 eq $ha^{-1}yr^{-1}$; 34% in the next lowest class (350 eq $ha^{-1}yr^{-1}$); and 56% in the next lowest class (750 eq $ha^{-1}yr^{-1}$).

Acidity critical loads for squares dominated by peat soils are based on a version of the SSMB using soil pH as the chemical criterion (see Hall *et al*, 2003a, 2004a for methods) – these are not covered by this report.

Table 2.1.15: Sensitivity analysis for a heathland site at Climoor by two methods

Parameter	***$CL_{max}S$***		***$CL_{min}N$***		***$CL_{max}N$***		***CL(A)***		***Exceedance***	
Method	**SSMB**	**Emp.**	**SSMB**	**Emp.**	**SSMB**	**Emp.**	**SSMB**	**Emp.**	**SSMB**	**Emp.**
BC_w	50.8	95.4			49.2	90.7	58.3	99.8	8.3	5.6
Bc_w	0.6				0.5		0.7			
Bc/Al_{crit}	19.1				18.8		21.8		3.0	
K_{gibb}	3.7				3.7		4.2		0.6	
Q										
N_i			90.4	90.8	2.6	5.1			0.5	
N_{de}			9.5	8.9		0.5				
[1]Catch	**74.2**	**95.4**	**99.9**	**99.7**	**74.8**	**96.3**	**85.0**	**99.8**	**12.4**	**5.6**
$*S_{dep}$	3.2	0.5			3.2		1.8		25.3	24.9
$*BC_{dep}$	7.8	1.6			7.6	1.3	4.5		12.6	15.6
Bc_{dep}	7.8	1.5			7.5	1.2	4.5		12.2	15.3
NO_{3dep}	4.8	0.7			4.7	0.6	2.8		23.4	24.8
NH_{4dep}	1.8				1.8		1.1		13.6	13.3
[2]Dep.	**25.4**	**4.3**	**0.0**	**0.0**	**24.8**	**3.1**	**14.7**	**0.0**	**87.1**	**93.9**

Units are percent contribution to uncertainty. [1]Sum of catchment parameters. [2]Sum of deposition parameters. Values less than 0.5 are not shown – hence totals do not add exactly to 100%.

Table 2.1.15 shows a sensitivity analysis for heathland at the Climoor site using both methods. As expected, *CL(A)* for the empirical model is exclusively sensitive to uncertainty in BC_w, whereas the SSMB is sensitive to other parameters, though BC_w is still the most important. The absence of the uptake parameters emphasises the importance of BC_w (compare Aber and Liphook, Table 2.1.13, where the uptake parameters dominate). However, the most striking result in Table 2.1.15 is that for both methods, uncertainty in deposition makes a much greater contribution to exceedance than uncertainty in the terrestrial parameters. This is similar to Aber (Table 2.1.15), another site with a high deposition load, though the magnitude of the effect at Climoor is surprising.

Critical loads for unmanaged woodland at Climoor (if there were any) would be calculated in the same way as the SSMB results in Tables 2.1.14 and 2.1.15. The above discussion can also be taken as a comparison between critical loads for heathland and unmanaged woodland on the same site. Heathland critical loads will always be lower than woodland critical loads calculated by this method, but the patterns of sensitivity may vary from site to site in the same way as coniferous woodland critical loads.

2.1.10 Critical loads for nutrient nitrogen

UNECE recommend two approaches to calculating critical loads for nutrient nitrogen (see UBA, 2004). The first is a steady state mass balance approach 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 critical load is thus the sum of nitrogen uptake, immobilisation, denitrification, and the acceptable nitrogen leaching, as explained in Appendix A. In the UK, this method is applied to managed woodlands (coniferous and deciduous) only. The second is an empirical approach, in which critical loads are estimated for different ecosystems based on experimental or field evidence of thresholds for change in species composition, plant vitality or soil processes. Empirical critical loads come with a rough estimate of reliability (in increasing order, ‘expert judgement’, ‘quite reliable’ and ’reliable’) and estimating uncertainty for these would involve quantifying these terms. There seems no objective way to do this, and hence this section is concerned only with quantifying the uncertainty in the mass balance approach as applied to woodlands. Liphook and Aber were chosen as example sites, and input parameters were as shown in Table A9. Uncertainty statistics are shown in Table 2.1.16.

Table 2.1.16: Uncertainty statistics for nutrient nitrogen critical loads at two sites

Site	Liphook			Aber		
Param.	Mean	SD	CV (%)	Mean	SD	CV (%)
$CL_{nut}N$	1071	152	14	638	132	21
Exceed	-109	287	-	1614	579	-

Units are eq $ha^{-1}yr^{-1}$.

Critical loads for nutrient nitrogen at Liphook are higher because of higher uptake and immobilisation values. Exceedance is much greater at Aber because of the lower critical load and much higher deposition of ammonium and nitrate (Table A10). The coefficients of variation are quite low at 14 and 21 per cent – lower at Liphook largely because of the higher critical load. Cumulative distribution functions (CDFs) for $CL_{nut}N$ and exceedance are shown in Figures 2.1.4 and 2.1.5. As seen with the acidity critical loads, the range is quite wide and the tails of the distributions are long. The nutrient nitrogen critical load can be said to be exceeded at Aber with 100 per cent confidence (Figure 2.1.5). At Liphook, although the mean shows no exceedance, the CDF indicates a 30 per cent chance that the critical load is, in fact, exceeded. Also shown on Figure 2.1.5 are cumulative normal distributions with the same mean and standard deviation as the experimental data. The cumulative normal distribution appears to be a reasonably close fit: closer at Liphook than Aber. This may be a manifestation of the Central Limit Theorem, and could be useful for modelling exceedances.

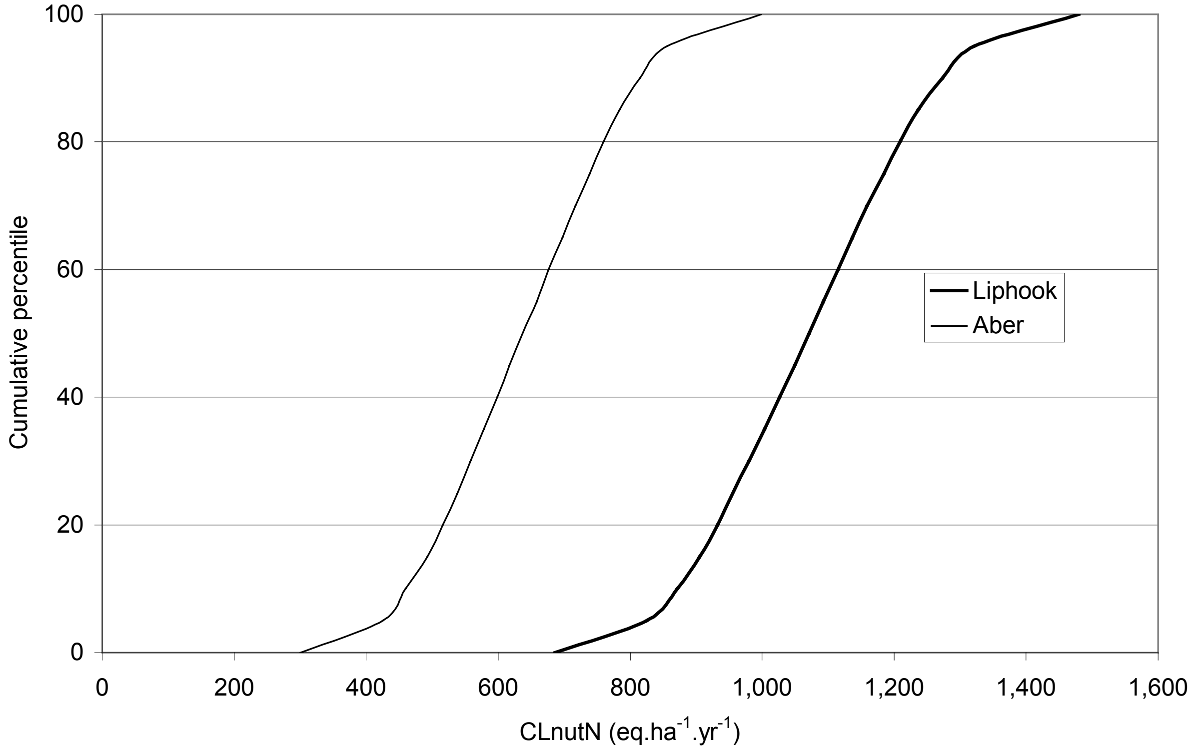


Figure 2.1.4: Cumulative distribution functions for $CL_{nut}N$ at Liphook and Aber

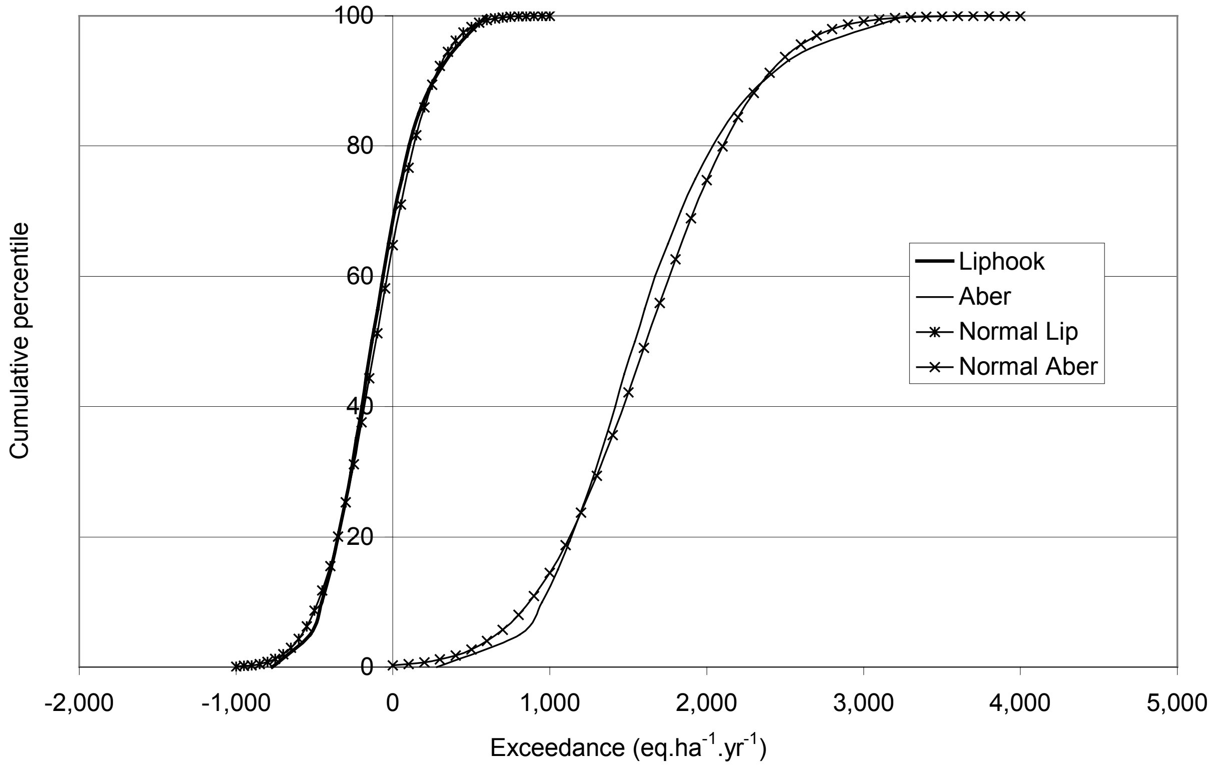


Figure 2.1.5: Cumulative distribution functions for exceedance of $CL_{nut}N$ at Liphook and Aber

Table 2.1.17 shows a sensitivity analysis for $CL_{nut}N$ and exceedance at Liphook and Aber.

Table 2.1.17: Sensitivity analysis for nutrient N critical loads at two sites

Parameter	***$CL_{nut}N$***		***Exceedance***	
Method	**Liphook**	**Aber**	**Liphook**	**Aber**
N_u	12.9	18.9	5.0	1.4
N_i	13.4	2.0	4.8	
N_{de}	2.7	1.6	1.2	
$N_{le(acc)}$	70.8	77.4	25.2	5.5
[1]Catch	**99.8**	**99.9**	**36.2**	**6.9**
NO_{3dep}			42.5	61.8
NH_{4dep}			21.3	31.1
[2]Dep.	**0.0**	**0.0**	**63.8**	**92.9**

Uncertainty in acceptable N leaching dominates the uncertainty in $CL_{nut}N$. For exceedance, deposition parameters are more important, especially at Aber where deposition is larger and the critical load is smaller.

2.1.11 Conclusions (terrestrial ecosystems)

- The results for terrestrial ecosystems reflect a lot of experimentation with methodology and development of the most appropriate techniques and parameters for estimating uncertainties.
- In agreement with previous work, output parameter uncertainties have a universal tendency to be narrower than input uncertainties, presumably due to a 'compensation of errors' mechanism.
- Equally reasonable parameter sets for the same sites can lead to somewhat different conclusions in terms of magnitudes of output uncertainties and relative importance of various input parameters.
- As an example of the above, the CV of the critical load for acidity at Liphook was 37 per cent in the previous work for the Environment Agency (which used measured parameters), but 60 per cent with revised (national default) parameters. In previous work, weathering was the most important input parameter, but with the revised parameters it was calcium deposition.
- This highlights the importance of a good estimate of calcium and base cation deposition for critical load calculation.
- Calcium deposition is likely to be more important than calcium weathering at sites with low critical loads. This is borne out by the national-scale analysis (Section 3.2).
- Intercorrelation between input parameters must be considered. Incorporating correlations between deposition parameters reduced the spread of the results and the influence of deposition uncertainty on output parameter uncertainty, especially for $CL_{max}S$ and $CL_{max}N$. In contrast, the results for the Liphook site were not sensitive to correlation between the weathering rate parameters.
- Choice of an appropriate statistical distribution to represent deposition proved to be difficult. A particular form of the lognormal distribution gave good results for modelled deposition, but other distributions were also used.
- The results were only somewhat sensitive to choice of distribution type for terrestrial parameters (from rectangular, triangular, normal and truncated normal). This is useful, because there is rarely any basis other than expert judgement to select terrestrial parameter distributions.
- Application of the SSMB model to three contrasting coniferous sites showed that virtually any input parameter could be significant in determining output parameter uncertainty.

Thus, it is not easy to target research towards narrowing uncertainties except on a site-specific basis.

- However, several runs show that on-site measurements can be used to narrow uncertainties. This could be useful at valuable or disputed sites.
- Output distributions typically seem to have long tails, and look roughly normal, although this has been tested only on a few runs. The long tails on these graphs imply that it would be expensive or difficult to achieve very high probabilities of protection, even just taking into account parameter uncertainty.
- Uncertainties in critical loads at a heathland site, calculated according to the UK empirical method, show an exclusive dependence on weathering rate. Uncertainties in exceedance depend much more on deposition. This is a high deposition site, so this pattern may be different elsewhere.
- These empirical critical loads are normally expressed as one of five classes rather than in numerical form. The site had a 10 per cent probability of being in the lowest class, 34 per cent in the next lowest and 56 per cent in the next. These results could be useful for further analysis.
- Critical loads for heathland can also be calculated by the SSMB method, and this is the method used for unmanaged woodland. Here the base cation:aluminium ratio is important as well as weathering rate, and base cation deposition also features. The critical loads are higher than the empirical method, and the spread (CV) slightly lower, except for exceedance.
- The critical load for nutrient N has been calculated by the mass balance method for two example sites. The CVs were quite low, and the method at these sites was most sensitive to the definition of acceptable nitrogen leaching. For exceedance, deposition parameters were more important, at the high deposition – low critical load site.

2.2 Uncertainty in critical loads for freshwaters

2.2.1 Introduction

The First Order Acidity Balance Model (FAB) model is now the method of choice in the UK for calculating freshwater critical loads. The model is derived from a combination of charge and mass balance approaches as described in Posch *et al.* (1997), Henriksen and Posch (2001) and UBA (2004). The implementation of the model in the UK is described in Hall *et al.* (1998, 2003ab, 2004a). The critical load criterion is acid neutralising capacity (ANC). The most common criterion value is 20 µeq L^{-1}, and although the UK originally used zero, it now uses 20 µeq L^{-1} except where there is evidence that the pre-industrial ANC was less than this (Hall *et al.,* 2004a: Section 5). The essential idea of the FAB model is that catchments should be able to supply enough ANC to maintain at least the criterion ANC concentration indefinitely. Various routines and assumptions are used to calculate the sinks of deposited S and N in terrestrial catchments and lakes (if any), and the base cation supply from catchment and atmosphere. From this, the combinations of S and N deposition which result in the water draining the catchment meeting the critical load (that is, the critical load function, Appendix A) can be calculated.

The FAB model and its implementation in this study is described in detail in Appendix A. FAB is the most complex of the critical load models, with 18 input parameters and dependent on a complex set of assumptions and empirical equations derived from different parts of the world. This might thus be expected to be reflected in the overall model uncertainty.

2.2.2 Application of FAB to the UK Acid Waters Monitoring Network

It was decided to use as test data the UK Acid Waters Monitoring Network (AWMN). The data were kindly supplied by Dr Chris Curtis of ENSIS at the Department of Geography, University College London. The AWMN has been in operation since 1988 and consists of 22 sites chosen to represent a range of acid-sensitive habitats. Stream sites are sampled 12 times a year, and lake sites four times. The geographical distribution of sites is shown in Figure 2.2.1. Using the AWMN has the advantage that:

- there is a long run of quality-assured data;
- the habitats are all potentially acid-sensitive;
- the network deliberately includes a variety of habitats, such as lakes and streams, non-forested, coniferous and broad-leaved forest, uplands and lowlands;
- the data required for FAB were all available.

The AWMN is described in a recent Special Issue of *Environmental Pollution* (see Monteith and Evans, 2005). A detailed description of how FAB was implemented for Monte Carlo analysis will be found in Appendix A.

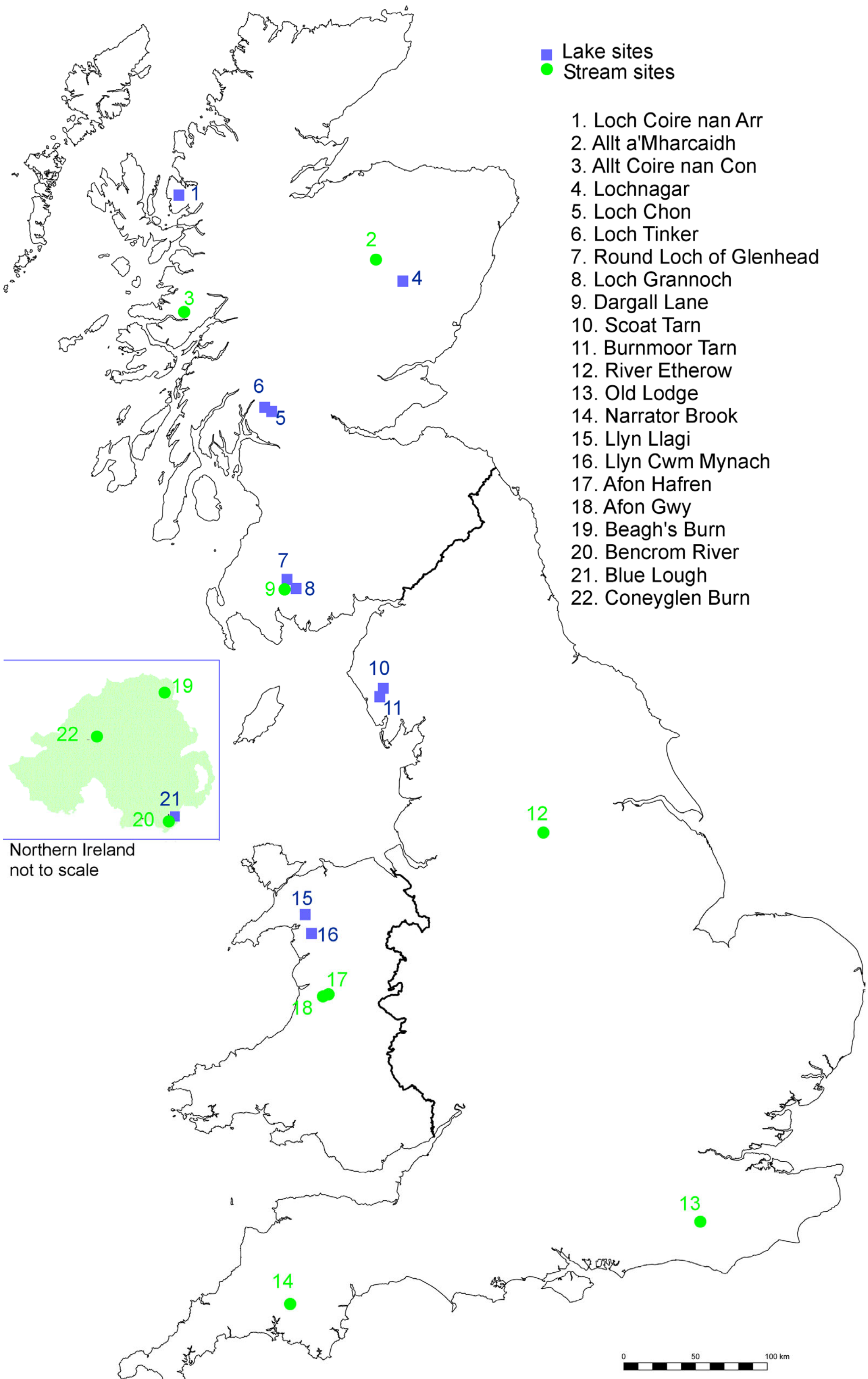


Figure 2.2.1: The UK Acid Waters Monitoring Network

2.2.2.1 Choice of parameter values

The results of Monte Carlo analysis are heavily influenced by the choice of parameter distributions and ranges. In this section we discuss the reasons for these choices. The values chosen for the parameters are shown in Tables 2.2.1 to 2.2.2.

Run-off (*Q*)
It was assumed that run-off in ungauged catchments is known to ± 10 per cent. Run-off is a function of precipitation and evapotranspiration, both reasonably well known in the UK, though some catchments may lose or gain water underground. Some catchments in the AWMN are in fact gauged (such as the Gwy and the Hafren) and here the run-off is known to within two to five per cent, but the 10 per cent spread was kept the same for this exercise for comparative purposes. A triangular distribution was used, reflecting lack of knowledge of the distribution type but a feeling that the central value was more likely. This is the reason for use of all the triangular distributions below. The spread of values for run-off does not include year-to-year variation.

Catchment area (*A*)
Catchment areas are derived from digital terrain models (DTMs) which are now accurate to a few metres. It is well known, however, that topographic catchments may not coincide with hydrological ones, and in gently sloping terrain even topographic catchment may be hard to define exactly. It was thought, however, that the use of DTMs allowed ± 5 per cent error on overall catchment area, with a triangular distribution.

Lake area (A lake)
Lake area is known with reasonable accuracy, as lake boundaries are easy to identify from aerial and satellite photographs. Lake areas fluctuate somewhat with hydrological conditions (though not by much in the case of these upland lakes). Once again, an error of ± 5 per cent with a triangular distribution was thought appropriate.

Vegetation areas
The areas of managed and unmanaged coniferous and broad-leaved woodland are used as inputs to the FAB calculations. These are identified from national databases as explained in Section 4, except that for the AWMN catchments local knowledge is also used. Even with local knowledge there is some uncertainty over definitions, so it was thought appropriate to use ± 10 per cent with a triangular distribution as the uncertainty bounds for all types of vegetated areas in these catchments. This is perhaps a little generous, and the bounds would be wider for catchments with no local data.

Nitrogen uptake (N_u)
Nitrogen uptake was parameterised at the UK default values of 0.42 keq $ha^{-1}yr^{-1}$ for managed broadleaves and 0.21 keq $ha^{-1}yr^{-1}$ for managed conifers. Nitrogen uptake by real forests will vary quite a lot – it is not known how much, though the data probably exist to make an estimate. Since there is no reason to believe the central value is any more accurate than the extremes, a rectangular distribution was used with a spread of ± 50 per cent on the default values, as employed in work on the SSMB equation (see above).

Nitrogen immobilisation (N_i) and denitrification (N_{de})
Nitrogen immobilisation and denitrification are both calculated from area-weighted means of default values attached to different soil types. Errors thus arise in assignment of the default values to the soil types, and identification of the soil types on the ground. There seems little or no objective way to judge the accuracy and precision of these estimates, especially N_i which is the long-term sustainable value rather than the present value, which is likely to be much greater. As with N_u, a rectangular distribution was used for N_i and N_{de} with a spread of ± 50 per cent on the default values. This is fairly arbitrary.

Mass transfer coefficients (S_N and S_S)
The mass transfer coefficients are used to calculate in-lake S and N retention. The original authors calculated a range for these values (reproduced in UBA, 2004) based on studies in various lakes. The range was 2 to 8 m s^{-1} for S_N, and 0.2 to 0.8 m s^{-1} for S_S. Rectangular distributions with these limits were therefore used.

Pre-industrial sulphate calculation (*a* and *b*)
Pre-industrial non-marine sulphate ($*SO_4^{2-}{}_0$) is calculated from present day non-marine base cation concentrations ($*BC_t$) by an equation of the form ($*SO_4^{2-}{}_0$) = a + b$*BC_t$. The parameters *a* and *b* are derived from empirical observations of a range of lakes with differing current S depositions. Henriksen and Posch (2001) give a number of different equations where *a* ranges from 8 to 19 µeq L^{-1} and *b* from 0.08 to 0.17. These values were therefore used except where the current non-marine sulphate concentration was less than 8 µeq L^{-1}, when the range of *a* was scaled down to 5 to 12 µeq L^{-1}. Rectangular distributions were once again used.

Present-day observed concentrations ($*SO_4^{2-}{}_t$, $*BC_t$ and $NO_3^-{}_t$)
Present day concentrations are used in FAB to estimate the pre-industrial base cation weathering and deposition rates, as explained above. The analytical error on a single measurement of these parameters is quite small (no more than five per cent). However, since streams and lakes are naturally variable, and a single measurement can be used to estimate the critical load (and is, for the majority of UK sites), it was thought appropriate to use this variability as the uncertainty estimate. As each site in the AWMN has 13 annual mean values at present, the mean and standard deviation of these annual means was calculated for each of these parameters. Use of the individual values would have led to somewhat greater standard deviations. Visual inspection suggested the distributions were close to normal. For input to the uncertainty analysis, the 13-year mean was thus used as a central value, with a normal distribution with the observed standard deviation. However, since normal distributions extend to ± infinity, negative concentration values are clearly meaningless, and there are chemical limits which constrain the possible range of concentrations; it was thus decided to truncate the distributions at two standard deviations (or at zero if this was less than two standard deviations from the mean). If the distribution was truly normal, this would comprise 95 per cent of the distribution in any case, and truncating prevents unrealistic extreme values from biasing the results. The uncertainty range on these parameters, therefore, is the uncertainty which might be expected if only one year was sampled.

Correlations
The correlation structure of the input variables is an important factor in computing the overall uncertainty. Correlations between variables are likely to be restricted to the three concentration variables in the paragraph above. Product-moment correlations were computed separately for each site and used in the analysis if they were greater than 0.1. The correlations used are shown in Tables 2.2.1 and 2.2.2.

Deposition (N_{dep} and S_{dep})
Nitrogen deposition is required to decide which formulation of $CL_{max}N$ to use, and both depositions are required to compute exceedance. Deposition is taken from the national database for the years 1998-2000. Pending the outcome of discussions about the most appropriate distribution to use, lognormal distributions with 50 per cent coefficients of variation were employed. As with the observed concentrations, the distribution was truncated at the upper end at two standard deviations (the lower limit is of course zero). Deposition is often lognormally distributed over time, but whether this should extend to uncertainty distributions is a moot point.

2.2.2.2 Results of the uncertainty analysis

The means and coefficients of variation for various critical loads are shown for all the AWMN sites in Tables 2.2.3 and 2.2.4. These comprise the maximum critical load for sulphur, $CL_{max}S$, the maximum critical load for nitrogen, $CL_{max}N$, and the minimum critical load for nitrogen, $CL_{min}N$. The critical ANC leaching L_{crit} is also shown. These parameters are defined in Appendix A1.2. Means and standard deviations of exceedance are also shown, except for sites in Northern Ireland where deposition values were not available.

A sensitivity analysis, showing the percentage contribution of each input parameter to variance in the estimates of each output parameter, is given in Tables 2.2.5 to 2.2.12.

Finally, the 10^{th}, 50^{th} and 90^{th} percentiles for exceedance are shown for each site in Tables 2.2.13 and 2.2.14. The tables also show the percent confidence that the critical load for acid deposition at the site is not exceeded. For two sites, Scoat and Burnmoor Tarns, cumulative distribution functions for exceedance were also plotted (Figure 2.2.2).

Table 2.2.1: Input parameters – lake Sites

Code	Units	ARR	NAGA	CHON	TINK	RLGH	LGR	SCOAT	BURN	LAG	MYN	BLU
Parameter	**Site No**	**1**	**4**	**5**	**6**	**7**	**8**	**10**	**11**	**15**	**16**	**21**
Q	m yr^{-1}	2.59	1.37	1.86	1.86	1.83	1.83	2.58	2.02	2.85	1.75	1.24
A	ha	900	109	1,590	133	103	1,432	87	626	157	125	50
A lake	ha	14.35	9.90	105.71	11.12	12.68	111.41	4.32	23.92	5.09	5.66	2.10
A mcf	ha	0.00	0.00	324.12	0.00	0.00	569.59	0.00	0.00	0.00	37.58	0.00
A ucf	ha	0.00	0.00	218.84	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A mbl	ha	0.00	0.00	3.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00
A g	ha	886	99	938	121	90	751	83	602	152	81	48
Nu	keq $ha^{-1}yr^{-1}$	0.00	0.00	0.05	0.00	0.00	0.09	0.00	0.00	0.00	0.07	0.00
Ni	keq $ha^{-1}yr^{-1}$	0.21	0.19	0.18	0.07	0.11	0.19	0.07	0.11	0.21	0.10	0.16
Nde	keq $ha^{-1}yr^{-1}$	0.07	0.06	0.12	0.25	0.06	0.14	0.07	0.07	0.07	0.07	0.09
BC*t	µeq L^{-1}	53.55	58.50	75.41	102.27	38.30	36.63	42.24	116.23	53.74	64.91	54.65
BC*t SD	µeq L^{-1}	15.87	8.63	18.32	21.42	8.63	18.15	12.21	19.80	12.68	13.86	17.32
SO4*t	µeq L^{-1}	12.39	45.20	43.91	34.35	39.26	60.74	39.77	5.41	35.11	49.14	56.31
SO_4* SD	µeq L^{-1}	4.09	6.09	9.98	10.68	10.71	18.85	4.82	9.64	9.45	9.68	14.77
NO_{3t}	µeq L^{-1}	2.44	16.94	11.89	2.79	7.16	18.22	19.80	5.92	7.93	10.30	27.75
NO_3SD	µeq L^{-1}	0.88	5.67	3.53	1.57	3.08	6.90	5.43	1.83	3.98	4.15	9.12
Correlations												
BC- SO_4		0.36	0.67	0.61	0.16	0.53	0.63	0.40	0.61	0.63	0.59	0.77
BC -NO_3		0.38	0.24	0.50	0.30	0.36	0.28	0.40	-	0.28	-	0.16
SO_4 NO_3		0.48	-0.48	0.29	-	0.16	0.63	0.24	0.39	0.63	0.22	0.46
N_{dep}	keq $ha^{-1}yr^{-1}$	0.37	1.06	1.43	1.44	1.42	1.16	1.61	1.28	1.03	0.76	-
S_{dep}	keq $ha^{-1}yr^{-1}$	0.35	0.66	0.91	0.92	0.85	0.67	1.00	0.82	0.85	0.48	-

Abbreviations: As in Section 2.2.2.1 above, and: mcf, managed coniferous forest; ucf, unmanaged coniferous forest; mbl, managed broadleaves; g, grassland (non-forest); SD, standard deviation; BC-SO_4, correlation between annual mean non-marine base cations and annual mean non-marine sulphate concentrations in observations; BC – NO_3, SO_4 – NO_3 similarly. The parameters *a*, *b*, S_N and S_S were the same for each site as explained above.

Table 2.2.2: Input parameters – stream sites

Code	Units	MHAR	ANCC	DARG	ETHR	LODG	NART	HAFR	GWY	BEAH	BENC	CONY
Parameter	**Site No.**	**2**	**3**	**9**	**12**	**13**	**14**	**17**	**18**	**19**	**20**	**22**
Q	m yr^{-1}	1.06	2.10	1.72	1.14	0.40	1.23	1.92	2.04	1.22	1.34	1.10
A	ha	979	817	216	1,295	301	446	373	389	302	216	1,312
A lake	ha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A mcf	ha	1.75	96.40	1.32	0.00	11.49	0.61	147.91	6.03	5.08	0.89	52.16
A ucf	ha	50.81	4.58	0.00	0.69	0.00	0.00	0.00	0.00	0.00	0.06	0.23
A mbl	ha	0.00	3.92	0.00	0.00	13.35	21.54	0.00	0.00	0.00	0.00	0.00
A g	ha	926	712	215	1,294	276	424	225	383	297	215	1,260
Nu	keq $ha^{-1}yr^{-1}$	0.00	0.03	0.01	0.00	0.03	0.02	0.08	0.00	0.00	0.00	0.01
Ni	keq $ha^{-1}yr^{-1}$	0.20	0.21	0.10	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21
Nde	keq $ha^{-1}yr^{-1}$	0.07	0.16	0.07	0.09	0.07	0.11	0.07	0.07	0.07	0.07	0.09
BC*t	μeq L^{-1}	83.80	66.62	71.32	292.03	141.05	66.27	74.63	63.72	139.15	97.75	220.18
BC*t SD	μeq L^{-1}	16.99	18.96	11.53	60.07	35.18	10.78	13.63	11.02	33.11	20.02	40.64
SO4*t	μeq L^{-1}	27.85	28.55	56.03	211.07	164.47	45.47	55.11	41.99	29.06	59.38	24.21
SO_4 SD	μeq L^{-1}	3.66	7.31	8.23	51.09	55.61	2.90	9.58	8.81	14.33	11.89	11.28
NO_3t	μeq L^{-1}	2.06	4.05	11.63	46.60	7.78	7.04	21.69	9.44	3.17	27.45	2.63
NO_3SD	μeq L^{-1}	0.33	1.52	5.44	7.87	2.81	1.04	4.55	3.93	1.13	8.21	1.82
Correlations												
BC- SO_4		0.78	0.51	0.42	0.83	0.65	0.71	-0.15	0.33	0.50	0.64	0.54
BC -NO_3		-	-	-	-	0.57	-	-0.25	-0.33	0.14	0.49	-
SO_4 NO_3		-	0.24	-	-	-	0.24	-0.34	-	0.22	0.21	0.41
N_{dep}	keq $ha^{-1}yr^{-1}$	0.42	0.57	1.24	2.21	1.14	1.14	1.25	1.25	-	-	-
S_{dep}	keq $ha^{-1}yr^{-1}$	0.25	0.51	0.78	1.54	0.50	0.54	0.78	0.78	-	-	-

Abbreviations: As in Section 2.2.2.1 above, and: mcf, managed coniferous forest; ucf, unmanaged coniferous forest; mbl, managed broadleaves; g, grassland (non-forest); SD, standard deviation; BC-SO_4, correlation between annual mean non-marine base cations and annual mean non-marine sulphate concentrations in observations; BC – NO_3, SO_4 – NO_3 similarly. The parameters *a*, *b*, S_N and S_S were the same for each site as explained above.

Table 2.2.3: Means and coefficients of variation of critical load values: lake sites

Code		ARR	NAGA	CHON	TINK	RLGH	LGR	SCOAT	BURN	LAG	MYN	BLU	Overall
Site No		**1**	**4**	**5**	**6**	**7**	**8**	**10**	**11**	**15**	**16**	**21**	**Mean**
Forest	**%**	0.0	0.0	20.6	0.0	0.0	39.8	0.0	0.0	0.0	30.1	0.0	
L_{crit}	keq $ha^{-1}yr^{-1}$	0.864	0.395	0.842	1.462	0.253	0.140	0.404	1.652	0.818	0.616	0.508	0.723
CV	%	42	20	29	25	45	145	56	18	33	28	27	43
$CL_{max}S$	keq $ha^{-1}yr^{-1}$	0.870	0.408	0.857	1.495	0.262	0.143	0.407	1.668	0.822	0.624	0.518	0.734
CV	%	42	20	29	25	45	145	56	18	33	28	27	43
$CL_{min}N$	keq $ha^{-1}yr^{-1}$	0.283	0.261	0.351	0.327	0.175	0.420	0.136	0.176	0.276	0.231	0.256	0.263
CV	%	22	23	19	20	21	18	20	21	23	17	21	20
$CL_{max}N$	keq $ha^{-1}yr^{-1}$	1.159	0.787	1.344	2.117	0.514	0.566	0.578	1.984	1.140	0.926	0.844	1.087
CV	%	33	16	22	22	31	63	43	17	26	22	20	29
Exceed	keq $ha^{-1}yr^{-1}$	-0.437	0.785	0.850	0.238	1.369	1.037	1.780	0.061	0.480	0.256	-	0.642
SD	keq $ha^{-1}yr^{-1}$	0.406	0.413	0.640	0.670	0.540	0.486	0.711	0.631	0.533	0.365	-	

Means and coefficients of variation (CV) are shown. As CV is not a meaningful measure for exceedance, the standard deviation is shown as an estimate of spread. Negative exceedances mean non-exceedance.

Table 2.2.4: Means and coefficients of variation of critical load values: stream sites

Code		MHAR	ANCC	DARG	ETHR	LODG	NART	HAFR	GWY	BEAH	BENC	CONY	Overall
Site No.		**2**	**3**	**9**	**12**	**13**	**14**	**17**	**18**	**19**	**20**	**22**	**Mean**
Forest	**%**	0.2	12.3	0.6	0.0	8.3	5.0	39.7	1.6	1.7	0.4	4.0	
L_{crit}	keq $ha^{-1}yr^{-1}$	0.661	0.924	0.668	0.954	0.178	0.473	0.759	0.737	1.412	0.731	2.301	0.891
CV L_{crit}	%	24	37	22	32	46	22	26	23	25	22	18	27
$CL_{max}S$	keq $ha^{-1}yr^{-1}$	0.661	0.924	0.668	0.954	0.178	0.473	0.759	0.737	1.412	0.731	2.301	0.891
CV	%	24	37	22	32	46	22	26	23	25	22	18	27
$CL_{min}N$	keq $ha^{-1}yr^{-1}$	0.275	0.400	0.171	0.308	0.312	0.347	0.369	0.288	0.284	0.287	0.316	0.305
CV	%	23	19	20	22	21	20	19	23	23	23	21	21
$CL_{max}N$	keq $ha^{-1}yr^{-1}$	0.922	1.309	0.839	1.262	0.488	0.818	1.127	1.025	1.697	1.018	2.617	1.193
CV	%	18	27	18	25	21	15	18	18	21	17	16	19
Exceed.	keq $ha^{-1}yr^{-1}$	0.284	-0.270	1.046	2.218	1.046	0.750	0.768	0.874	-	-	-	0.840
SD	keq $ha^{-1}yr^{-1}$	0.237	0.434	0.574	1.069	0.490	0.496	0.602	0.585	-	-	-	

Means and coefficients of variation (CV) are shown. As CV is not a meaningful measure for exceedance, the standard deviation is shown as an estimate of spread. Negative exceedances mean non-exceedance.

Table 2.2.5: Sensitivity analysis for lake sites: $CL_{max}S$

Code		ARR	NAGA	CHON	TINK	RLGH	LGR	SCOAT	BURN	LAG	MYN	BLU	Means
Site No		**1**	**4**	**5**	**6**	**7**	**8**	**10**	**11**	**15**	**16**	**21**	
Q	m yr^{-1}	1.1	1.8	1.3	2.7	0.7	0.1	0.2	3.8	1.2	1.9	2.6	1.6
A	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A lake	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mcf	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A ucf	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mbl	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nu (mcf)	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nu (mbl)	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ni	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
Nde	keq $ha^{-1}yr^{-1}$	0.1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
BC*t	µeq L^{-1}	81.3	65.7	69.3	90.3	81.6	83.6	83.3	79.4	79.1	83.0	90.2	80.6
SO4*t	µeq L^{-1}	7.8	28.2	15.4	0.1	9.7	15.9	8.1	13.7	17.5	14.3	5.3	12.4
NO_3t	µeq L^{-1}	9.6	2.7	13.3	6.0	7.4	0.9	7.9	1.1	1.4	0.0	0.9	4.7
S_N	m yr^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S_S	m yr^{-1}	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.1	0.0	0.1	0.0
a	µeq L^{-1}	0.0	1.3	0.3	0.1	0.5	0.0	0.4	1.2	0.5	0.5	0.7	0.5
b		0.0	0.3	0.2	0.4	0.0	0.0	0.1	0.7	0.2	0.2	0.3	0.2
N_{dep}	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S_{dep}	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0

The table shows the percentage contribution of each parameter to variance in $CL_{max}S$. The largest contributor to each site is highlighted in gold. Parameters contributing less than 0.05 per cent of the variance are highlighted in turquoise.

Table 2.2.6: Sensitivity analysis for stream sites: $CL_{max}S$

Code		MHAR	ANCC	DARG	ETHR	LODG	NART	HAFR	GWY	BEAH	BENC	CONY	Means
Site No.		**2**	**3**	**9**	**12**	**13**	**14**	**17**	**18**	**19**	**20**	**22**	
Q	m yr^{-1}	1.7	1.0	2.9	6.4	2.0	2.4	2.5	2.8	2.0	3.4	4.5	2.9
A	ha	0.0	0.0	0.0	0.2	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
A lake	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mcf	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A ucf	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
A mbl	ha	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nu (mcf)	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nu (mbl)	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ni	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
Nde	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
BC*t	µeq L^{-1}	63.2	83.6	89.1	32.6	0.5	67.4	94.3	90.6	88.2	74.0	80.6	69.5
SO4*t	µeq L^{-1}	34.2	14.1	3.8	-15.9	-95.0	28.2	-1.6	0.9	8.5	13.4	12.1	20.7
NO_3t	µeq L^{-1}	0.0	0.7	-1.9	-17.4	0.0	1.1	0.3	4.8	0.7	7.5	1.5	3.3
S_N	m yr^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
S_S	m yr^{-1}	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
a	µeq L^{-1}	0.2	0.4	0.0	3.5	1.0	0.6	0.9	0.8	0.3	1.1	0.1	0.8
b		0.4	0.0	0.0	23.9	1.1	0.2	0.2	0.0	0.2	0.4	1.0	2.5
N_{dep}	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S_{dep}	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0

The table shows the percentage contribution of each parameter to variance in $CL_{max}S$. The largest contributor to each site is highlighted in gold. Parameters contributing less than 0.05 per cent of the variance are highlighted in turquoise. Negative values mean $CL_{max}S$ declines as the parameter increases.

Table 2.2.7: Sensitivity analysis for lake sites: $CL_{max}N$

Code		ARR	NAGA	CHON	TINK	RLGH	LGR	SCOAT	BURN	LAG	MYN	BLU	Means
Site No		**1**	**4**	**5**	**6**	**7**	**8**	**10**	**11**	**15**	**16**	**21**	
Q	m yr^{-1}	1.1	1.3	0.6	2.3	0.4	0.1	0.1	3.2	0.6	1.4	1.8	1.2
A	ha	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A lake	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mcf	ha	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A ucf	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mbl	ha	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
Nu (mcf)	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.3	0.0	0.1
Nu (mbl)	keq $ha^{-1}yr^{-1}$	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ni	keq $ha^{-1}yr^{-1}$	1.7	13.2	1.9	0.3	3.5	3.2	0.7	0.5	3.0	1.6	6.7	3.3
Nde	keq $ha^{-1}yr^{-1}$	0.2	2.1	1.3	2.3	1.0	1.4	0.6	0.5	0.3	1.0	1.8	1.1
BC*t	µeq L^{-1}	80.8	49.8	66.2	82.8	75.3	78.4	82.0	76.9	77.1	78.9	80.5	75.3
SO4*t	µeq L^{-1}	6.6	21.6	15.2	0.0	9.6	15.2	7.9	13.4	16.9	14.1	4.8	11.4
NO_3t	µeq L^{-1}	9.0	1.6	12.6	5.8	6.8	0.6	7.8	1.1	1.1	0.0	0.8	4.3
S_N	m yr^{-1}	0.0	9.1	1.5	5.7	3.0	0.0	0.4	2.2	0.0	1.6	2.9	2.4
S_S	m yr^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
a	µeq L^{-1}	0.1	0.8	0.2	0.3	0.3	0.1	0.4	1.2	0.5	0.6	0.6	0.5
b		0.0	0.2	0.2	0.3	0.1	0.0	0.1	0.8	0.1	0.2	0.2	0.2
N_{dep}	keq $ha^{-1}yr^{-1}$	0.3	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.1
S_{dep}	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0

The table shows the percentage contribution of each parameter to variance in $CL_{max}N$. The largest contributor to each site is highlighted in gold. Parameters contributing less than 0.05 per cent of the variance are highlighted in turquoise. Negative values mean $CL_{max}N$ declines as the parameter increases.

Table 2.2.8: Sensitivity analysis for stream sites: $CL_{max}N$

Code		MHAR	ANCC	DARG	ETHR	LODG	NART	HAFR	GWY	BEAH	BENC	CONY	Means
Site No.		**2**	**3**	**9**	**12**	**13**	**14**	**17**	**18**	**19**	**20**	**22**	
Q	m yr^{-1}	1.5	0.9	2.7	5.9	0.6	1.3	2.1	2.3	2.1	2.8	4.3	2.4
A	ha	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A lake	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mcf	ha	0.0	0.0	0.1	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
A ucf	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
A mbl	ha	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nu (mcf)	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.2
Nu (mbl)	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.1	0.0	0.5	0.2	0.0	0.0	0.0	0.0	0.0	0.1
Ni	keq $ha^{-1}yr^{-1}$	7.9	1.9	3.9	11.9	52.5	19.3	9.7	12.6	2.6	10.5	1.9	12.2
Nde	keq $ha^{-1}yr^{-1}$	0.6	1.6	1.8	2.3	4.9	5.7	0.8	0.8	0.3	1.5	0.2	1.9
BC*t	µeq L^{-1}	57.7	79.7	84.3	28.7	0.7	50.9	82.2	78.3	85.5	66.0	79.2	63.0
SO4*t	µeq L^{-1}	31.3	14.4	3.5	-13.0	-39.4	21.3	-1.7	0.7	8.4	11.5	11.9	14.3
NO_3t	µeq L^{-1}	0.0	0.6	-1.6	-14.9	0.0	0.8	0.3	3.9	0.7	6.7	1.5	2.8
S_N	m yr^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S_S	m yr^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
a	µeq L^{-1}	0.1	0.2	1.2	3.1	0.3	0.3	1.2	0.8	0.2	0.7	0.1	0.7
b		0.2	0.2	0.7	20.0	0.9	0.2	0.2	0.4	0.2	0.3	0.9	2.2
N_{dep}	keq $ha^{-1}yr^{-1}$	0.5	0.5	0.5	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.1
S_{dep}	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

The table shows the percentage contribution of each parameter to variance in $CL_{max}N$. The largest contributor to each site is highlighted in gold. Parameters contributing less than 0.05 per cent of the variance are highlighted in turquoise. Negative values mean $CL_{max}N$ declines as the parameter increases.

Table 2.2.9: Sensitivity analysis for lake sites: $CL_{min}N$

Code	Units	ARR	NAGA	CHON	TINK	RLGH	LGR	SCOAT	BURN	LAG	MYN	BLU	Means
Site No		**1**	**4**	**5**	**6**	**7**	**8**	**10**	**11**	**15**	**16**	**21**	
Q	m yr^{-1}	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
A	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A lake	ha	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mcf	ha	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.7	0.0	0.1
A ucf	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mbl	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nu (mcf)	keq $ha^{-1}yr^{-1}$	0.0	0.0	3.3	0.0	0.0	10.3	0.0	0.0	0.0	0.0	21.5	3.2
Nu (mbl)	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ni	keq $ha^{-1}yr^{-1}$	90.7	89.9	67.3	6.9	79.6	57.2	49.9	72.9	91.6	55.1	78.4	67.2
Nde	keq $ha^{-1}yr^{-1}$	9.1	9.8	29.0	92.9	20.2	31.5	49.8	26.9	8.1	22.6	21.5	29.2
BC*t	µeq L^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.1	0.0
SO4*t	µeq L^{-1}	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
NO_3t	µeq L^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S_N	m yr^{-1}	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S_S	m yr^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
a	µeq L^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0
b		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N_{dep}	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
S_{dep}	keq $ha^{-1}yr^{-1}$	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

The table shows the percentage contribution of each parameter to variance in $CL_{min}N$. The largest contributor to each site is highlighted in gold. Parameters contributing less than 0.05 per cent of the variance are highlighted in turquoise.

Table 2.2.10: Sensitivity analysis for stream sites: $CL_{min}N$

Code	Units	MHAR	ANCC	DARG	ETHR	LODG	NART	HAFR	GWY	BEAH	BENC	CONY	Means
Site No.		**2**	**3**	**9**	**12**	**13**	**14**	**17**	**18**	**19**	**20**	**22**	
Q	m yr^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
A	ha	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A lake	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mcf	ha	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0
A ucf	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mbl	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nu (mcf)	keq $ha^{-1}yr^{-1}$	0.1	0.6	0.1	0.0	0.1	0.0	11.1	0.2	0.0	0.0	0.1	1.1
Nu (mbl)	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.5	0.8	0.0	0.0	0.0	0.0	0.0	0.1
Ni	keq $ha^{-1}yr^{-1}$	88.8	65.1	67.3	85.7	89.9	78.6	80.2	90.3	89.7	89.0	86.8	82.9
Nde	keq $ha^{-1}yr^{-1}$	10.9	33.8	32.4	14.1	9.3	20.3	8.3	9.3	10.1	10.5	13.0	15.6
BC*t	µeq L^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SO4*t	µeq L^{-1}	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
NO_3t	µeq L^{-1}	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
S_N	m yr^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S_S	m yr^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0
a	µeq L^{-1}	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0
b		0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N_{dep}	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
S_{dep}	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0

The table shows the percentage contribution of each parameter to variance in $CL_{min}N$. The largest contributor to each site is highlighted in gold. Parameters contributing less than 0.05 per cent of the variance are highlighted in turquoise.

Table 2.2.11: Sensitivity analysis for lake sites: exceedance

Code	Units	ARR	NAGA	CHON	TINK	RLGH	LGR	SCOAT	BURN	LAG	MYN	BLU	Means
Site No		**1**	**4**	**5**	**6**	**7**	**8**	**10**	**11**	**15**	**16**	**21**	
Q	m yr^{-1}	-0.8	-0.2	-0.3	-1.2	-0.1	0.0	0.0	-0.5	-0.3	-0.3	-	-0.4
A	ha	0.0	0.0	-0.1	-0.1	-0.3	0.0	-0.3	-0.1	0.0	-0.6	-	-0.2
A lake	ha	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	-	0.0
A mcf	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-	0.0
A ucf	ha	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-	0.0
A mbl	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-	0.0
												-	
Nu (mcf)	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	-0.2	0.0	0.0	0.0	-0.1	-	0.0
Nu (mbl)	keq $ha^{-1}yr^{-1}$	0.0	-0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-0.1	-	0.0
Ni	keq $ha^{-1}yr^{-1}$	-0.6	-1.4	-0.5	0.0	-0.2	-1.0	0.0	-0.2	-0.9	-0.5	-	-0.5
Nde	keq $ha^{-1}yr^{-1}$	-0.2	-0.3	-0.3	-0.6	0.0	-0.3	0.0	-0.2	-0.1	-0.3	-	-0.2
												-	
BC*t	µeq L^{-1}	-70.1	-2.6	-13.0	-27.6	-4.3	-15.1	-10.0	-21.2	-22.2	-20.4	-	-20.7
SO4*t	µeq L^{-1}	-6.9	-0.9	-2.6	0.0	-0.5	-3.6	-0.8	-3.1	-5.0	-3.2	-	-2.7
NO_3t	µeq L^{-1}	-8.1	0.0	-2.8	-1.7	-0.5	-0.1	-0.7	0.0	-0.5	0.0	-	-1.4
												-	
S_N	m yr^{-1}	0.0	-1.7	-0.5	-0.6	-1.5	-0.3	-0.4	-0.2	0.0	-0.5	-	-0.6
S_S	m yr^{-1}	0.0	0.0	0.0	-0.2	0.0	0.0	0.0	0.0	0.0	0.0	-	0.0
a	µeq L^{-1}	-0.1	-0.1	-0.2	0.0	-0.2	0.0	0.0	-0.1	-0.3	-0.3	-	-0.1
b		0.0	-0.1	0.0	-0.2	0.0	0.0	0.0	-0.2	-0.1	0.0	-	-0.1
												-	
N_{dep}	keq $ha^{-1}yr^{-1}$	4.7	56.7	53.0	43.7	58.0	53.1	61.1	49.7	47.9	49.9	-	47.8
S_{dep}	keq $ha^{-1}yr^{-1}$	8.5	36.9	26.7	24.0	34.3	25.9	26.5	24.4	22.7	23.6	-	25.4

The table shows the percentage contribution of each parameter to variance in exceedance. The largest contributor to each site is highlighted in gold. Parameters contributing less than 0.05 per cent of the variance are highlighted in turquoise. Negative values mean exceedance declines as the parameter increases.

Table 2.2.12: Sensitivity analysis for stream sites: exceedance

Code	Units	MHAR	ANCC	DARG	ETHR	LODG	NART	HAFR	GWY	BEAH	BENC	CONY	Means
Site No.		2	3	9	12	13	14	17	18	19	20	22	
Q	m yr^{-1}	-1.1	-0.5	-0.2	-0.1	-0.1	-0.3	-0.4	-0.2	-	-	-	-0.4
A	ha	-0.1	-0.1	0.0	0.0	-0.1	-0.1	-0.1	-0.1	-	-	-	-0.1
A lake	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-	-	-	0.0
A mcf	ha	0.0	0.0	0.0	0.0	0.0	0.0	-0.1	0.0	-	-	-	0.0
A ucf	ha	0.0	-0.1	0.0	0.0	0.0	0.0	0.0	0.0	-	-	-	0.0
A mbl	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-	-	-	0.0
										-	-	-	
Nu (mcf)	keq $ha^{-1}yr^{-1}$	0.0	0.0	-0.1	0.0	0.0	0.0	-0.1	-0.1	-	-	-	0.0
Nu (mbl)	keq $ha^{-1}yr^{-1}$	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-	-	-	0.0
Ni	keq $ha^{-1}yr^{-1}$	-2.4	-1.3	-0.2	-0.2	-1.7	-2.1	-1.0	-0.8	-	-	-	-1.2
Nde	keq $ha^{-1}yr^{-1}$	-0.4	-0.4	-0.3	0.0	-0.1	-0.6	0.0	0.0	-	-	-	-0.2
										-	-	-	
BC*t	μeq L^{-1}	-37.3	-56.9	-5.6	-1.0	-0.1	-4.0	-9.0	-7.9	-	-	-	-15.2
SO4*t	μeq L^{-1}	-20.0	-9.6	-0.1	-0.3	-0.9	-2.0	-0.4	0.0	-	-	-	-4.2
NO_3t	μeq L^{-1}	0.0	-0.5	-0.1	-0.7	0.0	-0.1	-0.2	-0.6	-	-	-	-0.3
										-	-	-	
S_N	m yr^{-1}	0.0	0.0	-0.1	0.0	0.0	-0.1	0.0	0.0	-	-	-	0.0
S_S	m yr^{-1}	0.0	-0.1	-0.1	0.0	0.0	-0.2	0.0	0.0	-	-	-	-0.1
a	μeq L^{-1}	0.0	-0.3	0.0	0.0	0.0	0.0	0.0	-0.2	-	-	-	-0.1
b		-0.3	0.0	-0.1	-0.9	-0.2	0.0	0.0	0.0	-	-	-	-0.2
										-	-	-	
N_{dep}	keq $ha^{-1}yr^{-1}$	24.5	13.9	66.2	65.7	81.6	73.0	63.8	64.8	-	-	-	56.7
S_{dep}	keq $ha^{-1}yr^{-1}$	13.8	16.4	26.9	31.0	15.2	17.6	24.8	25.4	-	-	-	21.4

The table shows the percentage contribution of each parameter to variance in exceedance. The largest contributor to each site is highlighted in gold. Parameters contributing less than 0.05 per cent of the variance are highlighted in turquoise. Negative values mean exceedance declines as the parameter increases.

Table 2.2.13: Percentile exceedances: lake sites

Code	Units	ARR	NAGA	CHON	TINK	RLGH	LGR	SCOAT	BURN	LAG	MYN	BLU
Percentile	**Site No**	**1**	**4**	**5**	**6**	**7**	**8**	**10**	**11**	**15**	**16**	**21**
10	keq $ha^{-1}yr^{-1}$	-0.986	0.282	0.055	-0.606	0.726	0.427	0.887	-0.738	-0.186	-0.204	-
50	keq $ha^{-1}yr^{-1}$	-0.444	0.747	0.815	0.203	1.341	1.002	1.725	0.019	0.458	0.238	-
90	keq $ha^{-1}yr^{-1}$	0.102	1.349	1.700	1.130	2.123	1.685	2.742	0.915	1.197	0.743	-
Ex 0	percentile	84	5	8	38	2	4	2	49	19	24	-

Exceedance is shown as a positive number. Deposition was estimated for the years 1998 – 2000. Percentiles are thus the percent confidence that exceedance is the stated value or less. Non-exceeded values are highlighted in green. “Ex 0” is the percent confidence that the critical load for acid deposition at the site is *not* exceeded.

Table 2.2.14: Percentile exceedances: stream sites

	Code	MHAR	ANCC	DARG	ETHR	LODG	NART	HAFR	GWY	BEAH	BENC	CONY
Percentile	**Site No.**	**2**	**3**	**9**	**12**	**13**	**14**	**17**	**18**	**19**	**20**	**22**
10	keq $ha^{-1}yr^{-1}$	-0.582	-0.835	0.349	0.875	0.454	0.145	0.028	0.144	-	-	-
50	keq $ha^{-1}yr^{-1}$	-0.295	-0.281	0.986	2.141	0.989	0.699	0.713	0.833	-	-	-
90	keq $ha^{-1}yr^{-1}$	0.032	0.291	1.847	3.670	1.745	1.448	1.569	1.674	-	-	-
Ex 0	percentile	88	77	5	3	3	8	9	4	-	-	-

Exceedance is shown as a positive number. Deposition was estimated for the years 1998 – 2000. Percentiles are thus the percent confidence that exceedance is the stated value or less. Non-exceeded values are highlighted in green. “Ex 0” is the percent confidence that the critical load for acid deposition at the site is *not* exceeded.

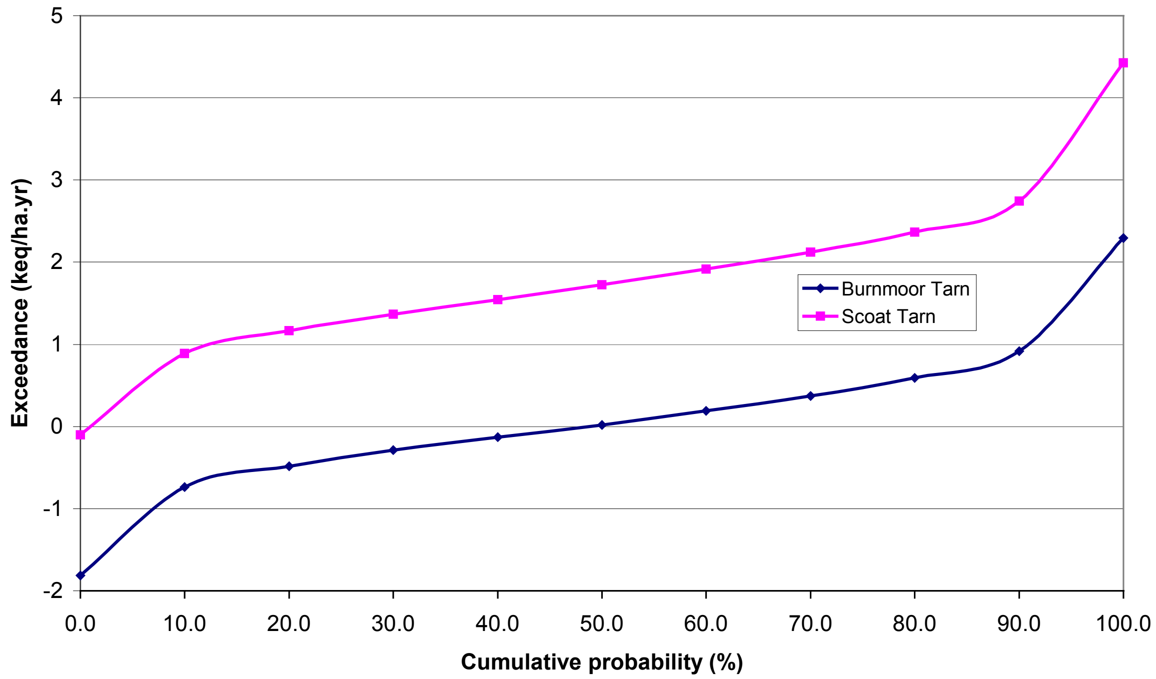


Figure 2.2.2: Cumulative Distribution Functions for Exceedance, Scoat and Burnmoor Tarns

Figure 2.2.2 shows cumulative distribution functions for exceedance for Scoat and Burnmoor Tarns only. Scoat Tarn can be said to be exceeded with a high degree of confidence (98 per cent) whereas the corresponding value for Burnmoor is only around 51 per cent (inverse of Table 2.2.13). The function shows that the tails of the distributions for both tarns are quite extensive, where exceedance changes rapidly with percentile below the 10th percentile and above the 90th percentile. This could have implications for the use of exceedances in probabilistic risk assessments.

2.2.3 Effect of uncertainty in the critical limit

2.2.3.1 Methods

The analysis above did not include any uncertainty in the value of the critical limit, which was set to an ANC of 20 μeq L^{-1}. The critical limit is, however, also subject to uncertainty; for instance, the UK has recently changed the value used from an ANC of 0 μeq L^{-1} to 20 μeq L^{-1}. Morgan and Henrion (1990), in a standard textbook on decision analysis, suggest that "decision variables" such as ANC_{lim} are best treated parametrically rather than probabilistically. In other words, although there is clearly uncertainty about the appropriate value for ANC_{lim}, the most profitable approach is to explore the effect of different choices of value on the outcome, rather than treat ANC_{lim} as an empirically variable quantity which contributes uncertainty on the same basis as other such quantities. It was thus decided to re-run some sites with a limiting ANC of 0 μeq L^{-1} for comparative purposes.

The results below use two sites from the Lake District as examples. Scoat Tarn is a very sensitive site; Burnmoor Tarn is less so. The FAB model was run in a Monte Carlo framework for these sites to provide a prediction for future steady state ANC and exceedance. Three scenarios were used for each lake:

- parameters as used in Section 2.2.2, but with the ANC limit set to 0 µeq L^{-1} rather than 20 µeq L^{-1};
- as above, but with deposition at the critical load (ANC 0 µeq L^{-1});
- as above, but with deposition at the critical load (ANC 20 µeq L^{-1}).

Nitrogen and sulphur deposition were altered to meet the critical loads in the same proportion as in current deposition. The values used are shown in Table 2.2.15.

Table 2.2.15: S and N deposition scenarios used

	Burnmoor Tarn			Scoat Tarn		
[1]Dep	Present	CL 0	CL 20	Present	CL 0	CL 20
N dep	1.2829	1.4641	1.2028	1.6093	0.6771	0.3473
*S dep	0.8208	0.9367	0.7696	0.9996	0.4332	0.2157

[1]Deposition in keq $ha^{-1}yr^{-1}$. Present = present (1998-2000) deposition; CL 0 = deposition at the critical load (ANC 0 µeq L^{-1}); CL 20 = deposition at the critical load (ANC 20 µeq L^{-1}).

2.2.3.2 Results

The effect of changing the critical limit on the mean and coefficient of variation of the critical load parameters for Burnmoor and Scoat Tarns is shown in Table 2.2.16. Increasing the critical limit reduces the critical loads substantially (except for $CL_{min}N$, which is not affected by the critical limit). With 1998-2000 deposition, Burnmoor Tarn moves into exceedance from non-exceedance. Another effect is to increase the coefficient of variation. Since everything in the runs except the ANC limit is the same, the spread of Monte Carlo outputs is the same and hence a lower mean implies a higher CV.

A sensitivity analysis for the revised critical limits revealed the same pattern as for ANC 20 (Table 2.2.3) and hence is not shown. For exceedance, the sensitivity analysis is shown in Table 2.2.17 below.

Table 2.2.16: Effect of changing the critical limit

Site		Burnmoor		Scoat	
[1]Critical limit		CL 0	CL 20	CL 0	CL 20
L_{crit}	keq $ha^{-1}yr^{-1}$	2.052	1.652	0.910	0.404
CV	%	15	18	25	56
$CL_{max}S$	keq $ha^{-1}yr^{-1}$	2.072	1.668	0.918	0.407
CV	%	15	18	25	56
$CL_{min}N$	keq $ha^{-1}yr^{-1}$	0.176	0.176	0.136	0.136
CV	%	21	21	20	20
$CL_{max}N$	keq $ha^{-1}yr^{-1}$	2.422	1.984	1.131	0.578
CV	%	14	17	22	43
Exceedance	keq $ha^{-1}yr^{-}$	-0.367	0.061	1.273	1.780

	1				
SD	keq ha^{-1}yr^{-1}	0.619	0.631	0.719	0.711

[1]CL 0 means a critical limit of 0 μeq L^{-1} was used; CL 20 means 20 μeq L^{-1}. Exceedance was calculated using 1998-2001 deposition.

2.2.4 Uncertainty in future fish status

2.2.4.1 Introduction

The choice of ANC as an appropriate critical chemical limit is largely based on work in Norwegian lakes which shows a correlation between ANC and fish population status. The most comprehensive study of this kind is by Lien *et al.* (1996), which gives cumulative distribution graphs showing the relationship between ANC and three fish status variables (healthy, reduced and extinct) in a large population of Norwegian lakes. These are shown for a variety of fish species. These relationships make it possible to go beyond critical loads and investigate the probabilities of fish population recovery given different deposition scenarios. The relationships between fish populations and ANC in Lien *et al.* (1996) can be interpreted as probabilities that a lake will be in one or other of three population modes (healthy, reduced and extinct) at a given ANC. Since the FAB model calculates the future value of ANC for a given value of S and N deposition, the probability that lakes will be in one or more of these states can be calculated. Though this is an extension of the critical loads approach as such, it does provide an unusual opportunity to set uncertainty bounds on real environmental consequences rather than chemical abstractions. Probability distributions for fish health variables were calculated as described in Appendix A1.5.3. These were then applied to the distribution of ANC predictions as calculated by FAB.

2.2.4.2 Results: ANC predictions

FAB calculates the distribution of future ANC, and can also be used to generate an estimate of what ANC was in pre-industrial times (ANC_0). These are shown for Scoat and Burnmoor Tarns in Figures 2.2.3 to 2.2.6 below.

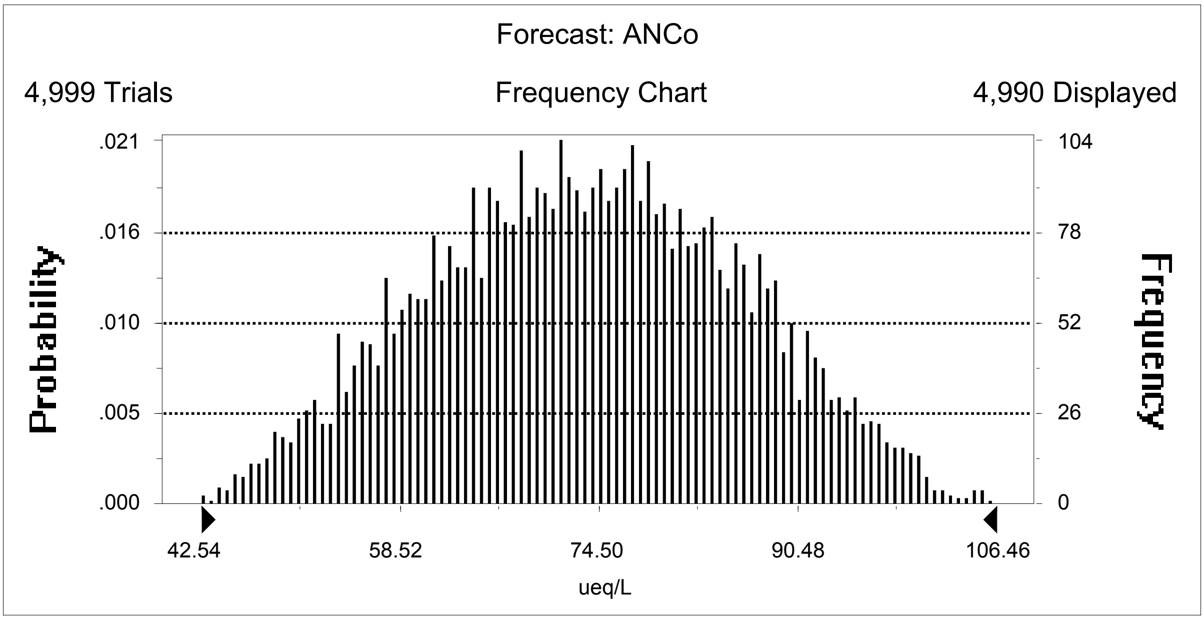


Figure 2.2.3: Modelled probability distribution for ANC_0 in Burnmoor Tarn in pre-industrial times

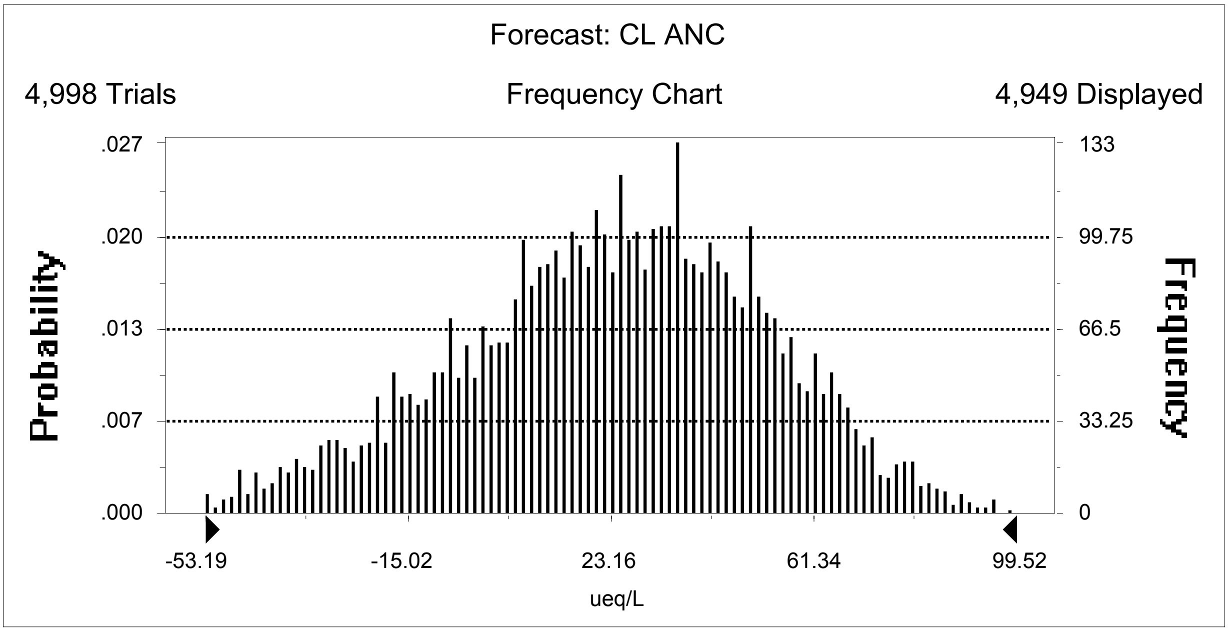


Figure 2.2.4: Modelled future probability distribution for ANC in Burnmoor Tarn if deposition was held at the critical load

For Burnmoor Tarn, the pre-industrial ANC was (according to the model) definitely positive, with a mean value of 74 μeq L^{-1} and a range of 40 to 108 μeq L^{-1} (Figure 2.2.3). If deposition were to be held at the critical load, then the model predicts a mean ANC of 23 μeq L^{-1} (the deterministic value is of course 20 μeq L^{-1}), and a much larger range of –113 to +109 μeq L^{-1} (the most extreme values are not shown on the diagram). Meeting the critical load does not imply a return to pristine conditions, and scientific uncertainty is such that the ultimate outcome could still encompass environmentally disastrous water qualities.

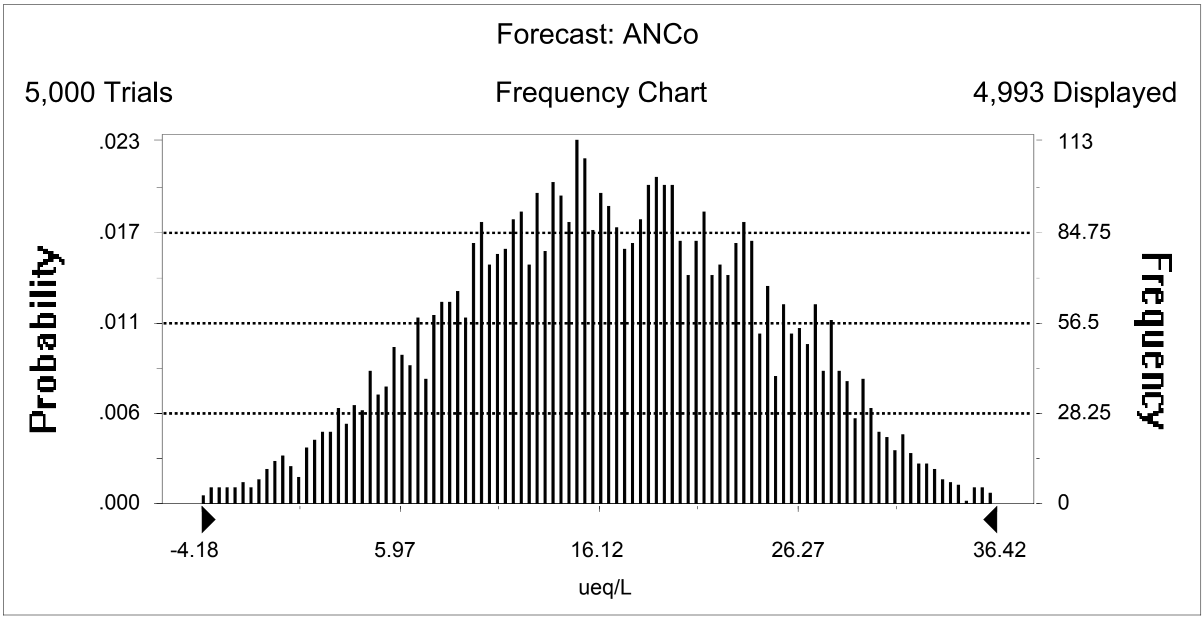


Figure 2.2.5: Modelled probability distribution for ANC_0 in Scoat Tarn in pre-industrial times

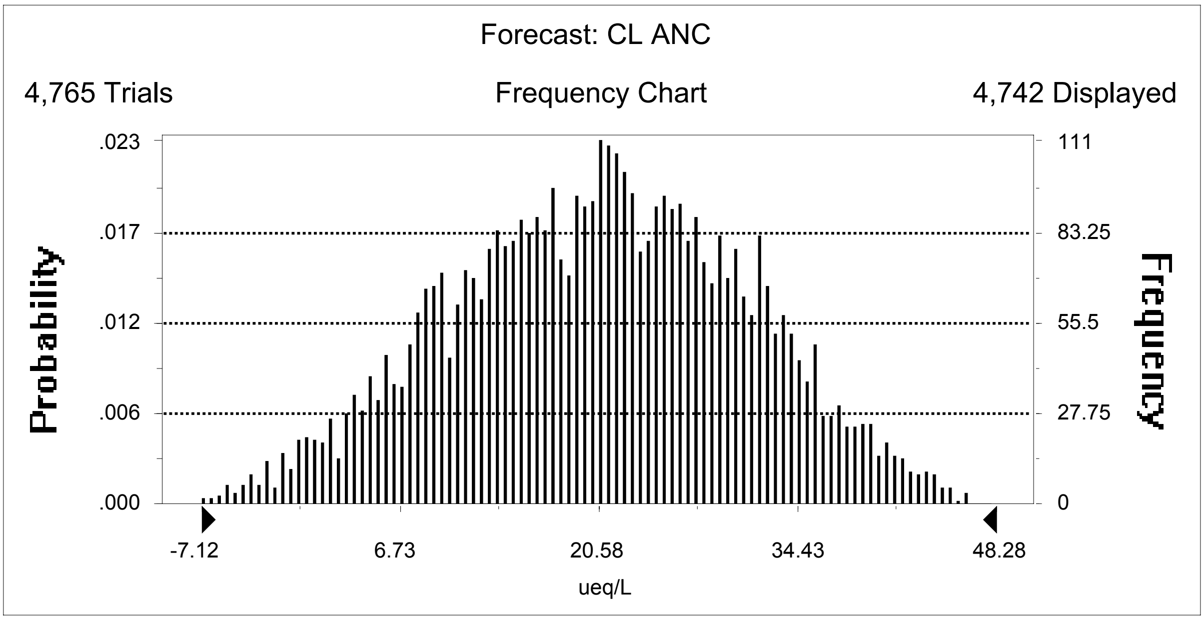


Figure 2.2.6: Modelled future probability distribution for ANC in Scoat Tarn if deposition was held at the critical load

For Scoat Tarn, the pre-industrial ANC is much lower than Burnmoor, with a mean of 17 μeq L^{-1} and a range of –5 to +38 μeq L^{-1}. The mean pre-industrial ANC is lower than the ANC criterion of 20 μeq L^{-1}, suggesting perhaps that this is too stringent. Deposition at this critical load leads to a higher mean (20.6 μeq L^{-1}) and a slightly wider range than the pre-industrial ANC. Nevertheless, about 18 per cent of predicted critical load values are below zero.

2.2.4.3 Results: uncertainty in fish responses

These ANC distributions can be used in conjunction with the fish response functions in Appendix A1.5.3 to estimate the probability that each lake would, in the long term, contain reduced or extinct fish populations. These can be plotted as cumulative distribution functions which have two probabilistic axes: the probability of population reduction or extinction on the y-axis, and the cumulative probability that this will occur on the x-axis. As well as predictions of fish status given the current (1998-2000) deposition, it is of interest to see how these functions change if deposition is reduced to the critical load. In Figures 2.2.7 to 2.2.10 below, the probability of fish damage with current deposition and with deposition at critical loads based on both 0 µeq L^{-1} ANC and 20 µeq L^{-1} ANC is plotted. Deposition scenarios were as in Table 2.2.15 above.

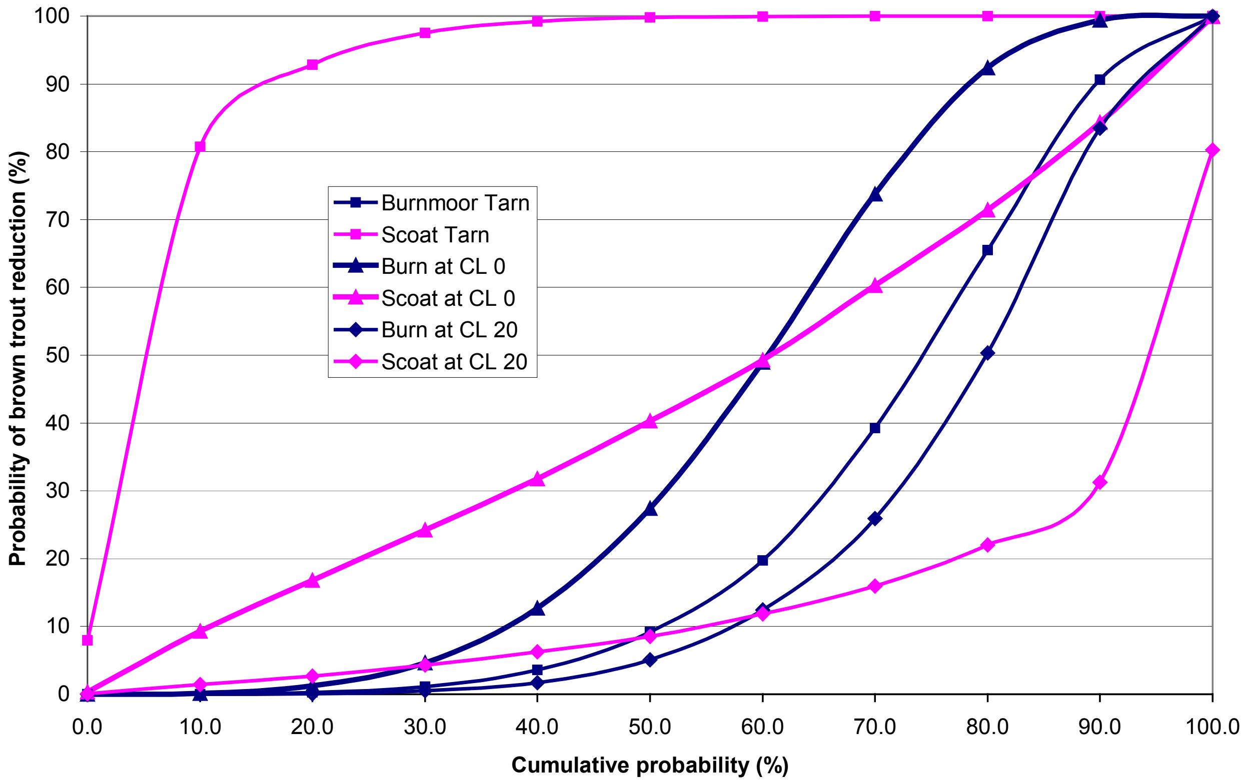


Figure 2.2.7: Probability of a reduction in brown trout populations in Scoat Tarn and Burnmoor Tarn with various deposition reduction scenarios

Figures 2.2.7 to 2.2.10 can be seen as summarising the effects of parameter uncertainty on predictions of brown trout and 'all fish' population status in the future. Figure 2.2.7 shows that although the same critical load criteria have been applied to both lakes, the responses are quite different. With present deposition, there is a high probability of a reduced brown trout population in Scoat Tarn (more than 50% of Monte Carlo runs show a 100% probability of population reduction). For Burnmoor Tarn, less than 1% of Monte Carlo runs indicate a 100% probability of a population reduction, and 50% of the simulations have a probability of 9% population reduction or less. If a precautionary criterion were adopted, that we should be at least 90% certain that the probability of brown trout population reduction was 50% or less, then neither lake meets the criterion with present deposition. The situation changes if deposition is changed to meet the old UK critical load criterion of 0 µeq L^{-1} ANC. Note that as this critical load was not exceeded at Burnmoor using the default parameters, this involves an *increase* of deposition at Burnmoor and a reduction at Scoat.

The responses of the lakes coincide where there is a 50% probability of brown trout population reduction, as predicted from the dose-response functions in Appendix A. Otherwise, Burnmoor is much more sensitive to the possible range of input parameters than Scoat. For instance, for Burnmoor about 22% of Monte Carlo runs have a population reduction probability of greater than 90%, whereas for Scoat only 5% of runs do so. At the other end of the distribution function, about 38% of Monte Carlo runs at Burnmoor have a population reduction probability of 10% or less, the corresponding figure for Scoat being 11%. Almost certainly, this is because of the larger deposition needed to exceed the critical load at Burnmoor, and hence the greater influence of deposition uncertainty.

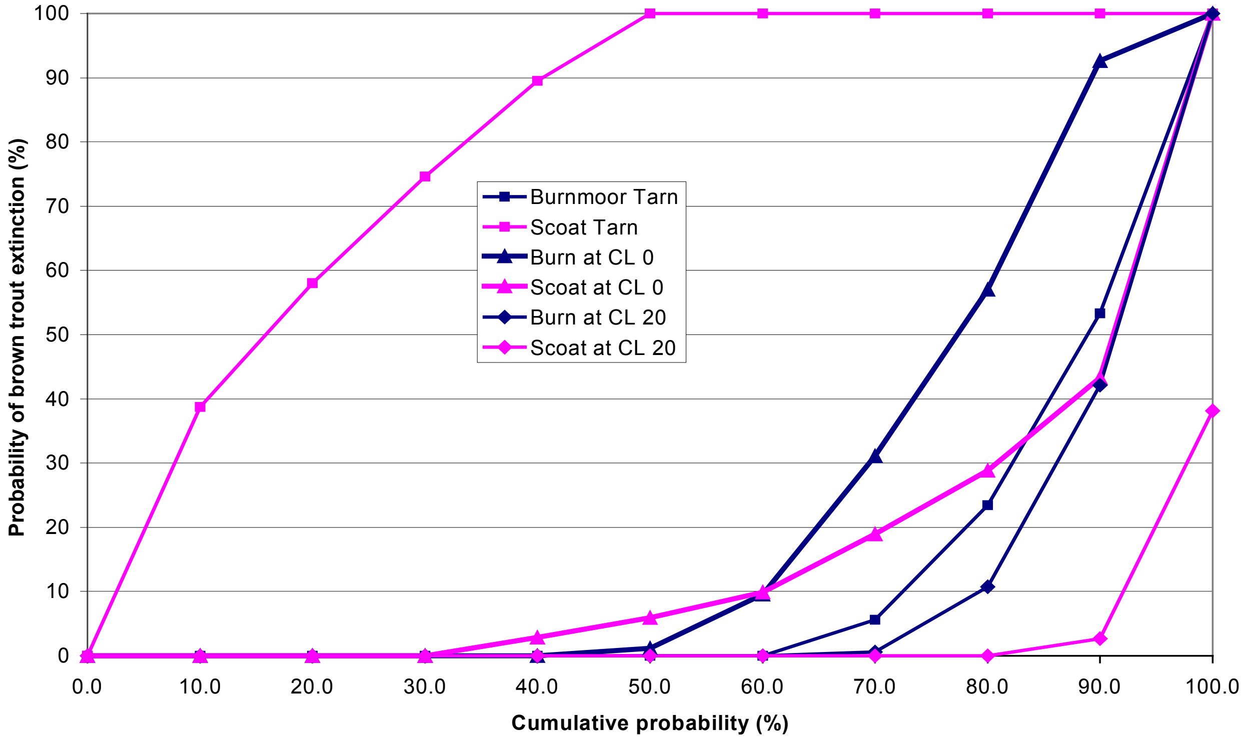


Figure 2.2.8: Probability of extinction of brown trout populations in Scoat Tarn and Burnmoor Tarn with various deposition reduction scenarios

This is borne out by the sensitivity analyses which show that deposition contributes 78% of the variation in exceedance at Burnmoo,r but 55% at Scoat with this scenario (see Table 2.2.17 below). The pattern is similar for the more stringent criterion of ANC 20 μeq L^{-1}. The responses of the lakes now coincide at about 12% probability of brown trout population reduction, but otherwise Burnmoor is again more sensitive than Scoat to the range of input parameters. The precautionary criterion that we should be at least 90% certain that the probability of brown trout population reduction is 50% or less is now met at Scoat (where we can be 95% certain) but not at Burnmoor (80% certain) with both sites at the critical load. This is in spite of (or possibly because of) the fact that Burnmoor is the less sensitive site using default parameters, and illustrates that the use of probabilistic criteria may change the relative sensitivity of sites.

For brown trout extinction (Figure 2.2.8), the probabilities are generally a little lower, but the patterns are similar to population reduction. With present deposition, 50% of Monte Carlo runs showed an extinction probability of 100% at Scoat, but no runs gave such a high probability at Burnmoor. Reducing or increasing deposition to meet ANC 0 μeq L^{-1}, both sites show the same extinction probability at about 9%, but as with population reduction, Burnmoor is more sensitive to the range of possible values. At ANC 20 μeq L^{-1}, the default parameters should generate no extinction (Figure 2.2.2), but there is a finite probability of population extinction which is greater at Burnmoor (Figure 2.2.8). At Scoat we can be 100% certain that the probability of extinction is 50% or less, and about 92% certain at Burnmoor.

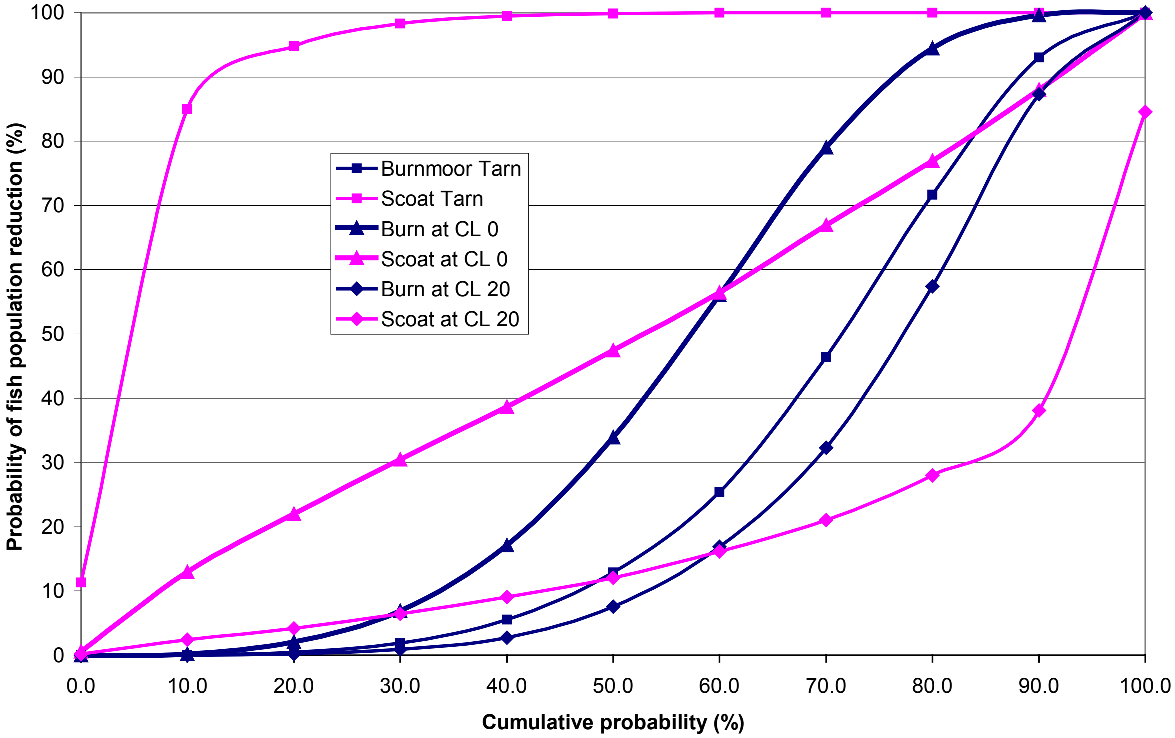


Figure 2.2.9: Probability of a reduction in fish populations in Scoat Tarn and Burnmoor Tarn with various deposition reduction scenarios

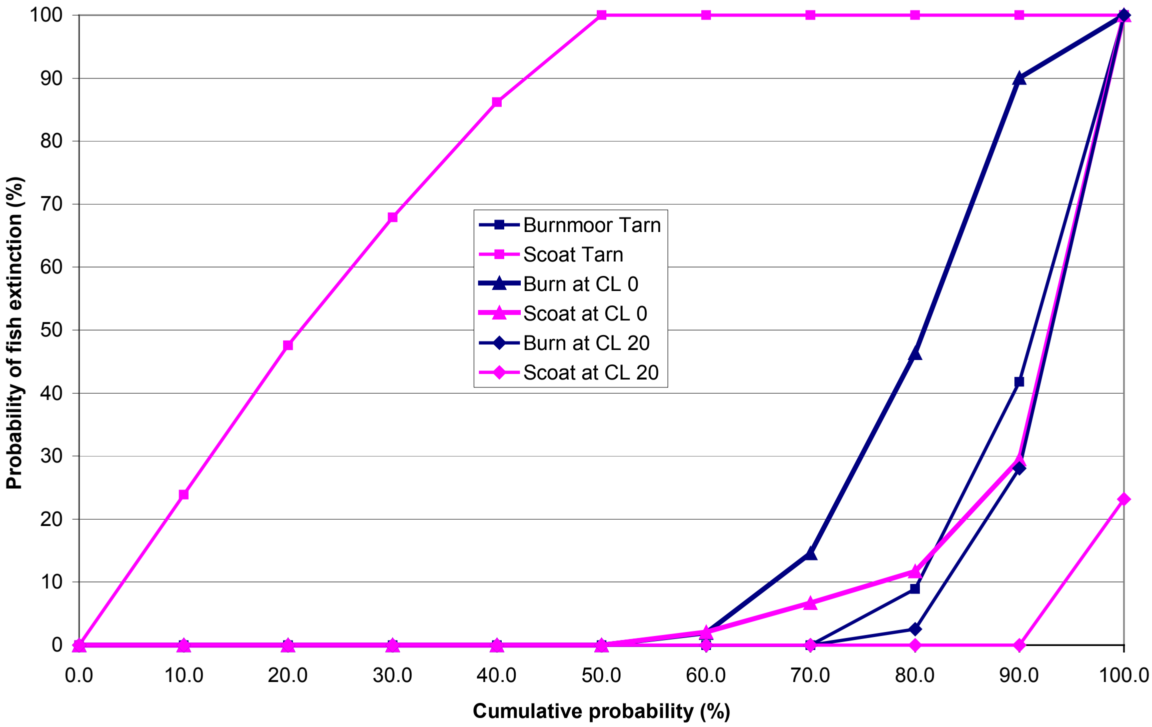


Figure 2.2.10: Probability of extinction of fish populations in Scoat Tarn and Burnmoor Tarn with various deposition reduction scenarios

For fish populations in general, the pattern of observations is very similar to that of brown trout (Figures 2.2.9 and 2.2.10). Fish population reduction is slightly more likely than brown trout population reduction, and fish extinction is slightly less likely than brown trout extinction. This simply reflects the functions in Lien *et al.* (1996) encapsulated in Appendix Figures A2 to A4. The comments about relative sensitivity apply to brown trout and 'all fish' equally.

2.2.4.4 Sensitivity analysis

Sensitivity analyses for exceedance and predicted steady state ANC at Scoat and Burnmoor Tarns are shown in Table 2.2.17 below. The analysis was performed for Scoat Tarn and Burnmoor Tarn with three levels of deposition as described above (Table 2.2.15).

The pattern of uncertainty is very similar to that of the critical load parameters (see Tables 2.2.5 to 2.2.12). N deposition has the major influence on exceedance, followed by non-marine S deposition, reflecting their different magnitudes (Table 2.2.17). As deposition reduces, deposition uncertainty becomes less important and catchment parameters, particularly present non-marine base cation concentration (BC*t), become more dominant. Other parameters play minor roles and some none at all. The sensitivity pattern for exceedance and predicted ANC is essentially identical. As the trout population functions are calculated from ANC it might be expected that they would also be very similar, and this proved to be the case (data not shown), though extinction for some reason was more sensitive to N deposition and less sensitive to base cation concentration than population reduction. The sensitivity analysis for all fish was essentially identical to that for trout (data not shown).

Table 2.2.17: Sensitivity analysis for exceedance and ANC, Scoat and Burnmoor Tarns

[1]Site		Ex: Burnmoor Tarn			Ex: Scoat Tarn			ANC: Burnmoor Tarn			ANC: Scoat Tarn		
[2]Deposition		**Pres**	**CL 0**	**CL 20**	**Pres**	**CL 0**	**CL 20**	**Pres**	**CL 0**	**CL 20**	**Pres**	**CL 0**	**CL 20**
Q	m yr^{-1}	1.4	1.0	0.9	0.1	0.8	0.4	0.8	0.8	0.7	1.0	0.7	0.4
A	ha	0.3	0.2	0.1	0.0	0.0	0.1	0.3	0.2	0.1	0.0	0.0	0.1
A lake	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mcf	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A ucf	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A mbl	ha	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nu (mcf)	keq ha^{-1}yr^{-1}	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nu (mbl)	keq ha^{-1}yr^{-1}	0.0	0.0	0.0	0.0	0.1	0.3	0.0	0.0	0.0	0.0	0.1	0.0
Ni	keq ha^{-1}yr^{-1}	0.2	0.5	0.3	0.1	0.2	0.0	0.2	0.5	0.3	0.1	0.2	0.2
Nde	keq ha^{-1}yr^{-1}	0.3	0.2	0.1	0.0	0.1	0.1	0.3	0.2	0.1	0.0	0.1	0.1
BC*t	µeq L^{-1}	19.2	15.9	23.5	9.8	36.7	64.3	19.4	15.9	23.5	9.8	36.8	64.2
SO4*t	µeq L^{-1}	3.7	3.0	4.5	0.6	3.4	6.8	3.7	3.0	4.5	0.6	3.4	6.8
NO_3t	µeq L^{-1}	0.2	0.3	0.3	0.8	3.2	5.9	0.2	0.3	0.3	0.8	3.2	5.9
S_N	m yr^{-1}	0.3	0.2	0.2	0.4	0.1	0.0	0.3	0.2	0.2	0.4	0.1	0.0
S_S	m yr^{-1}	0.1	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.1
a	µeq L^{-1}	0.4	0.1	0.2	0.0	0.0	0.0	0.4	0.1	0.2	0.0	0.0	0.0
b		0.2	0.0	0.1	0.0	0.0	0.0	0.2	0.0	0.1	0.0	0.0	0.0
N_{dep}	keq ha^{-1}yr^{-1}	50.3	53.1	47.8	59.7	38.2	14.3	50.6	53.5	47.8	59.2	38.2	14.4
S_{dep}	keq ha^{-1}yr^{-1}	23.4	25.4	22.1	28.3	17.1	7.7	23.5	25.5	22.1	28.0	17.1	7.7

2.2.5 Conclusions (freshwater ecosystems)

Tables 3.1 to 3.13 contain a wealth of interesting information. Some of the main points are:

- In spite of the complexity of the FAB model, coefficients of variation for the various critical load values are remarkably low.
- CVs for streams are generally lower than for lakes, which appears to be related to the lower variability of present non-marine base cation concentrations (CV 26% lakes, 20% streams). This in turn may be due to the higher sampling frequency for streams (12 times a year as opposed to four for lakes), as streams would be expected to be more variable than lakes.
- The CVs for all critical load parameters are lower than those in the trial sites for the SSMB model for soils.
- Only Loch Grannoch has a high CV for $CL_{max}S$ and $CL_{max}N$. The values of $CL_{max}S$ and $CL_{max}N$ are very low in absolute terms. The pre-industrial ANC was only 7.5 µeq L^{-1} according to the default parameters, which suggests that ANC = 20 µeq L^{-1} is not an appropriate critical load criterion for this site.
- It is noticeable that though all the relevant input parameters have spreads of ± 50%, the CVs for $CL_{min}N$ are only around 20%.
- The uncertainty analysis shows the remarkable result that for all sites except Old Lodge, uncertainty in the present non-marine base cation concentration is the major source of uncertainty in L_{crit}, $CL_{max}S$ and $CL_{max}N$, and for all except Old Lodge and the River Etherow, it is overwhelmingly the dominant source.
- The next most important sources are present non-marine sulphate and present nitrate concentrations. However, they appear to be important only to the extent that they are correlated with non-marine base cation concentrations, and become unimportant when these correlations are low or absent.
- Otherwise, the only parameters to have any consistent influence on $CL_{max}S$ are run-off and the parameters *a* and *b* in the equation linking pre-industrial sulphate with present non-marine base cation concentrations. The parameter *b* is particularly important at the River Etherow.
- Old Lodge is exceptional in being the only lowland site, otherwise it is difficult to see why it is so different.
- For $CL_{max}N$, the nitrogen immobilisation and denitrification parameters also have a small influence, and for lakes, the in-lake N reduction parameter S_N.
- $CL_{min}N$ is influenced only by the three N sink parameters, and of these, immobilisation is dominant except at Loch Tinker where it is denitrification. Uptake is never the dominant parameter. This result reflects the different magnitudes of the three sinks.
- The dominant influence on uncertainty of exceedance, however, is uncertainty in nitrogen deposition, except at three sites where deposition is very low. Sulphur deposition is normally second, followed by those factors which influence $CL_{max}N$. The predominance of nitrogen deposition reflects the fact that it is greater than S deposition at all these sites.
- If 90% confidence of non-exceedance is required, all 18 GB sites were exceeded with 1998-2000 deposition. For 50% confidence, there were 15 sites, and for 10% confidence, 12 sites. The choice of exceedance probability is fairly arbitrary, but will have a very large effect on perceptions of damage.
- Exceedance distributions looked close to normal, but the spreads of the distributions varied considerably.

- Overall, it appears from these sites that there is little scope for reducing the uncertainty attached to predictions of the current FAB model, because uncertainty is driven largely by natural variation in measured base cation concentrations. This analysis tells us nothing about whether the *structure* of the current model is correct.
- Uncertainty in exceedance can be reduced by decreasing the uncertainty in deposition.
-

2.3 Meta-analysis

Since we now have considerable experience in estimating uncertainty in critical loads at individual sites, some conclusions can be drawn by considering all the data together in a simple meta-analysis. The data meet the requirements for meta-analysis, as they can be expressed in a common metric representing the size of the effect we are interested in. Using the coefficient of variation for this purpose allows us to compare critical loads. Results have not been weighted in any way, as they have been generated using the same methods and are considered equally reliable. Coefficients of variation of all terrestrial critical loads calculated are shown in Figure 2.3.1. Statistics for the coniferous forest sites are shown in Table 2.3.1.

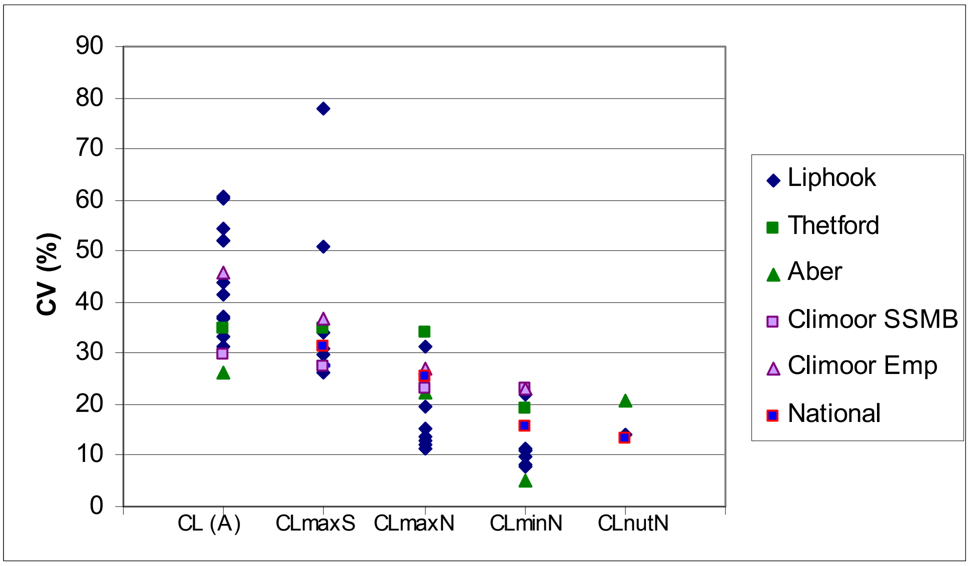


Figure 2.3.1: Coefficients of variation of terrestrial critical loads calculated during this study. All sites except Climoor are coniferous forest.

Figure 2.3.1 and Table 2.3.1 show that most uncertainty attaches to *CL(A)*, followed by $CL_{max}S$, $CL_{max}N$ and $CL_{min}N$. Coefficients of variation are generally surprisingly low, but there is some variation, with one value of 78 per cent for $CL_{max}S$. A restricted sample of three points for $CL_{nut}N$ also indicates relatively low uncertainty, but more data are clearly required.

Table 2.3.1: Statistics for the coefficient of variation (%) of terrestrial critical loads

	$CL(A)$	**$CL_{max}S$**	**$CL_{max}N$**	**$CL_{min}N$**	**$CL_{nut}N$**
Mean (%)	43.4	37.7	19.2	11.7	16.0
SD	11.4	16.7	8.4	5.4	5.1

Statistics are for the coniferous forest sites only.

The standard deviations in Table 2.3.1 indicate 'uncertainty about uncertainty'; that is, the variability in the mean uncertainty of this small sample of sites. The CVs are about 45 per cent for $CL_{max}S$, $CL_{max}N$ and $CL_{min}N$ and somewhat less for the others.

Similarly, Figure 2.3.2 and Table 2.3.2 show a meta-analysis for all the aquatic critical loads. For $CL_{max}S$ there is an outlier for Loch Grannoch, which has the lowest and most variable value in the data set for the key parameter 'non-marine base cation concentration'. This gives a very large CV of 145 per cent, and also the highest CV for $CL_{max}N$. Even with Loch Grannoch removed (Table 2.3.2), critical load values for lakes appear to be more uncertain than those for streams, possibly because they are sampled four times a year instead of 12. Compared to terrestrial critical loads, $CL_{max}S$ values for waters are generally lower, $CL_{max}N$ values about the same, and $CL_{min}N$ values higher. There is little variation in uncertainty for aquatic $CL_{min}N$ values, probably because the assumed distributions and variability are the same at each site, and the soil types (which determine two of the three parameters contributing to $CL_{min}N$) are very similar for these sites.

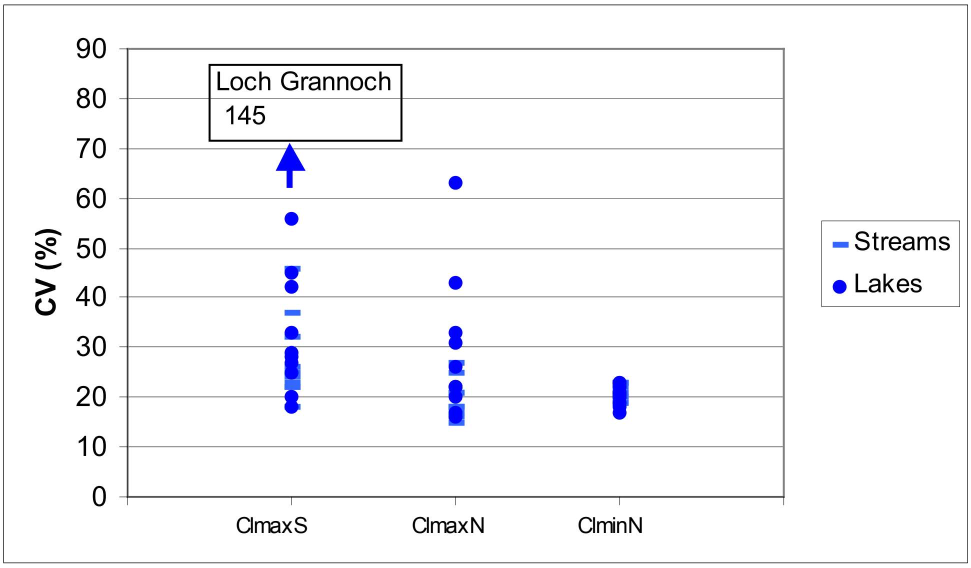


Figure 2.3.2: Coefficients of variation of aquatic critical loads calculated during this study

The data for aquatic systems show a pattern similar to that of terrestrial systems, in that $CL_{max}S$ is more uncertain than $CL_{max}N$, which is more uncertain than $CL_{min}N$. As discussed above, this is probably due to compensation of errors between $CL_{max}S$ and $CL_{max}N$.

Table 2.3.2: Statistics for the coefficient of variation (%) of aquatic critical loads

	CL$_{max}$S		***CL$_{max}$N***		***CL$_{min}$N***	
	Mean	**SD**	**Mean**	**SD**	**Mean**	**SD**
Streams	27.0	8.2	19.5	3.7	21.3	1.6
Lakes	42.6 (32.3)	35.8 (11.9)	28.6 (25.2)	13.9 (8.3)	20.5	1.9

Values in parentheses are without the Loch Grannoch outlier.

More meta-analysis can be performed on exceedance. Here it is not possible to use a coefficient of variation, since exceedance can be negative as well as positive, and it seems most appropriate to calculate cumulative distribution functions (CDFs) for each implementation of the Monte Carlo analysis. The CDFs for terrestrial exceedances of all types of critical load at Liphook, Aber and Climoor are shown in Figure 2.3.3.

The exceedance graph for Thetford requires a different scale and is shown as Figure 2.3.4.

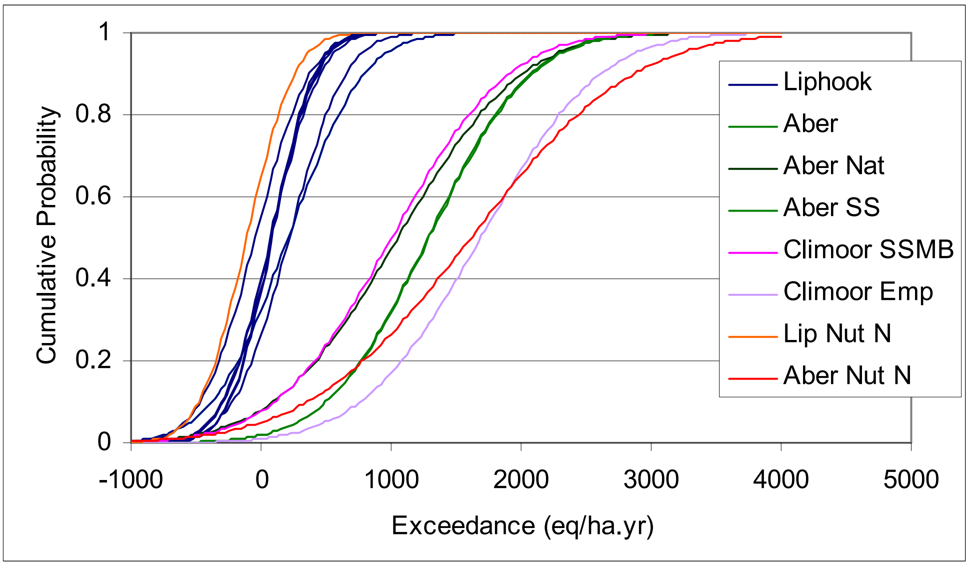


Figure 2.3.3: CDFs of critical load exceedance at terrestrial sites

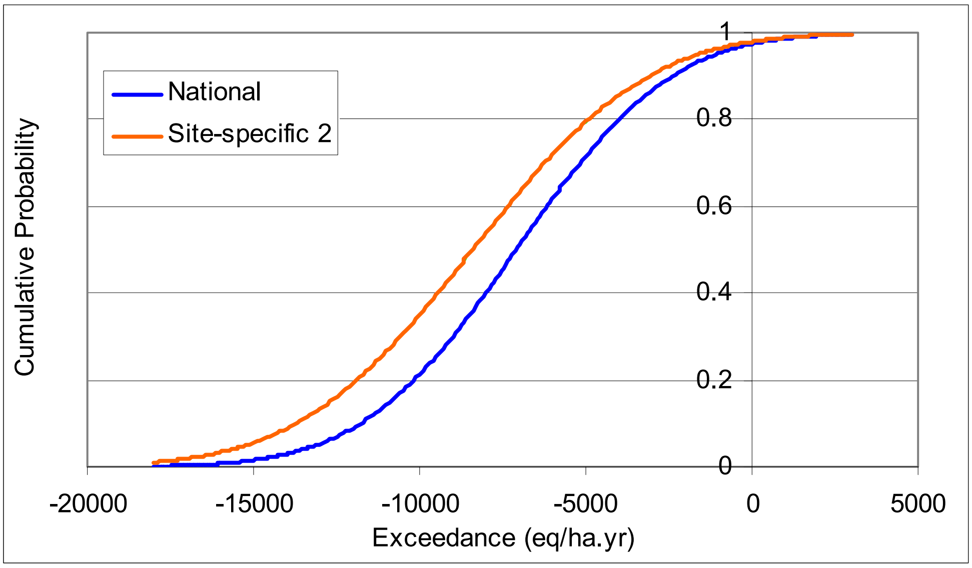


Figure 2.3.4: Exceedance at Thetford, using national or site-specific parameters

Figures 2.3.3 and 2.3.4 show several interesting features. At no site can we be 100% confident of exceedance or non-exceedance. Even at Thetford, a calcareous site with a high critical load, there is an exceedance probability of about 2%. The range from 0-100% probability is about 1.5 keq $ha^{-1}yr^{-1}$ at Liphook, but over 3 keq $ha^{-1}yr^{-1}$ at Aber and Climoor, and about 20 keq $ha^{-1}yr^{-1}$ at Thetford. These are significant ranges relative to the typical amounts of acid deposition. The implication of these ranges for regulation is that requiring a high probability of non-exceedance is likely to carry high costs. Figures 2.3.3 and 2.3.4 appear to show that the probabilistic range of exceedance estimates is site-dependent, and is likely to depend on which parameters are important and how well they can be estimated.

The cumulative distribution functions (CDFs) for the aquatic sites are shown in Figure 2.3.5.

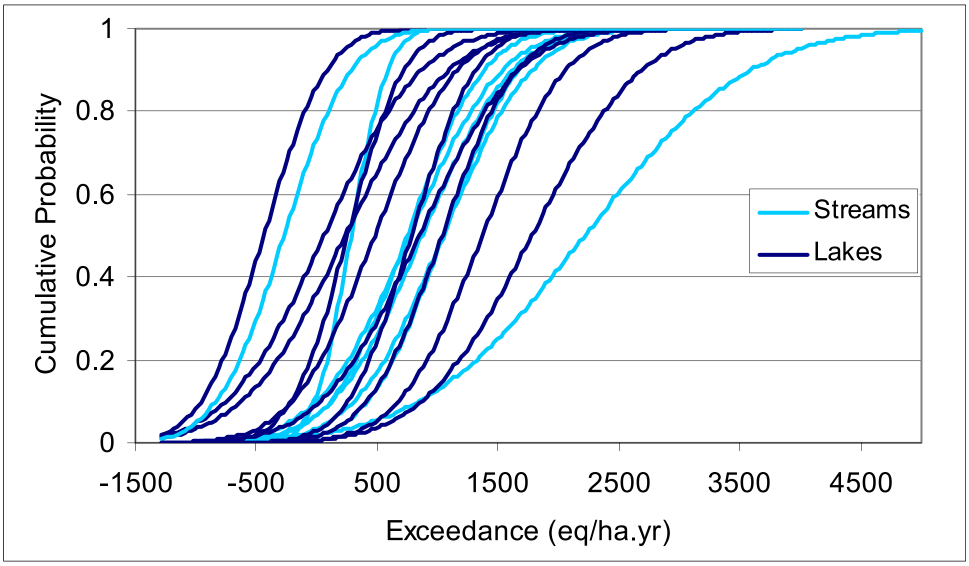


Figure 2.3.5: CDFs of aquatic critical load exceedance

As with the terrestrial sites, at no site can we be 100% confident of exceedance or non-exceedance. Even at sites in the North of Scotland (on the left of the diagram), there is a small probability of exceedance. Streams and lakes occupy the same envelope, and there appears to be a slight tendency for sites with higher exceedance to have a wider spread.

2.4 Conclusions

The detailed conclusions for site-specific critical load uncertainty are given at the end of the terrestrial and freshwater sections. Some general points can however be made:

1. The variability of the output parameters is nearly always smaller than the variability of the input parameters, often considerably. This means the spread of the critical loads is lower than might otherwise be expected, and they become manageable tools in spite of uncertain knowledge of input parameters.

2. The distributions, however, cover a wide range. They often look close to normal, though this should be checked further. Attempts to operate in the tails of these distributions, for example by specifying a high probability of non-exceedance of critical loads as an environmental target, are likely to carry high costs.

3. Sites differ in the parameters which have most influence on the uncertainty of the critical load outputs. For the FAB model, present-day base cation concentrations have the most influence at 21 out of 22 sites. For terrestrial sites, the range of dominant parameters is wider.

4. It may be possible to produce a typology of sites which would predict which parameters were most important to uncertainty.

5. Sites differ in whether deposition or catchment parameters are more important in determining the uncertainty in exceedance. As deposition decreases, however, the relative importance of catchment parameters will clearly increase.

6. From a very limited number of runs, the acidity critical loads for heathland and nutrient nitrogen critical loads for managed woodlands appear to have lower spreads than acidity critical loads for coniferous forests. This need checking on a national scale with default parameters.

7. The availability of on-site measurements can reduce the uncertainties in calculated critical loads significantly.

8. There is clearly an interaction between parameter uncertainty and other types of uncertainty. In FAB, the current base cation concentration parameter includes estimates of base cation weathering and deposition, which are treated separately in the SSMB, with the aid of certain assumptions. Thus, parameter uncertainty in FAB has probably been reduced by increasing the uncertainty in model structure.

3 National scale uncertainty

Summary

- This section describes the uncertainty studies based on national data sets for the case of: (a) critical loads and exceedances for managed coniferous woodland; (b) acidity critical loads for soils; (c) mapping broad habitats.
- Around 35% of managed coniferous woodland have a greater than 95% probability of exceedance for both acidity and nutrient nitrogen.
- 'Compensation of errors' results in the coefficient of variation for critical loads (acidity and nitrogen) for the managed coniferous woodland habitat being smaller than many of the input uncertainty ranges.
- Where woodland acidity critical loads are small, calcium deposition has a greater influence on the critical load values than calcium weathering.
- The variance due to soil type differs by region of the country; for example, soils in Snowdonia are less variable than soils in the Brecklands.
- In nearly half (42.5%) of 1x1 km squares in England and Wales, the critical load is the same whether the dominant or sub-dominant soil is used.
- Applying the national critical loads data to designated sites can give rise to anomalous values, particularly for larger sites with variable soils and habitats.
- The uncertainties in mapping broad habitats can be identified but not quantified.
- The habitat critical load maps provide a national picture of their distribution and sensitivity to acidification and eutrophication, appropriate for assessments at the national scale. They may be inappropriate for site-specific assessments.

3.1 Uncertainties in critical loads for managed coniferous woodland

This section describes the national scale uncertainty analysis in critical loads and their exceedances for both acidity and nutrient nitrogen, and using managed coniferous woodland on mineral soils as an example habitat type.

3.1.1 Input data and methods

The critical load calculations are based on the mass balance equations for acidity and nitrogen (Appendix A). The 2004 critical loads data were used for which, according to the national habitat maps developed for UK critical loads research, managed coniferous woodland on mineral soils is found in approximately 31,500 1x1 km grid squares across the UK. The deposition data, used in the critical load and exceedance calculations, are mapped on a 5x5 km grid; mean values for woodland for 1999-2001 have been used.

Maximum ranges of potential values relevant to the acidity critical load equations and subjective probability distributions are presented in Table 3.1.1. Plausible ranges of input parameters and their uncertainties were identified based on a literature survey, collected data, and interviews with experts.

In order to calculate the uncertainty in critical load exceedances, an estimate of the uncertainty in acidifying deposition parameters is also required. This is a continuing task. A complete assessment of uncertainty in measured deposition at the national scale has not yet been carried out. However, a subjective assessment by UK deposition experts suggests that a 95% confidence band around the deposition estimate for a given five km square of ± 30% is probably over-optimistic, ± 50% is optimistic and ± 100% is quite likely, all assuming a normal distribution (Smith, pers. comm.). The uncertainty analysis presented in this section applies the optimistic estimate (CV = 25%) of uncertainty to the national five km sulphur and nitrogen deposition data.
No spatial auto-correlations were taken into account between the parameters of neighbouring grid squares. This is probably a fair assumption except for the deposition parameters, where Smith *et al.* (1995) report that there is spatial auto-correlation in the national deposition data sets, with larger uncertainties expected in upland regions, but this has not been quantified for the current models and so spatial auto-correlation is not included in the analysis here.

Monte Carlo methods were once again used to propagate the uncertainty in the model parameters. The Monte Carlo simulations gave 5,000 critical load values ($CL_{max}S$, $CL_{min}N$, $CL_{max}N$, $CL_{nut}N$) and critical load exceedance values for every grid square.

Table 3.1.1: Summary of the uncertainty estimates and ranges identified in the inputs to UK critical load exceedance calculations.

Parameter*	Type of distribution	Uncertainty range
S_{dep}	Normal	25%[b]
N_{dep}	Normal	25%[b]
BC_{dep}	Normal	25%[b]
Ca_{dep}	Normal	25%[b]
BC_w	Rectangular	± 33-100%[a], depending on soil type
Ca_{corr}	Rectangular	± 10-100%[a], depending on soil type
BC_u	Normal	23%[b]
Ca_u	Normal	27%[b]
Q	Normal	23%[b]
$Ca:Al$	Rectangular	± 50%[a]
K_{gibb}	Rectangular	± 20%[a]
N_{immob}	Rectangular	± 50%[a]
N_{uptake}	Normal	27%[b]
N_{denit}	Rectangular	± 20-50%[a], depending on soil type
$N_{le(acc)}$	Triangular	-80% to +25%[c]

[*] Refer to List of Abbreviations
[a]Minimum value = mean value - x% * mean value
Maximum value = mean value + x % * mean value
[b]Coefficient of variation x% = (mean value / standard deviation) * 100
[c]Most likely values of the distribution have been assumed to be the default values

3.1.2 Critical load uncertainty

The mean critical load value for all coniferous woodland grid squares was determined from the Monte Carlo runs. Table 3.1.2 shows the mean $CL_{max}S$, $CL_{min}N$, $CL_{max}N$, $CL_{nut}N$ values together with their standard deviations and coefficients of variation.

Table 3.1.2: Means, standard deviations and coefficients of variation of the predicted coniferous woodland critical load function for all one km grid squares in the UK

Critical load	Mean (keq ha^{-1} yr^{-1})	Standard deviation (keq ha^{-1} yr^{-1})	Coefficient of variation (%)
$CL_{max}S$	2.15	0.67	31
$CL_{min}N$	0.45	0.07	16
$CL_{max}N$	2.60	0.66	25
$CL_{nut}N$	0.68	0.09	13

The coefficients of variation on the critical load function parameters are quite small. The CV for $CL_{max}N$ is smaller than $CL_{max}S$ which is the opposite of what might be intuitively assumed, since $CL_{max}N$ incorporates more sources of uncertainty. The CV for $CL_{nut}N$ is strikingly small. The reason for these results appear to be a compensation of errors mechanism, a phenomenon also noted by Suutari *et al*. (2001) and Skeffington *et al*. (2006). This is where a negative effect on one parameter is compensated for by a positive effect on another, which depends on the correlations assumed in the data.

The results of the analysis using 2004 critical loads are different for $CL_{max}S$ and $CL_{max}N$ from those of a similar study carried out for the 2001 data by Hall *et al*. (2004c), who reported a CV of 29% for $CL_{max}S$ and 21% for $CL_{max}N$. There are a number of possible reasons why these CVs, although comparable, are smaller (for $CL_{max}S$ and $CL_{max}N$) than those calculated for the current data set:

1. Heywood *et al*. (2006a) carried out the analysis for all coniferous woodland (on mineral, organic and peat soils), whereas the current analysis is for coniferous woodland on mineral soil only.
2. The underlying (deterministic) input data has changed. For example, the mean of the 2001 critical loads data set for $CL_{max}S$ was 1.83 keq ha^{-1} yr^{-1} and for 2004 it was 1.97 keq ha^{-1} yr^{-1}.
3. Different uncertainty ranges were used for the deposition parameters.
4. Different sequences of random numbers were used.
5. The habitat was mapped differently so that in 2001 there was 7,379 km^2 of coniferous woodland estimated in the UK, and in 2004 this estimate rose to 7,970 km^2 (for acidity).

The results for $CL_{min}N$ and $CL_{nut}N$ are almost identical to the earlier study (Hall *et al.,* 2004c).

3.1.3 Exceedance uncertainty

Figures 3.1.1(a), (b) and (c) show the frequency, cumulative and inverse cumulative distribution charts of exceedance values for a single one km grid square at Liphook. These charts represent different ways of presenting the same information and are discussed further in Section 6.

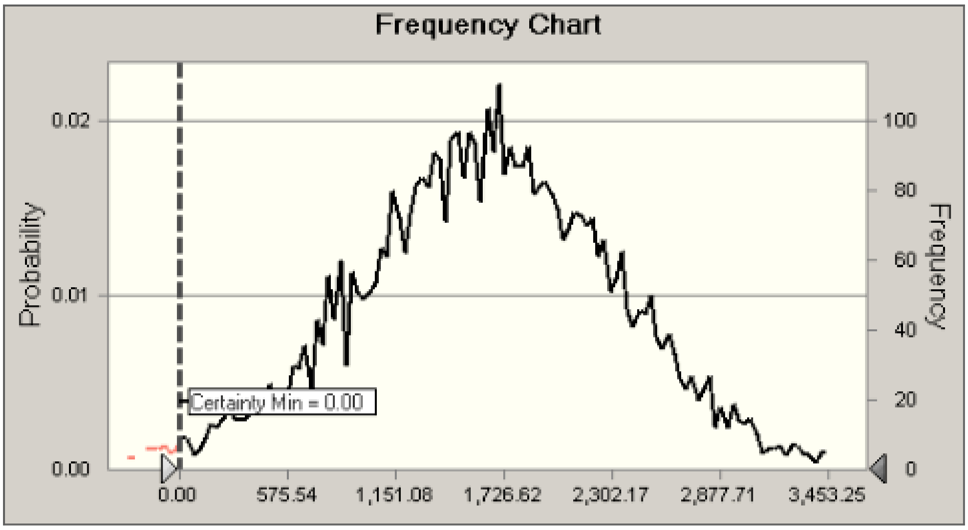


Figure 3.1.1(a): Frequency chart for one km grid square at Liphook. The x-axis shows critical load exceedance in eq.ha^{-1}yr^{-1} and the y-axis probability/frequency. The broken line shows zero exceedance.

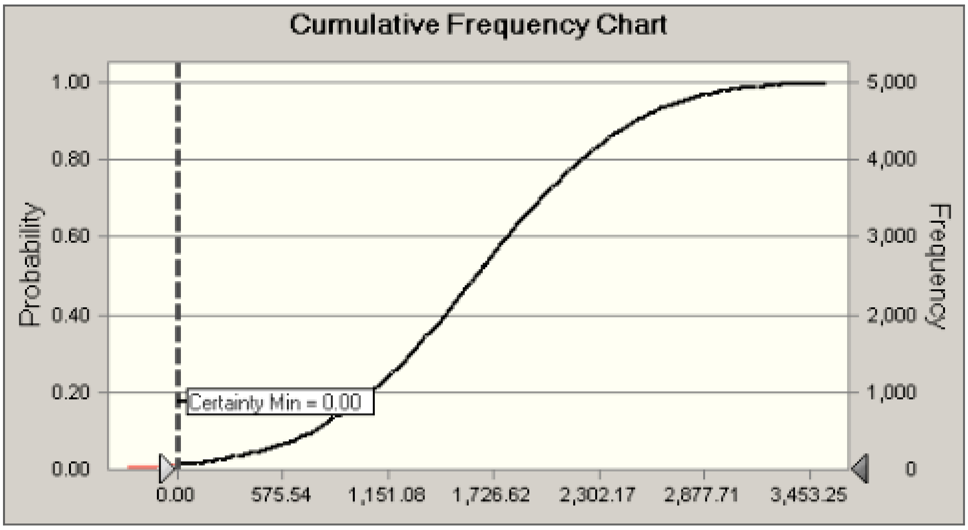


Figure 3.1.1(b): Cumulative frequency chart for one km grid square at Liphook. The x-axis shows critical load exceedance in eq ha^{-1}yr^{-1} and the y-axis probability/frequency. The broken line shows zero exceedance.

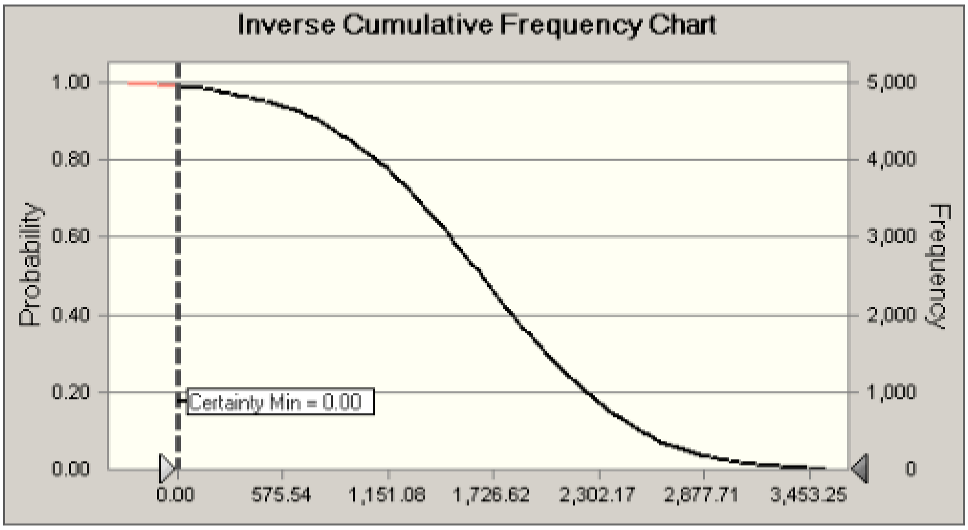


Figure 3.1.1(c): Inverse cumulative frequency chart for one km grid square at Liphook. The x-axis shows critical load exceedance in eq ha^{-1}yr^{-1} and the y-axis probability/frequency. The broken line shows zero exceedance.

The dashed lines on each chart represent zero exceedance. Figure 3.1.1 shows distributions which lie almost entirely on the right hand side of the dashed line, showing that Liphook has a very high probability of being exceeded.

The mean and standard deviation of the exceedance distribution for every 1x1 km^2 of coniferous woodland (on mineral soils) were generated from 5,000 Monte Carlo runs varying the critical load and deposition parameters simultaneously. The probability of exceedance, where the percentage of the cumulative frequency chart lies above zero, was calculated for every one km grid square. The results were then assigned to one of the following five classes:

- 0-5% probability: unlikely to be exceeded;
- 5-25% probability: relatively low risk of exceedance;
- 25-75% probability: potential risk of exceedance;
- 75-95% probability: relatively high risk of exceedance;
- > 95% probability: highly likely to be exceeded.

The results are shown in Figure 3.1.2(a) for acidity and Figure 3.1.2(b) for nutrient nitrogen. Those areas that have less than 5% probability of exceedance are those with a high degree of confidence that the critical loads are not exceeded; conversely, areas with more than a 95% probability of exceedance are the most certain to be exceeded. Figure 3.1.2 shows for acidity exceedance that 33% of coniferous woodland (on mineral soils) has a greater than 95% probability of exceedance and 10% has less than 5% probability of exceedance. For nutrient nitrogen, most (98%) of the country has a very high (>95%) probability of exceedance, implying that reduction in the emissions of nitrogen oxides and ammonia are necessary.

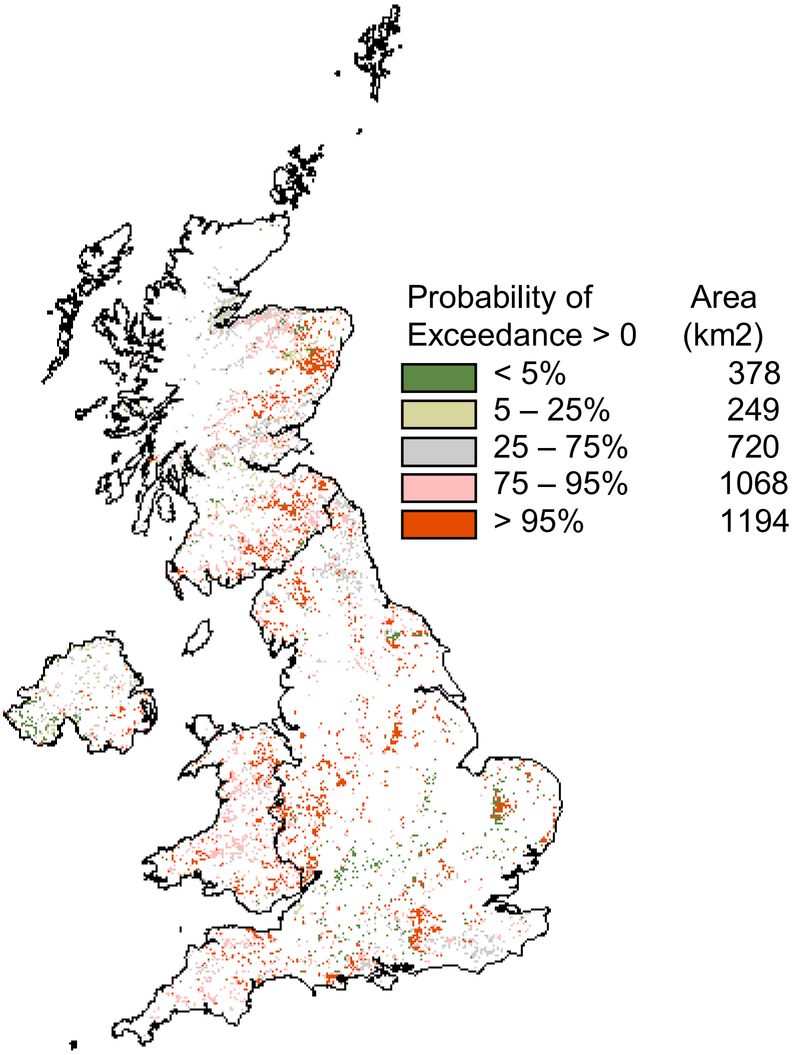


Figure 3.1.2(a): Probability of acidity exceedance for coniferous woodland on mineral soils using 1999-2001 measured deposition

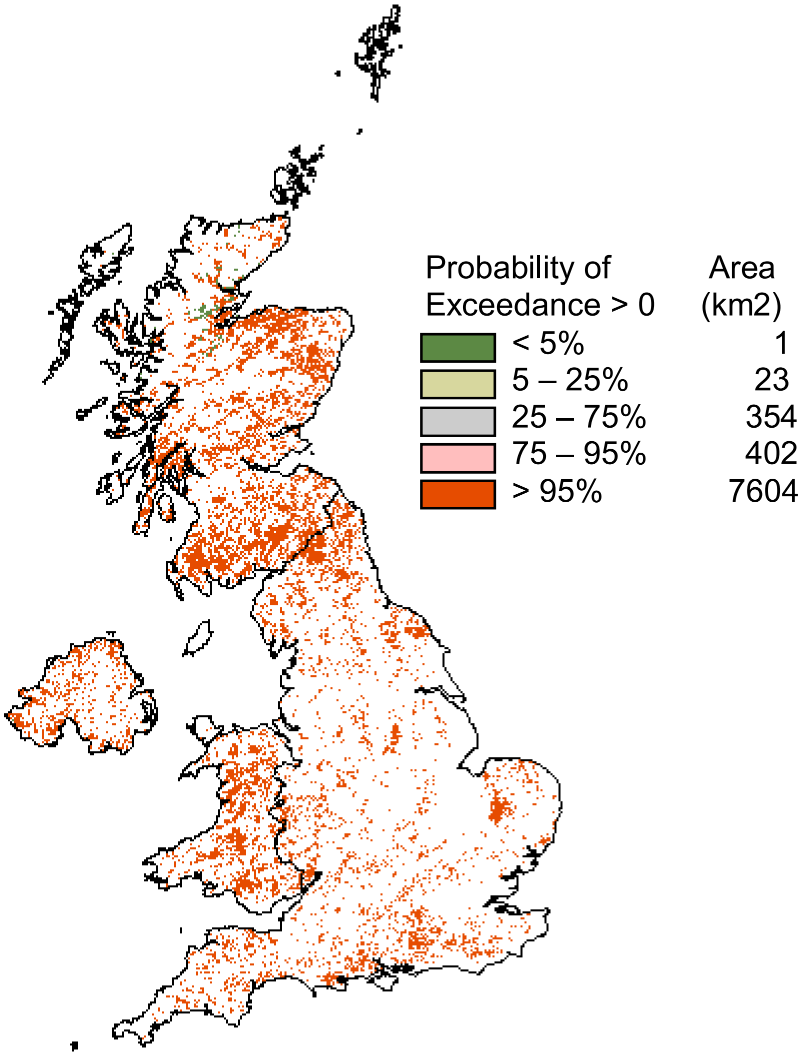
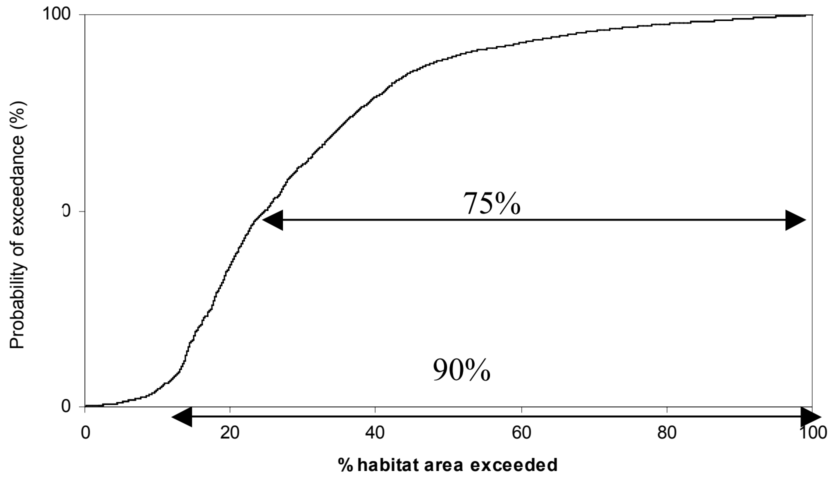


Figure 3.1.2(b): Probability of nutrient nitrogen exceedance for coniferous woodland on mineral soils using 1999-2001 measured deposition

Figures 3.1.3(a) and (b) show the cumulative frequency charts of the percentage total area of coniferous woodland exceeded for acidity or nutrient nitrogen. Cumulative frequency charts of habitat exceedance are generated by ranking the probability of exceedance for each grid cell from lowest to highest and calculating the cumulative sum of the areas. Figure 3.1.3(a) shows the probability of acidity exceedance on the y-axis and the cumulative sum of the area, normalised to the total habitat area on the x-axis. A precautionary approach would dictate that to be 95% certain of protecting all the areas at risk of acidification (that is, to protect all those areas with a greater than 5% probability of exceedance), 90% of the coniferous woodland (on mineral soils) would have to be protected. Adopting a 50% probability of exceedance, we would aim to protect 75% of coniferous woodland (all those areas with a greater than 50% probability of exceedance). The deterministic percentage area of coniferous woodland (the area we normally aim to protect) is 70%.

Figure 3.1.3(b) shows the cumulative frequency chart of habitat exceedance probability for nutrient nitrogen. The slope of the chart is initially very steep, indicating that most areas have a very high probability of exceedance. The deterministic percentage area exceeded is 93%. Taking a precautionary approach (protecting all areas with a greater than 5% probability of exceedance), we should aim to protect approximately 100% of coniferous woodland. Protecting all those areas with a greater than 50% probability of exceedance suggests we should be protecting about 98% of coniferous woodland. For nutrient nitrogen, there is a narrow range of areas of coniferous woodland we would wish to protect.

(a)



(b)

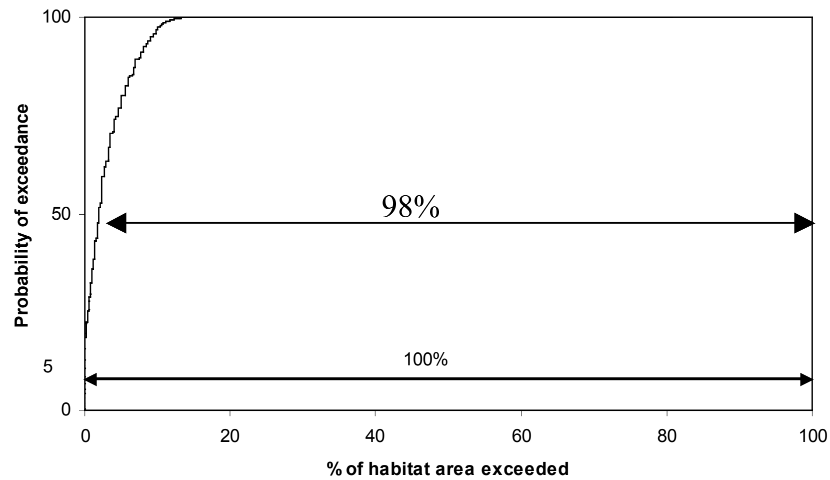


Figure 3.1.3: Cumulative frequency charts of the percentage total area of coniferous woodland exceeded for acidity (a) and nutrient nitrogen (b)

3.2 Relative importance of calcium weathering and deposition in low critical load areas

The site-specific uncertainty analysis showed that the SSMB equation for acidity critical loads for managed woodlands was very sensitive to the total calcium deposition parameter. At the site in question (Liphook), the weathering rates (base cation and calcium) were very low. A simple sensitivity analysis was carried out on national data sets to determine which parameter, calcium weathering or calcium deposition, had the greatest influence on the calculated critical load, particularly in areas where the weathering rates are small.

Plotting total calcium deposition against the acidity critical load for all managed conifer areas in the UK (Figure 3.2.1) shows that the critical load increases with an increase in calcium deposition, decreasing slightly once the critical load is above 2.0 keq $ha^{-1}yr^{-1}$.

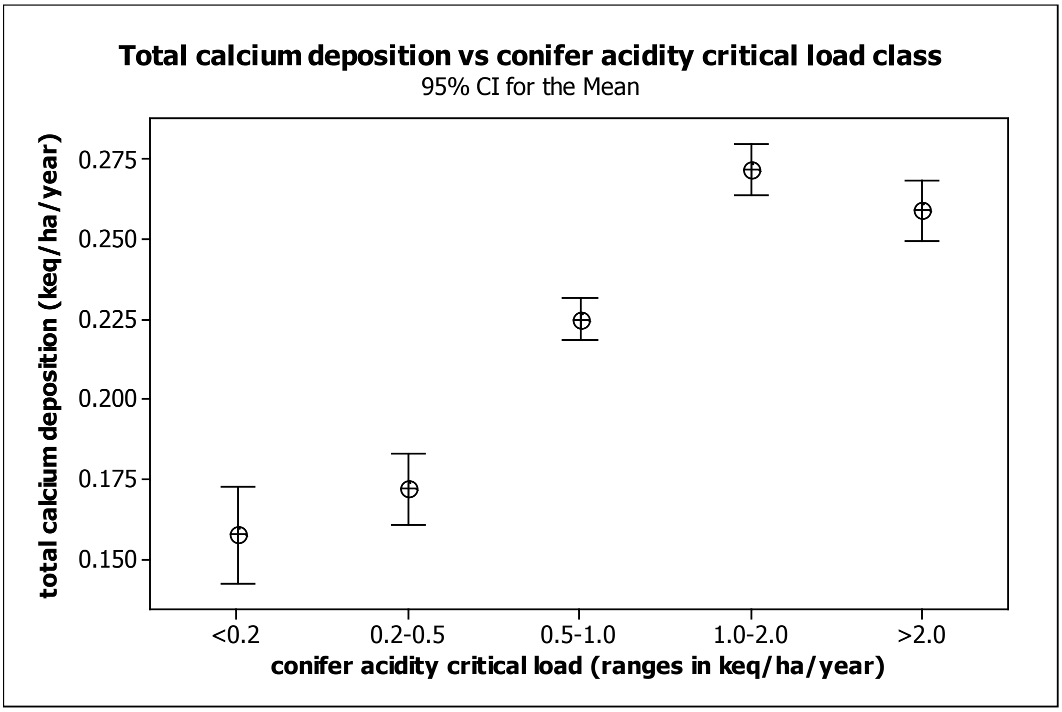


Figure 3.2.1: Total calcium deposition vs acidity critical load for managed conifers

Extracting the data for all one km managed conifer squares where the acidity critical load is ≤ 0.2 keq $ha^{-1}yr^{-1}$ yields 105 x 1 km^2 squares. The dominant soil in each of these squares is a mineral soil, so the same formulation of the SSMB equation would be applied. Other characteristics of these squares are given in Table 3.2.1 below.

Table 3.2.1: Characteristics of the 105 managed coniferous woodland 1x1 km squares with acidity critical loads ≤ 0.2 keq ha^{-1}yr^{-1}

Parameter	Minimum	Maximum#	Mean
Total Ca deposition (keq ha^{-1}yr^{-1})	0.15	0.17 (0.97)	0.158
Ca weathering (keq ha^{-1}yr^{-1})	0.01	0.02 (4.0)	0.01
Base cation weathering (keq ha^{-1}yr^{-1})	0.1	0.1 (4.0)	0.1
Run-off (m)	0.116	0.882 (3.4)	0.391

Value in brackets denotes maximum value nationally

Table 3.2.1 shows that for the 105 1x1 km squares being considered, the Ca deposition values are at least an order of magnitude greater than the Ca weathering rates. A simple sensitivity analysis was performed, calculating the acidity critical loads by varying the Ca deposition values in the range 0.1 to 1.0 keq ha^{-1}yr^{-1} and varying the Ca weathering values from 0.01 to 0.1 keq ha^{-1}yr^{-1} (Figure 3.2.2).

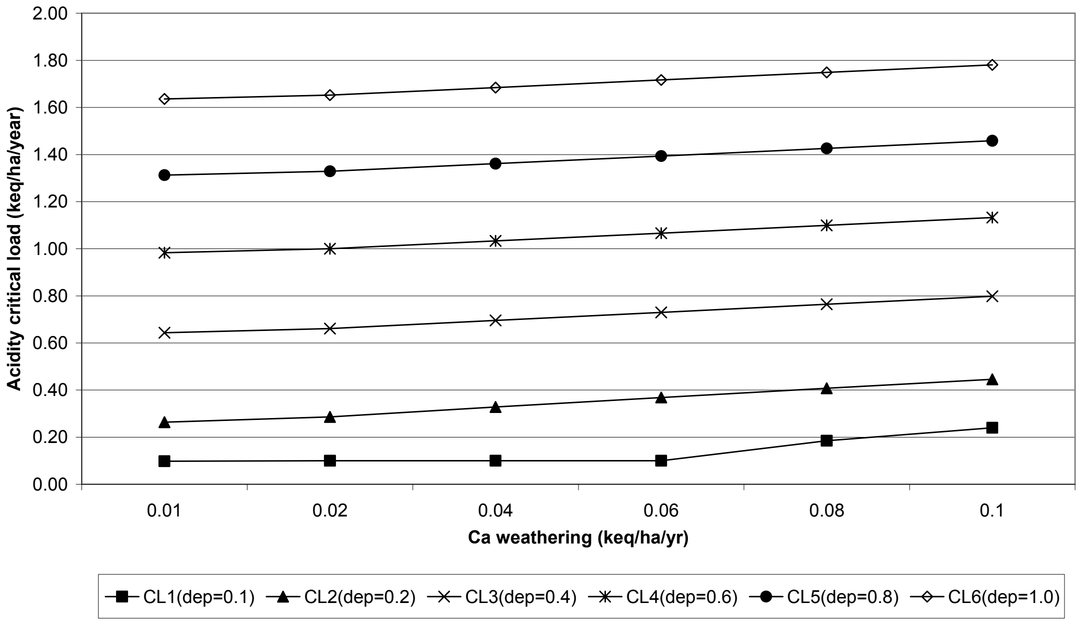


Figure 3.2.2: The effect of Ca deposition and Ca weathering rates on the calculation of acidity critical loads for managed coniferous woodland

Figure 3.2.2 shows that:

- Doubling Ca deposition from 0.1 to 0.2 keq ha^{-1}yr^{-1} is sufficient to move the critical load from the lowest category (critical load map class ≤ 0.2 keq ha^{-1}yr^{-1}) into the next highest category (0.2-0.5 keq ha^{-1}yr^{-1}). So, if we assume 50% uncertainty around the mean Ca deposition value of 0.158 keq ha^{-1}yr^{-1}, an increase in that value by 50% would result in a critical load in the class 0.2-0.5 keq ha^{-1}yr^{-1}.
- Because Ca weathering values are so small, a tenfold increase (from 0.01 to 0.1) is needed to shift the critical load into the next class (from map class ≤ 0.2 keq ha^{-1}yr^{-1} to class 0.2-0.5 keq ha^{-1}yr^{-1}).
- Where the critical load is small (less than 0.2 keq ha^{-1}yr^{-1}), Ca deposition is more important than Ca weathering.

3.3 Variance in acidity critical loads by soil type

3.3.1 Acidity critical loads for non-peat soils in England and Wales

The national acidity critical loads maps for broad habitats are based on the acidity critical loads for soils; for non-woodland habitats, the critical loads are set to the values for soils based on the empirical methods for non-peat and peat soils (Hall *et al.,* 2003a, 2004a). For woodland habitats, the empirical soil critical loads data provide the base cation weathering inputs to the SSMB equation. In all cases, critical loads are based on the dominant soil within each 1x1 km grid square.

The national soils database for England and Wales includes information on the percentage of all soil associations within each 1x1 km grid square, and critical load classes were originally assigned to each soil association (Loveland, 1991), with the exception of peat soils for which a different method is used (Hall *et al.,* 2003a, 2004a). Using these data, four acidity critical load maps have been generated at 1x1 km resolution for the non-peat soils in England and Wales, based on:

- the dominant soil
- the most sensitive soil
- the least sensitive soil
- an area-weighted mean of all soil types.

Maps are shown in Figures 3.3.1(a) to 3.3.1(d). All four maps show low critical loads in parts of Wales, Welsh borders, North West and South West England, and smaller areas in southern and eastern England. Similarly, there are areas across eastern England with high critical loads on all four maps. The maps based on the dominant soil (Figure 3.3.1(a)) and area-weighted (Figure 3.3.1(d)) are the most similar and are more 'speckled' in appearance, showing greater variability from one square to the next. The maps based on the least and most sensitive soil types tend to show larger areas within the same class. The breakdown of the number and percentage of grid squares in each critical loads class is given in Table 3.3.1.

Table 3.3.1: The number and percentage of 1x1 km grid squares by critical loads class for the acidity critical load maps of England and Wales for non-peat soils

Critical loads class (keq ha^{-1}yr^{-1}**)**	**Number and percentage of one km squares by critical loads class for acidity critical load maps based on:**			
	Dominant soil	Most sensitive	Least sensitive	Area-weighted
≤ 0.2	5,898 (4%)	14,092 (10%)	1,496 (1%)	3,374 (2%)
0.2 – 0.5	27,873 (19%)	40,932 (28%)	14,633 (10%)	21,554 (15%)
0.5 – 1.0	41,183 (28%)	48,359 (33%)	28,164 (19%)	40,869 (28%)
1.0 – 2.0	33,145 (23%)	21,110 (14%)	46,266 (32%)	37,609 (26%)
> 2.0	37,581 (26%)	21,187 (15%)	55,121 (38%)	42,274 (29%)

This information is easier to interpret in the graph (Figure 3.3.2) below.

Critical loads of acidity for soils based on:

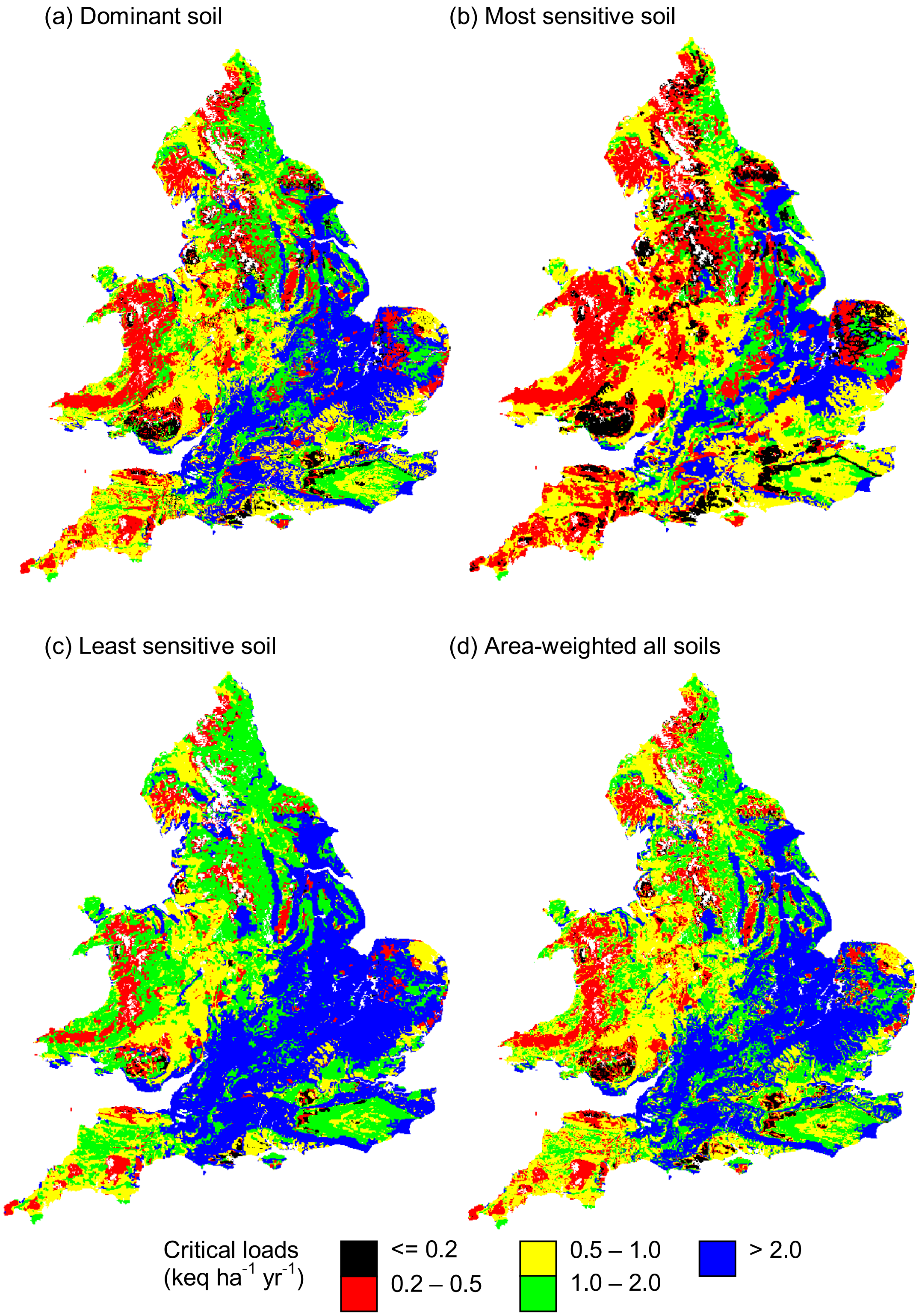


Figure 3.3.1: 1 x 1 km square critical loads for non-peat soils in England and Wales based on various criteria

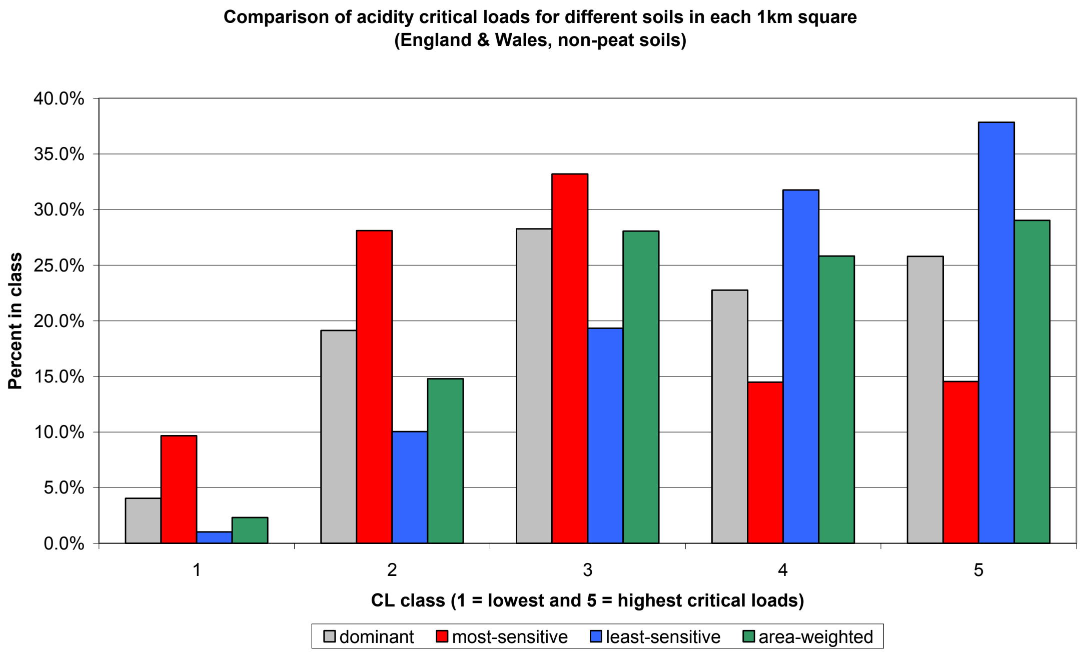


Figure 3.3.2: Comparison of acidity critical loads for different (non-peat) soils within each one km grid square in England And Wales

This figure shows, not surprisingly, that there is a higher proportion of grid squares with low critical loads for the map based on the most sensitive soil types, and conversely a higher proportion of squares with high critical loads for the map based on the least sensitive soil. The distribution of the area-weighted mean critical loads for all soil types within each 1x1 km square is similar to the distribution for the dominant soil map, due to the influence of the dominant soil on the area-weighted calculations.

Whilst Figure 3.3.2 shows the effect of different soil types on the critical load for the whole of England and Wales, this effect may vary from one region of the country to another. For example, we have compared the critical loads for two regions: around Snowdonia in North Wales, and the Breckland area in East Anglia. In each case, the data were extracted for a square or rectangular area around each region and the critical loads based on the dominant soil, least sensitive soil, most sensitive soil, and area-weighted mean critical load for all soils, compared (Figures 3.3.3(a)-(d)). These figures show that for Snowdonia, either all soils in each 1x1 km square are very sensitive to acidification, or only a single sensitive soil type is present, since on each graph the highest proportion of squares fall in the critical loads class with values in the range 0.2-0.5 keq $ha^{-1}yr^{-1}$. The results for Breckland vary according to the method used to set the critical load values, reflecting the greater range of soil types and sensitivities within this region. The results for both Snowdonia and Breckland differ from those for the whole of England and Wales.

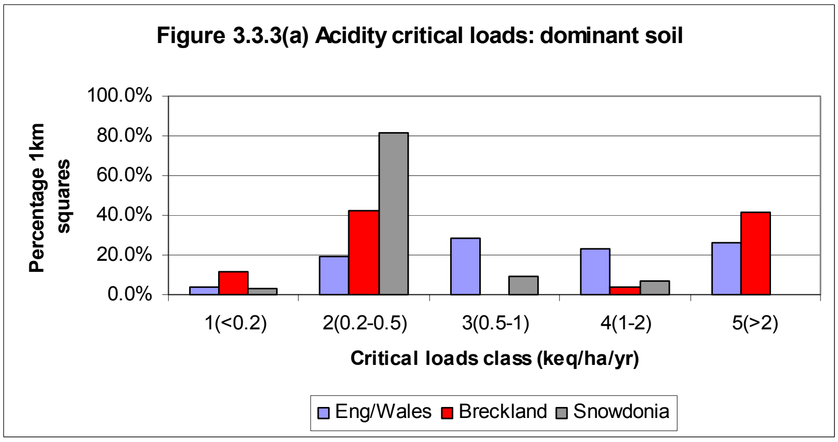


Figure 3.3.3(a) Acidity critical loads: dominant soil
Percentage 1km squares
100.0%
80.0%
60.0%
40.0%
20.0%
0.0%
1(<0.2)
2(0.2-0.5)
3(0.5-1)
4(1-2)
5(>2)
Critical loads class (keq/ha/yr)
Eng/Wales
Breckland
Snowdonia

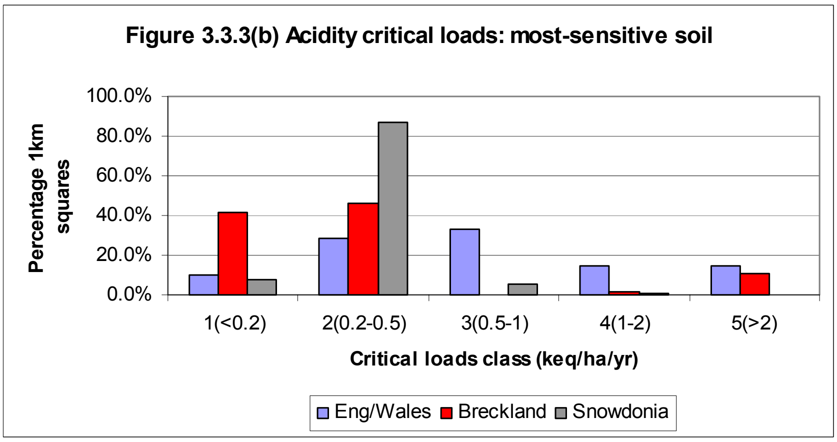


Figure 3.3.3(b) Acidity critical loads: most-sensitive soil
Percentage 1km squares
100.0%
80.0%
60.0%
40.0%
20.0%
0.0%
1(<0.2)
2(0.2-0.5)
3(0.5-1)
4(1-2)
5(>2)
Critical loads class (keq/ha/yr)
Eng/Wales
Breckland
Snowdonia

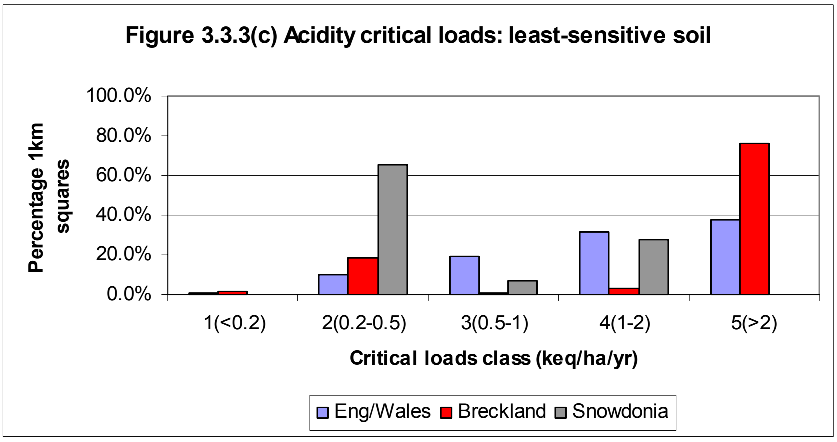


Figure 3.3.3(c) Acidity critical loads: least-sensitive soil
Percentage 1km squares
100.0%
80.0%
60.0%
40.0%
20.0%
0.0%
1(<0.2)
2(0.2-0.5)
3(0.5-1)
4(1-2)
5(>2)
Critical loads class (keq/ha/yr)
Eng/Wales
Breckland
Snowdonia

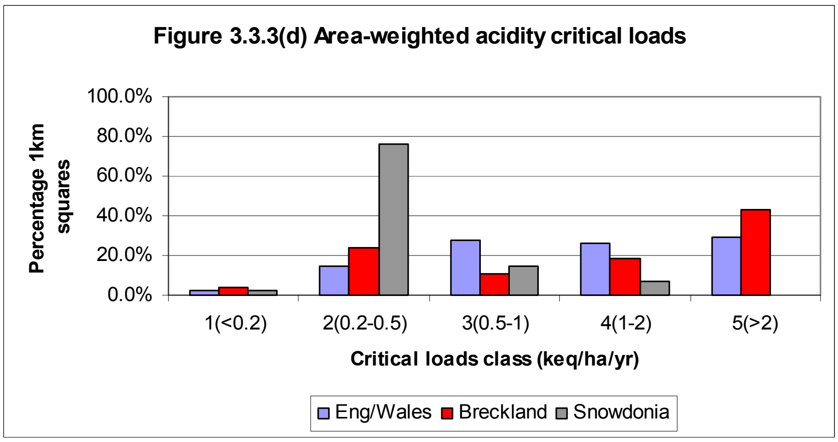


Figure 3.3.3(d) Area-weighted acidity critical loads
Percentage 1km squares
100.0%
80.0%
60.0%
40.0%
20.0%
0.0%
1(<0.2)
2(0.2-0.5)
3(0.5-1)
4(1-2)
5(>2)
Critical loads class (keq/ha/yr)
Eng/Wales
Breckland
Snowdonia

3.3.2 An index map comparing critical loads for different soil types

The maps generated in Section 3.3.1 are useful for visualising the impact the different soils within each 1x1 km grid square can have on the acidity critical loads maps. However, it is not easy to determine by eye where the critical loads are all the same or all different, for example. To provide this information an index map has been generated to compare the critical loads for the dominant soil type with the values based on the most sensitive, least sensitive and all soils (Figure 3.3.4). The breakdown of the percentage of squares in each category is given in Table 3.3.2 below. This shows that for over 40% of the non-peat dominated grid squares in England and Wales, the critical loads are the same for all soil types; the number of soil types occurring in a one km grid square can vary from one to 11. Just over 4% of squares have different critical loads for each soil type within them. The critical load for dominant soil is the same as the most sensitive or the least sensitive in equal proportions, at close to 25%.

Table 3.3.2: The number and percentage of one km grid squares in each category of the critical load index map.

Map class	Description of critical loads	Number and percentage one km squares
1	Same for all soils	61,892 (42.5%)
2	Different for all soils	6,049 (4.2%)
3	Dominant same as most sensitive and area weighted	67 (0.05%)
4	Dominant same as most sensitive	38,091 (26.1%)
5	Dominant same as least sensitive	39,587 (27.2%)
6	Dominant same as area-weighted	1 (<0.01%)

Figure 3.3.4: Critical load index map

3.3.3 Critical load variance map

The critical load index map provides one way of comparing the critical load values for different soil types, but it is not easy to interpret the detail visually. Therefore, this map has been simplified to give a three-class critical load variance map (Figure 3.3.5). This shows that for over 40% of grid squares, the critical loads of the sub-dominant soils are the same as the dominant soil. For a further 26% of squares, the critical load of the dominant soil is the same as that for the most sensitive soil occurring in the square; other sub-dominant soil types present may have higher critical loads. However, in these areas the user can be fairly confident in using the national map for local-scale assessments, although small areas of some soil types may not be represented as the one km soils data are derived from the 1:250,000 scale soil maps. The remaining 31% of squares, mapped in yellow, indicate areas where a sub-dominant soil has a lower critical load than the dominant soil. Therefore, the critical load for the dominant soil may be inappropriate for some site-level assessments in these areas and more information should be gathered on the soil types in the area of interest. This map could be useful when considering using national data for point or site-specific assessments (see Section 6).

Figure 3.3.5 Critical load variance map

3.4 Uncertainties in applying national critical loads to designated sites

To examine the use of national data at the site-specific scale, critical load values were extracted from the national maps for 17 Special Areas of Conservation (SACs) across England and Wales. The following acidity critical load values were compared:

(a) Four values extracted for the site-centroid (single 1x1 km square):

- critical load for the dominant soil;
- critical load for the most sensitive soil;
- critical load for the least sensitive soil;
- area-weighted critical load for all soils.

(b) Area-weighted critical loads for dominant soils for the whole site (all land parcels):

- for all site squares;
- for squares containing acid grassland on the national map;
- for squares containing bog on the national map;
- for squares containing dwarf shrub heath on the national map;
- for squares containing montane on the national map;
- for squares containing calcareous grassland on the national map;
- assuming all site squares contain unmanaged woodland;
- for squares containing unmanaged woodland on the national map.

The different critical load values are given in Table 3.4.1. The boxes shaded in grey denote the habitats for which there are designated features for that site. However, for some of these there are no critical loads on the national maps, due to the methods and data used to derive the national habitat maps. Four out of the 17 sites are dominated by peat soils and therefore critical loads for all soils, the least and most sensitive soils have not been assigned for these site centroids. The results for (a) above, and for the non-woodland habitats from (b) above, are also given in Figure 3.4.1. This shows that for most of the sites examined, the critical load values based on the methods listed above are very similar. The notable exceptions are:

Site 5: Ingleborough. This is a large site with varied vegetation cover. The values for the site centroid (dominant, minimum, area-weighted) are all lower than the area-weighted critical loads for the habitat areas of dwarf shrub heath and bog. The centroid maximum critical load is similar to the habitat critical load for calcareous grassland. The area-weighted value for the site (based on dominant soils) is about midway between the lowest and highest critical load values derived.

Site 11: Craven Limestone. The area-weighted value for the bog habitat is much lower than any of the other critical load values derived for the site, showing that in this case a value extracted and based on the site centroid would be inappropriate for applying to bog habitat.

Site 13: Breckland. The values for the dominant soil and the most sensitive soil at the site centroid are close to the habitat value for calcareous grassland. The value derived for the acid grassland habitat falls mid-way between the area-weighted values for both the centroid and the site as a whole. The critical load for the most sensitive soil at the site centroid is much lower than the values derived for the habitats at this site.

In addition, since critical loads for woodland habitats are based on a different method (simple mass balance SMB equation), there are just two values for comparison: (i) area-weighted critical loads assuming all site grid squares contain unmanaged woodland; (ii) area-weighted critical loads based on only those site grid squares containing unmanaged woodland on the national map. Ten of the 17 sites have designated areas of woodland and nine of these have unmanaged woodland according to the national habitat maps. One of these nine sites (no. 7) has the same critical load for both the unmanaged woodland squares and assuming all squares within the site are woodland; it is therefore possible that this site contains an area of unmanaged woodland within all the grid squares making up the site. There are small differences in the two critical load values for three of the nine sites, and larger differences for the other five sites (numbers 5, 9, 10, 11 and 14).

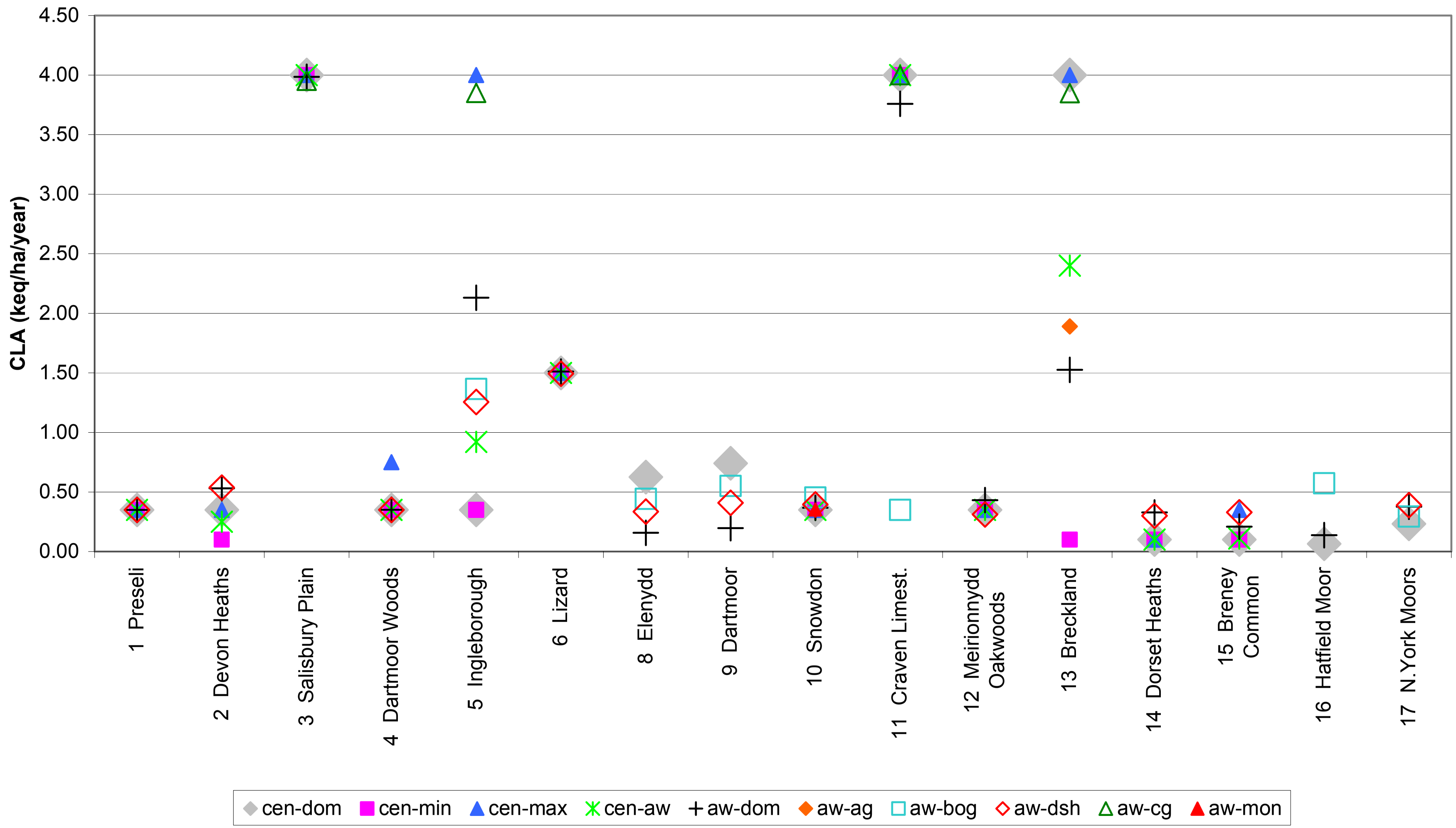
This exercise illustrates the problems that may arise if only the critical load for the dominant soil at the site centroid, and/or the area-weighted mean based on all squares and assuming the whole site is a single habitat type, is used in assessing the risk of acidification to habitats associated with designated features.

Table 3.4.1: Acidity critical loads for selected SAC feature habitats

SAC no.	Critical loads for site centroid				Area-weighted critical loads for site land parcels based on dominant soils							
	Dominant	Min	Max	Mean	All sqs	Calcareous grassland	Acid grassland	Dwarf shrub heath	Bog	Montane	All woodland sqs	Unmanaged wood
1	0.35	0.35	0.35	0.35	0.35		0.35	0.35			1.74	
2	0.35	0.10	0.35	0.25	0.53		0.34	0.54			1.04	0.85
3	4.00	4.00	4.00	4.00	3.99	3.95					10.74	10.82
4	0.35	0.35	0.75	0.35	0.35		0.34	0.35	0.35		1.70	1.61
5	0.35	0.35	4.00	0.92	2.13	3.85	1.99	1.26	1.37		3.97	6.74
6	1.50	1.50	1.50	1.50	1.51		4.00	1.49			2.64	
7	1.50	1.50	4.00	2.19	2.50	3.60					6.67	6.67
8	0.63	nd	nd	nd	0.16		0.49	0.34	0.44		1.22	1.93
9	0.74	nd	nd	nd	0.20		0.40	0.41	0.55		1.45	1.65
10	0.35	0.35	0.35	0.35	0.37		0.42	0.40	0.45	0.35	2.75	2.38
11	4.00	4.00	4.00	4.00	3.76	4.00	3.36	3.91	0.35		6.72	7.22
12	0.35	0.35	0.35	0.35	0.43		0.40	0.31	0.31		1.31	1.24
13	4.00	0.10	4.00	2.40	1.53	3.85	1.89				4.05	
14	0.10	0.10	0.10	0.10	0.33	4.00	0.30	0.30			1.29	0.94
15	0.10	0.10	0.35	0.11	0.21		0.17	0.33			1.27	
16	0.07	nd	nd	nd	0.14		0.26	0.43	0.57		0.54	
17	0.23	nd	nd	nd	nd	4.00	0.42	0.39	0.29		1.15	1.33

(a) The SAC names are given in Figure 3.4.1.
(b) All critical loads are in keq ha^{-1} yr^{-1}
(c) Boxes shaded in grey denote habitats for which there are designated feature(s) for that site; empty grey boxes denote sites where there is no critical load data for those habitats on the national maps.
(d) Critical loads for site centroid (single 1x1 km square) based on dominant, most sensitive (min), least sensitive (max) or all soils (mean).
(e) Area-weighted critical loads based on all areas of 1x1 km squares within a site (all squares), or those areas of 1x1 km squares containing the habitat on the national maps.
(f) Area-weighted critical loads for woodland habitats based on assuming all squares contain unmanaged woodland (all woodland squares) or only those where unmanaged woodland occurs on national maps (unmanaged wood).

Figure 3.4.1 Acidity critical loads for non-wood features of SACs (legend on next page)



Legend to Figure 3.4.1:

Critical loads based on soil type(s) for site centroid:

cen-dom	critical load for the dominant soil
cen-min	critical load for the most sensitive soil
cen-max	critical load for the least sensitive soil
cen-aw	area-weighted critical load for all soils

Area-weighted critical loads based on dominant soil type in each site square:

aw-dom	critical load for all site squares
aw-ag	critical load for squares with acid grassland on national map
aw-bog	critical load for squares with bog on national map
aw-dsh	critical load for squares with dwarf shrub heath on national map
aw-cg	critical load for squares with calcareous grass on national map
aw-mon	critical load for squares with montane habitat on national map

3.5 Uncertainties in mapping habitats

Since February 2003, national critical loads have been mapped for Biodiversity Action Plan (BAP) broad habitats sensitive to acidification and/or eutrophication. Other habitats are not considered in this study. The methods and data used to map the sensitive habitats nationally are described in detail by Hall *et al.* (2003a, 2004a) and will not be repeated here; however, the methods were agreed in consultation with representatives from the Department for Environment, Food and Rural Affairs (Defra) Terrestrial Umbrella (UK Research on the Eutrophication and Acidification of Terrestrial Ecosystems), Joint Nature Conservation Council (JNCC) CCW, Scottish Environmental Protection Agency (SEPA), other habitat experts and Defra. This section focuses on the uncertainties in the data and methods used.

Broad habitat maps were derived using the best available data to give national pictures of the main habitat types, adequate for national critical loads mapping purposes. As such, they may not include every small area of sensitive habitat at the regional, or especially, the local scale.

The broad habitat maps are based on combinations of national data sets and maps of land cover, forest land use, species distributions, the National Vegetation Classification (NVC), soils and altitude. Table 3.5.1 lists the data sets used in the generation of each broad habitat map. Individual data sets were generated at different times, for different purposes (not specifically for critical loads research) and at different spatial resolutions. Hence, there are inherent uncertainties in each data set as well as with the methods used to combine them.

Table 3.5.1: Data sets used in the derivation of Biodiversity Action Plan broad habitat maps for the UK

Broad habitat	Land cover[1]	Forest land use[2]	Species[3]	NVC[4]	Soils[5]	Altitude[6]
Acid grassland	✓		✓		✓	
Calcareous grass	✓		✓		✓	
Dwarf shrub heath	✓		✓		✓	
Bog	✓		✓			
Montane	✓			✓		✓
Supralittoral sediment	✓		✓			
Managed conifer	✓	✓				
Managed broadleaf	✓	✓				
Unmanaged wood	✓	✓				
Atlantic oak wood	✓	✓		✓		

[1] Land Cover Map 2000 (Fuller *et al.*, 2002)
[2] Forest Research data on forest land use (Forestry Commission, 2001, 2002a, 2002b)
[3] Species data from the Biological Records Centre, Centre for Ecology and Hydrology
[4] National Vegetation Classification data provided by the Joint Nature Conservation Council
[5] Soils data from the National Soils Resources Institute, Macaulay Institute and the Department of Agriculture and Rural Development, Northern Ireland
[6] Altitude data from a 50 m resolution digital terrain model

Habitat areas for freshwaters (lakes and streams) are based on digital catchment boundaries of the 1,722 sites sampled throughout the UK for critical loads research (Hall *et al.*, 2004a); uncertainties in the derivation of these areas are not considered in this report.

The sections below (Sections 3.5.1 to 3.5.4) discuss the:

- uncertainties in individual data sets;
- uncertainties in the methods used to map broad habitats;
- co-location of the broad habitat types;
- critical load values by broad habitat.

3.5.1 Uncertainties in individual data sets

Land Cover Map 2000 (LCM2000)

LCM2000 is a land cover map based on a thematic classification of spectral data recorded by satellite images. External data sets were used in its generation to help refine the spectral classification. The map is based on 25 m resolution imagery, subsequently converted to land parcels (polygons) and classified by land cover type. A full description of the derivation of LCM2000 is given by Fuller *et al.* (2002). LCM2000 identified 16 ‘target classes’ which were divided into ‘subclasses’ enabling the full complement of broad habitats to be mapped (Table 3.5.2).

For mapping BAP broad habitats for national critical loads research, one km summary land cover data have been used, which provide the area of each subclass in each one km square of the UK; the polygon land cover data are too detailed and too large to work with at the national scale. Thus, LCM2000 forms the base map for all the terrestrial habitats mapped for critical loads, enabling areas of each habitat to be determined at the one km and national scale, for quantifying the potential impact of deposition scenarios on habitats of high conservation value.

Fuller *et al.* (2002) examined the correspondence between LCM2000 and field survey data from Countryside Survey 2000 (CS2000) to provide some estimates of uncertainty in the LCM2000 data. CS2000 provides field survey data for 569 x 1 km grid squares across the UK (Haines-Young *et al.*, 2000) and the results of this survey have been compared with LCM2000 data for the same areas. However, it must be noted that such correspondences are not a true measure of accuracy, since the CS2000 was not used as 'ground truth' data for generating LCM2000. Fuller *et al.* (2002) reported an overall correspondence of around 85 per cent at the target class level. The correspondences varied across the UK, with the largest differences in upland areas where both field and satellite mapping are the most problematic and where some of the species most sensitive to air pollution effects are to be found. There are uncertainties in the areas of broad habitats determined from both LCM2000 and CS2000, since the habitats are artificial constructs rather than something real on the ground, consisting of aggregations of spectral classes in terms of LCM2000, and ground surveyors judgements in the case of CS2000.

Table 3.5.2: Broad habitats and their relationship to LCM2000 target classes and subclasses

Broad habitat	LCM target class	LCM subclasses
22. Inshore sublittoral	Sea/Estuary	Sea/Estuary
13. Standing water/canals	Water (inland)	Water (inland)
20. Littoral rock	Littoral rock and sediment	Littoral rock
21. Littoral sediment		Littoral sediment
		Saltmarsh
18. Supralittoral rock	Supralittoral rock & sediment	Supralittoral rock
19. Supralittoral sediment		Supralittoral sediment
12. Bogs	Bogs (deep peat)	Bogs (deep peat)
	Dwarf shrub heath (wet/dry)	Dense dwarf shrub heath
10. Dwarf shrub heath		Open dwarf shrub heath
15. Montane habitats	Montane habitats	Montane habitats
1. Broadleaved woodland	Broadleaved woodland	Broadleaved/mixed woodland
2. Coniferous woodland	Coniferous woodland	Coniferous woodland
4. Arable and horticultural	Arable and horticultural	Arable cereals
		Arable horticulture
		Non-rotational horticulture
5. Improved grassland	Improved grassland	Improved grassland
	Neutral / calcareous semi-natural / rough grasslands	Setaside grass
6. Neutral grassland		Neutral grass
7. Calcareous grassland		Calcareous grass
8. Acid grassland	Acid grass and bracken	Acid grass
9. Bracken		Bracken
11. Fen, marsh and swamp	Fen, marsh and swamp	Fen, marsh and swamp
17. Built-up areas, gardens	Suburban & urban	Suburban / rural developed
		Continuous urban
16. Inland rock	Inland bare ground	Inland bare ground

The use of external data sets to refine the spectral classification can lead to other uncertainties, since the data sets may be of different resolutions and be designed for other purposes, as well as having their own uncertainties. Such data were used for mapping the following LCM2000 classes:

(i) Semi-natural grasslands and bracken

These habitats present problems in their distinction using satellite imagery alone. For example, acid, neutral and calcareous grasslands give no consistent spectral characteristics by which they can be identified. An ancillary data set of soil acid sensitivity classes (Hornung *et al.*, 1995) was used, though it lacked sufficient discrimination due to the soil pH ranges on which the classes were based, that is:

- pH < 4.5 highly acid sensitive - used to determine acid grasslands;
- pH > 4.5 and < 5.5 moderately acid sensitive - used to identify neutral grasslands, but really slightly acid;
- pH > 5.5 low acid sensitive – used to identify calcareous grasslands, but would also contain some neutral areas.

Hence, for the national critical loads habitat mapping, the LCM2000 grassland data were combined with other data sets, to improve the distributions of acid grassland and particularly calcareous grasslands; neutral grasslands were not mapped for critical loads purposes.

(ii) Heath, moor and bog

The LCM2000 heath and moor classes were distinguished from the bog class using a British Geological Survey map[1] showing peat drift greater than 0.5 m deep; hence the LCM2000 bog class is more correctly 'bog (deep peat)'. Fuller *et al.* (2002) report that this method led to a conservative estimate of the area of bog broad habitat, compared to the CS2000 estimate.

(iii) Montane

This habitat class was defined as all vegetated ground above 600 m, and therefore made use of a digital terrain model to impose the altitude constraint to the class.

<u>Forest Research (FR) forest land use data</u>

LCM2000 distinguishes between coniferous and broadleaved woodland. However, for critical loads mapping (and critical load calculations) it is important also to distinguish between managed and unmanaged woodland areas, and this distinction is not possible from satellite-derived data alone. FR provided one km databases of the areas of managed coniferous, managed broadleaved and unmanaged (ancient and semi-natural) woodland. To maintain consistency with using LCM2000 as the base map for the UK critical load habitat maps, the FR data were used to determine the ratio of the different woodland types for each one km square. The ratios were then applied to the LCM2000 woodland data. For example, if the FR data for a single one km square consisted of 20 ha managed conifer, 20 ha managed broadleaved and 10 ha of unmanaged woodland and the total area of all woodland on LCM2000 for the same square is 70 ha, the calculated woodland areas applied to the grid square would be: 28 ha managed conifer (0.4 x 70), 28 ha managed broadleaved

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1 http://www.bgs.ac.uk/products/digitalmaps/digmapgb_drift.html

(0.4 x 70) and 14 ha unmanaged woodland (0.2 x 70). The areas of the different woodland types mapped by LCM2000 and FR for GB are compared in Section 3.5.2.

Species distribution data

Work undertaken within the Biological Records Centre at Monks Wood identified all species associated with each BAP broad habitat; species may be associated with more than one habitat type. The species data for each habitat were mapped by BRC to give the percentage of species in each 10 km grid square of the UK, adjusted for the latitudinal gradient in species diversity (Hall *et al.*, 2003a,b). In combining the species data with LCM2000 data, only those 10 km squares with a percentage of species above a specified threshold value were used. This aimed to represent the key areas of habitats nationally. For example, calcareous grassland is assumed to be present in a 10 km square, provided at least 50% of the associated species occur; for acid grassland, dwarf shrub heath and bog, a threshold of 40% was applied (Hall *et al.*, 2003a). A slightly different approach was used for supralittoral sediments, where the distributions of five key dune grassland species were used to represent the habitat.

Once the species distributions were defined, they were used to generate 'habitat masks' at 10 km resolution. The final habitat areas were subsequently mapped by selecting the one km squares of the appropriate LCM2000 class(es) that fell within each 10 km habitat mask square. Therefore, within each 10 km square it was assumed that the species representing the habitat could occur in every one km grid square; hence the habitat area could be overestimated. Conversely, where squares were omitted from the mapping procedure due to having less than the percentage threshold value, or one km squares occurred outside the 10 km squares, the habitat areas could be underestimated.

National Vegetation Classification (NVC)

Two of the habitats mapped used 10 km data of the distribution of NVC communities (Rodwell, 1990, 1991, 1992, 2000), namely *Racomitrium heath* for the montane map and woodland class W17 (*Quercus petraea – Betula pubescens – Dircranum majus*) for Atlantic oak, mapped as part of the unmanaged woodland for nutrient nitrogen. As for the species distribution data, we assumed that the NVC communities occupied all one km grid squares within each 10 km square for habitat mapping purposes, so again areas of habitat may be under- or overestimated.

Soils data

The acidity critical load values for the non-woodland terrestrial habitats are all based on the one km empirical soil acidity critical loads map. Critical loads are assigned according to the mineralogy and weathering rate of the *dominant* soil in each one km grid square. For squares dominated by peat soils, a method based on a critical soil solution pH of 4.4 (Hall *et al.*, 2004a; Calver, 2003; Skiba and Cresser, 1989; Calver *et al.* 2004) is applied. Sub-dominant soils in the one km grid squares may have lower or higher critical loads. Acidity critical loads for the woodland habitats are calculated using a SMB equation and use the soil critical load values as the weathering rate inputs to the equation. Where more than one habitat is mapped in a single one km grid square, the acidity critical loads are still based on the dominant soil type, even though the habitats may not all occur on the same soil. This is a limitation due to the spatial resolution of available soils data and the lack of knowledge on the precise relationships between habitat types and soil types. CEH have addressed the issue of soil and habitat relationships for the setting of appropriate critical loads under a separate contract with the Environment Agency (*An Investigation into the Best Method to Combine National and Local Data to Develop Site-Specific Critical Loads*, see Wadsworth and Hall, 2005).

To avoid setting unrealistically low acidity critical loads to areas of calcareous grassland, grid squares with a soil acidity critical load value less than 2.0 keq ha^{-1} yr^{-1} were removed from the calcareous grassland map. However, these grid squares were included in the habitat distribution map for nutrient nitrogen, where the critical load value assigned is not dependent on soil type.

An analysis of the dominant soil types within the one km grid squares of the bog broad habitat in England and Wales shows 55.7% of squares to be dominated by peat soil, 33.5% by organo-mineral soils (mineral soils with a peaty top) and 10.8% by mineral soils. Closer examination of the soils data (England and Wales only) shows that 30% of the squares dominated by mineral or organo-mineral soils contain some peat as a sub-dominant soil, leaving 14.4% of bog habitat squares that do not contain any peat soil. Hence, as the bog habitat could be expected to occur on peat soils only, the critical load values for the squares dominated by mineral or organo-mineral soils may be inappropriate for this habitat. Critical loads calculated for peat soils based on critical pH may give critical load values higher or lower than the critical load for the dominant soil based on the mineralogy and weathering rate. In areas of higher run-off, critical loads for peat soils may be higher than empirically derived values for the dominant soil, due to the dependence of the critical pH method on run-off.

In setting the nutrient nitrogen critical loads by EUNIS habitat class (Davies and Moss, 2002; Achermann and Bobbink, 2003), it was necessary to distinguish wet and dry areas of acid grassland and dwarf shrub heath. This was done using a one km map of classes of the Hydrology of Soil Types (HOST; Boorman *et al.*, 1995). These HOST classes are also based on the dominant soil type in each one km grid square; other soil types present may have different hydrological characteristics. Hence there are unquantified uncertainties in the classification of wet and dry habitat areas.

The key problem here is the lack of data, at the national scale, on the co-location of soils and habitats.

In addition to the uncertainties arising from how the data are used and applied, there will be uncertainties in the raw soils data; the one km soil databases generated by the soil surveys are derived from 1:250 000 scale soil maps and there is no estimate of the accuracy of these maps and data.

<u>Altitude data</u>

The LCM2000 montane class was defined using altitude criteria and selecting all vegetated areas above a threshold of 600 m. However, the JNCC description of the montane broad habitat states "an altitude limit is not a suitable marker for the start of the montane zone as the lower altitude limit of the zone varies in different parts of the UK. Therefore the presence of arctic/alpine species is used to define these types."

For critical loads mapping purposes three data sets were combined to estimate the habitat distribution, following discussion with habitat experts and trying a number of options. The final map was based on LCM2000 montane and inland bare ground classes that fell within 10 km squares of the NVC *Racomitrium heath* distribution and were above an altitude of 600 m as defined from a 50 m (horizontal) resolution digital terrain model (DTM). Uncertainties in the DTM have not been quantified, but are unlikely to be significant given other uncertainties.

3.5.2 Uncertainties in methods used to map broad habitats

The LCM2000 data provide the area of each land cover class, and therefore each broad habitat, within each one km grid square. For critical loads research, the areas of broad habitats are required for assessing the potential impacts of deposition scenarios across the UK. Hence, LCM2000 is used as the base map for all terrestrial broad habitat maps for critical loads work.

When overlaying LCM2000 with 10 km ancillary data (such as species distributions, NVC class data), only those one km LCM2000 grid squares that fall within the selected 10 km squares are included in the habitat maps. The woodland habitat areas mapped were restricted to where data existed on both LCM2000 and the FR maps, with the exception of NI where only LCM2000 data were available.

The main uncertainties associated with the methods used are:

- combining data of different resolutions, such as one km and 10km;
- combining data from different sources and designed for different purposes;
- applying a threshold in selecting the 10km squares representing the species for a habitat;
- overestimating habitat areas by assuming species data (or NVC communities) occupy all one km grid squares within each 10 km square selected;
- underestimating habitat areas by only including one km land cover squares that fall within 10 km species (or NVC) squares - the LCM2000 classes may occupy other areas;
- underestimating the area of woodland habitats by only selecting one km grid squares where both LCM2000 and FR woodland area data exist.

Whilst the uncertainties can be identified, it is not possible to quantify the uncertainty in the areas of habitats mapped. However, comparisons with other data sets can be made. For example, the Countryside Survey CS2000 uses 569 x 1 km squares as a stratified random sample; the stratification used is the ITE Land Classification in which each square is allocated to one of 40 different land classes, 24 from England and Wales and 16 from Scotland. National estimates for broad habitat extent were generated using sample data by calculating the mean and standard errors of the habitat areas for each land class; these sample means were weighted by the area the land class occupies nationally and the results summed to produce a national total (Howard *et al.*, 2003). For the non-woodland terrestrial habitats, we compared the habitat areas from LCM2000, CS2000 and the critical load broad habitat maps. For the woodland habitats, we included the FR National Inventory of Woodland and Trees. However, it should be noted in the comparison and interpretation below that no one data set can be considered to be the ‘correct’ one. Each data set has been generated at different times, using different methods and for different purposes and end-users. The comparison is provided for information only.

<u>Non-woodland habitats (Table 3.5.3)</u>

Table 3.5.3 shows that LCM2000 data give the largest areas for all non-woodland habitats for the UK, with the exception of bog. There are differences in the way the

bog and dwarf shrub heath habitats are identified and mapped by LCM2000 and CS2000. Bog is defined as deep peat in LCM2000 and is identified using both satellite data and a peatland mask based on British Geological Survey drift mapping (Section 3.5.1). Hence, the area of bog mapped is smaller than dwarf shrub heath. For CS2000, the bog and dwarf shrub heath habitats have been identified floristically on the basis of indicator species, resulting in a larger area of bog than dwarf shrub heath being recorded nationally.

LCM2000 produces the largest estimate of calcareous grassland. This may be influenced by the use of pH 5.5 to indicate low sensitivity. However, distinguishing different grassland types is not always easy in the field. Refining the LCM2000 distribution using species data (CL results) reduces the habitat area, though this is still larger than the CS2000 estimate.

The LCM2000 and CL areas of the montane habitat are considerably larger than the CS2000 estimate; again, this may be influenced by the use of the 600 metre elevation rule.

In summary, the non-woodland habitat maps derived for critical loads work are greater than those estimated from CS2000, with the exception of bog habitat for the reasons given above. The various refinements applied to CL habitat maps all have the effect of reducing the area, while no refinement increases the area; this is especially noticeable for calcareous grassland.

Woodland habitats (Table 3.5.4)

Table 3.5.4 shows good correspondence between LCM2000 and CS2000 areas for all woodland types. Fuller *et al.* (2002) explain that the extent and direct (spatial) agreement are similar for coniferous woodland, mainly because this woodland type tends to be planted in larger blocks. Although the extent of the broadleaved, mixed and yew woodland category is similar for both LCM2000 and CS2000, direct agreement is lower because many woodlands of this type tend to be smaller than the minimum mappable unit of 0.5 ha used by LCM2000, compared to that for CS2000 of 0.04 ha.

The FR data give a lower total area of woodland for GB, at 83% of the LCM2000 total. This difference may be due, at least in part, to the data sets provided by FR excluding woodlands less than two ha in area. In addition, the FR data map land use, not land cover. The CL map areas are considerably lower because of the LCM2000 one km grid squares selected due to their co-location with FR data; the total area of woodland on the CL maps is equivalent to 81% of the FR total, but only 67% of the LCM2000 total.

Table 3.5.3: Comparison of areas of non-woodland habitats for the UK

Broad habitat	Habitat areas (km^2) according to:			CL as percentage of CS2000	CL as percentage of LCM2000
	CS2000[(1)]	LCM2000[(2)]	CL[(3)]		
Acid grassland	13,240	15,793	15,334	116%	97.1%
Calcareous grassland	660	11,107	3,577	542%	32.2%
Dwarf shrub heath	15,000	27,023	24,703	165%	91.4%
Bog	23,670	5,657	5,463	23.1%	96.6%
Montane	490	3,971	3,054	623%	76.9%
Supralittoral sediment	1,930	2,708	2,128	110%	78.6%

(1) From: Haines-Young *et al.*, 2000, Table 2.1
(2) From: Fuller *et al.*, 2002, Table 5.
(3) From: Hall *et al.*, 2004b

Table 3.5.4: Comparison of woodland habitat areas (km^2) for GB[(a)]

Woodland habitat	Woodland areas (km^2) according to:				CL as % of CS2000	CL as % of FR	CL as % of LCM2000
	CS2000[(1)]	LCM2000[(2)]	FR[(3)]	CL[(4)]			
Coniferous	13,730	12,974	10,080	7,505	54.7%	74.5%	57.8%
Broadleaved	14,710	15,297	9,710	7,617	51.8%	78.4%	49.8%
Unmanaged	Not mapped	Not mapped	3,621	3,871	n/a	107%	n/a
All woodland	28,440	28,271	23,411	18,993	66.8%	81.1%	67.2%

(a) Results presented for GB only, as FR data not available for NI. See Hall *et al*. (2003a) for further details on mapping woodland habitats for the UK.
(1) From: Haines-Young *et al.* (2000) Table 2.1. Areas include managed and unmanaged woodland
(2) From: Areas derived from GB digital maps of appropriate LCM2000 classes. Areas include managed and unmanaged woodland.
(3) From: Areas derived from GB digital maps of data supplied by Forest Research. Coniferous and broadleaved woodland areas refer to managed woodland only.
(4) From: Areas derived from GB digital maps of habitats generated according to methods in Hall *et al*. (2003a, 2004a). Coniferous and broadleaved woodland areas refer to managed woodland areas only.

3.5.3 Co-location of broad habitat types

The broad habitats which have been mapped nationally for critical loads research are those sensitive to acidification and/or eutrophication. Areas of other habitats are not considered in this study (Section 3.5, Table 3.5.1).

The maps show the distributions of the various habitats across the UK, with different habitat types dominating in different regions. Calcareous grassland and managed broadleaved woodland generally dominate in the South and East of Britain, acid grassland, dwarf shrub heath and managed conifers dominate the uplands of Wales, northern England and southern Scotland, and dwarf shrub heath and montane dominate the upland regions of northern Scotland. Figure 3.5.1 maps the dominant sensitive habitat (that occupying the largest area) in each one km grid square of the UK. However, a number of different habitats may occur in the same one km grid square. This section describes the co-location of habitats being considered. Because distributions are bounded (0 to 100 per cent), standard regression is not appropriate as a measure of association. For example, although heaths and bogs are associated, once the proportion of heaths is greater than 50 per cent, the area of bog must be less than 50 per cent.

The percentage of one km grid squares across the UK containing one or more of the nine sensitive terrestrial habitats is shown in Table 3.5.5 below. The majority (69 per cent) of one km grid squares contain one or two sensitive habitats, and almost 22 per cent contain three habitats. The maximum number of sensitive habitats found to occur in the same one km grid square is seven, but this only occurs in three squares, representing 0.001% of all squares (206,732) containing any sensitive habitat.

Table 3.5.5: The percentage of one km grid squares in the UK containing one or more habitats sensitive to acidification and/or eutrophication.

Number of habitats	Number of one km squares	Percentage of one km squares
1	70,332	34.0%
2	71,451	34.6%
3	44,666	21.6%
4	16,354	7.9%
5	3,737	1.8%
6	189	0.09%
7	3	0.001%

There were 225 possible combinations of the nine sensitive habitat types occurring in any one km grid square of the UK. The occurrence of each individual habitat (area and spread across the UK) and its associations with other habitats have been examined and the results presented in Table 3.5.6. Note the following definitions that must be considered in interpreting the results:

- Total habitat area = sum of all sensitive habitats mapped for critical loads and as listed in Table 3.5.7.
- Percentage of UK squares = % of UK one km squares that contain sensitive habitat(s). Total = 206,732 x 1 km^2.
- Percentage of habitat one km squares = % of all one km squares that contain the named habitat, either singly or in combination with other habitats.

Dwarf shrub heath and acid grassland habitats

- Occupy largest areas of all habitats considered (34% and 20.6% respectively).
- Have largest spread across the country occurring in more than 40% (44.8% for dwarf shrub heath, 43.2% for acid grassland) of UK one km squares.
- Relatively low frequency (~10% of habitat squares) of habitats occurring alone.
- Most frequently occur in combination with other habitats (51-56% of possible habitat combinations), mainly with each other and with bog and managed coniferous woodland.

Managed coniferous woodland

- Occupies 11.1% of total habitat area and occurs in 22.6% of UK one km squares.
- Rarely occurs alone (5.5% of habitat squares).
- Found in 45.3% of possible habitat combinations and mainly with acid grass, dwarf shrub heath and managed broadleaved woodland.

Managed broadleaved woodland

- Similar in area to managed coniferous woodland.
- Widespread distribution, occurring in 41% of UK one km squares.
- Found both alone (24.3% of habitat squares) and in 45.8% of the possible habitat combinations, mainly with calcareous grassland and unmanaged woodland.

Bog

- Occupies 7.3% of total habitat area and occurs in just 10.1% of UK one km squares.
- Relatively low frequency (10.2% of habitat squares) of occurring alone.
- Found in 36.9% of the possible habitat combinations, mainly with acid grassland, dwarf shrub heath and managed coniferous woodland.

Unmanaged woodland

- Occupies 5.4% of total habitat area but occurs in 20% of UK one km squares.
- Rarely found alone (7.2% of habitat squares).
- Found in 45.3% of possible habitat combinations, mainly with managed broadleaved woodland, calcareous grassland and managed coniferous woodland.

Calcareous grassland

- Occupies less than 5% of total habitat area but occurs in 18.8% of UK one km squares.
- Found equally alone (38.7% of habitat squares) and in combination with other habitats (37.3% of possible combinations), mainly with managed broadleaved woodland, unmanaged woodland and managed coniferous woodland.
- The high occurrence of this habitat alone may also be due to its association with other habitat types (such as improved grassland) that are not mapped for critical loads.

Montane

- Occupies 4.1% of the total habitat area and only occurs in 2.8% of UK one km squares.
- The scarcity of this habitat means it is found in relatively few habitat combinations (14.7%) and occurs mainly with other upland habitat types, such as acid grassland and dwarf shrub heath.

<u>Supralittoral sediment</u>

- Occupies the smallest area at just 2.8% of the total habitat area and is found in only 5.8% of UK one km squares.
- Found mainly alone (52.5% of habitat squares), though this may be partly due to its coastal nature and association with other habitat types not mapped for critical loads.
- Occurs in 31.6% of possible habitat combinations, mainly with acid grassland and dwarf shrub heath.

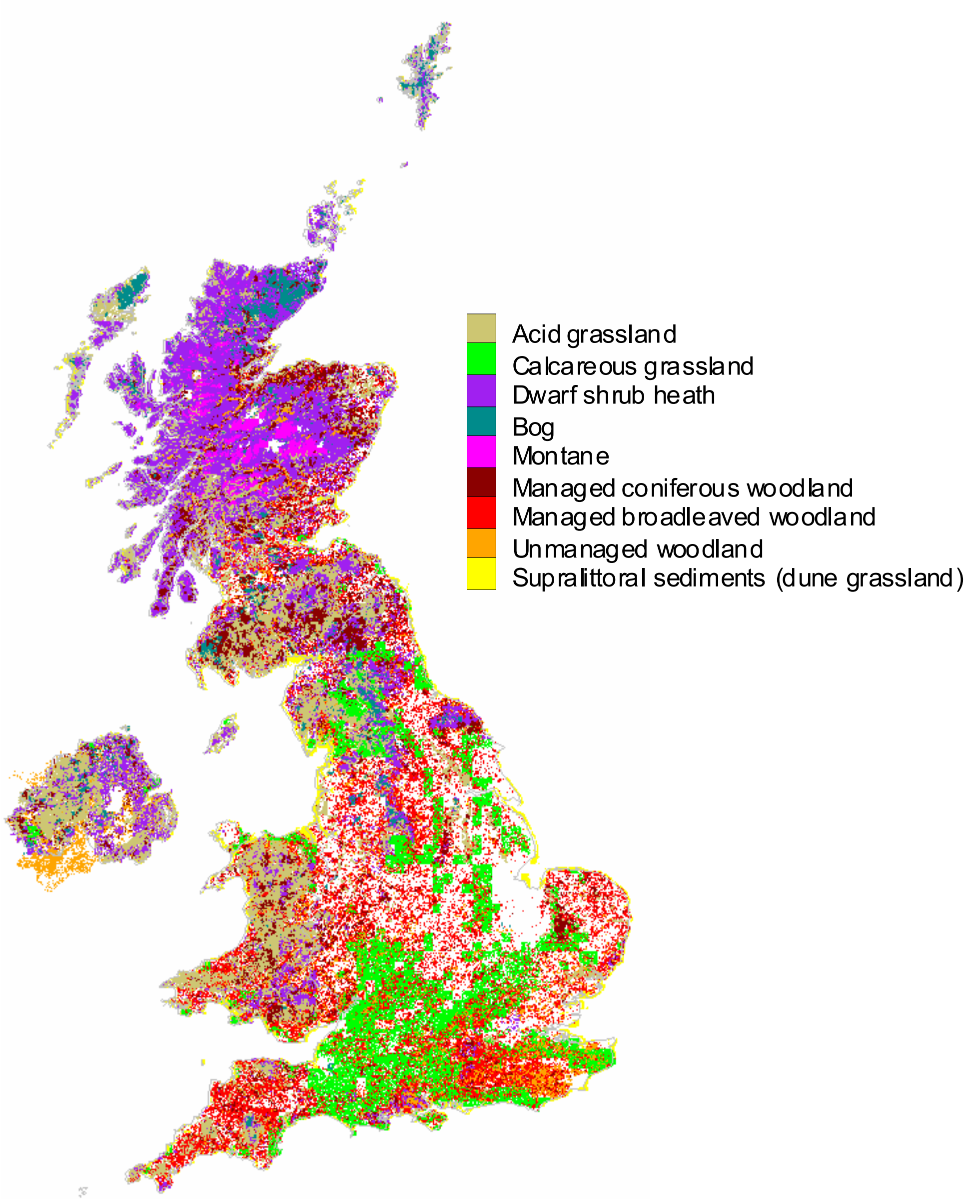

Figure 3.5.1: Map showing the dominant broad habitat in each one km grid square excluding habitat areas less than 10 ha (10% of a one km grid square). Individual maps used for critical loads purposes include all habitat areas.

Table 3.5.6: Statistics at the national and habitat-specific levels.

Broad Habitat	National statistics[1]				Habitat statistics[2]		
	Habitat area (km^2)	% of total habitat area	Percentage of UK one km squares that contain broad habitat[3]	Number & percentage of habitat combinations that contain broad habitat	Percentage of one km squares occupied by broad habitat alone	Top three habitat combinations and percentage of squares occupied by habitat combination: Habitat combination	% of habitat one km squares
Dwarf shrub heath (DSH)	25,748	34.0%	44.8%	126/225 = 56.0%	11.2%	DSH/AG DSH/AG/MCW DSH/AG/BOG	25.4% 8.1% 7.4%
Acid grassland (AG)	15,621	20.6%	43.2%	115/225 = 51.1%	10.3%	AG/DSH AG/DSH/MCW AG/DSH/BOG	26.3% 8.4% 7.6%
Managed conifer (MCW)	8,437	11.1%	22.6%	102/225 = 45.3%	5.5%	MCW/AG/DSH MCW/MBL MCW/MBL/AG/DSH	16.0% 9.3% 8.4%
Managed broadleaf (MBL)	7,515	9.9%	41.0%	103/225 = 45.8%	24.3%	MBL/CG MBL/UMW MBL/UMW/CG	10.5% 9.4% 5.7%
Bog (BOG)	5,541	7.3%	10.1%	83/225 = 36.9%	10.2%	BOG/AG/DSH BOG/DSH BOG/AG/DSH/MCW	32.6% 17.6% 8.3%
Unmanaged wood (UMW)	4,094	5.4%	20.0%	102/225 = 45.3%	7.2%	UMW/MBL UMW/MBL/CG UMW/MBL/MCW	19.2% 11.7% 6.7%
Calcareous grassland (CG)	3,577	4.7%	18.8%	84/225 = 37.3%	38.7%	CG/MBL CG/MBL/UMW CG/MBL/UMW/MCW	22.9% 12.4% 4.0%
Montane (MON)	3,129	4.1%	2.8%	33/225 = 14.7%	19.6%	MON/AG/DSH MON/DSH MON/AG	42.5% 21.8% 3.9%
Supralittoral sediment (SED)	2,128	2.8%	5.8%	71/225 = 31.6%	52.5%	SED/AG/DSH SED/DSH SED/AG	10.1% 9.9% 7.5%

[1] Statistics based on all one km grid squares in the UK (mapped for critical loads) containing any sensitive broad habitat.
[2] Statistics based on one km squares containing named habitat listed in left-hand column only.
[3] Either singly or in combination with other broad habitats.

3.5.4 Acidity critical load values by broad habitat

Acidity critical loads for non-woodland terrestrial habitats are set according to the weathering characteristics of the soil (Section 3.3). This information on weathering rates is also used in the SMB equations used to derive acidity critical loads for woodland habitats. As different habitats are likely to thrive in different soil conditions and therefore occur on different soil types, one may expect to observe a difference in the critical load values by habitat type. However, it must be remembered that critical load values are based on the dominant soil type in each one km grid square, regardless of habitat type (Section 3.3). Table 3.5.7 shows the minimum, maximum and mean acidity critical load values derived from all one km squares mapped for each habitat.

Table 3.5.7: Acidity critical load values by broad habitat. Results ordered by descending mean value.

Broad habitat	Acidity critical loads (keq ha^{-1} yr^{-1})			
	Minimum	**Maximum**	**Mean**	**SD**
Calcareous grassland	4.0	4.0	4.0	0
Unmanaged woodland	0.07	12.7	3.02	3.08
Managed broadleaved woodland	0.1	13.0	2.79	3.23
Managed coniferous woodland	0.1	13.4	1.94	2.35
Dwarf shrub heath	0.05	4.0	0.65	0.69
Acid grassland	0.05	4.0	0.62	0.71
Bog	0.05	4.0	0.53	0.54
Montane	0.1	4.0	0.38	0.22

These results show the highest critical loads for calcareous grassland; not surprising, given that the soils this habitat occurs on will have a high buffering capacity to offset incoming acid deposition. Hence, high critical load values were set for these soils (Hall *et al.*, 2003a, 2004a). The woodland habitats have a greater range of critical load values, and higher mean values than the remaining non-woodland habitats. This is because the SMB equation includes the critical leaching of acid neutralising capacity (ANC). In deriving empirical critical loads for soils (applied to non-woodland habitats), this term is effectively zero, resulting in lower critical loads. The remaining habitats are largely associated with upland areas and all have values in the same general range. The mean values for dwarf shrub heath and acid grassland are very similar, also reflecting their similar spatial distributions (Section 3.5.3). The mean value for the bog habitat falls between that of acid grassland and montane, and the latter has the lowest mean value, reflecting the thin, base poor soils found in most montane regions.

3.6 Conclusions (national scale uncertainty)

3.6.1 Critical loads in managed coniferous woodland

- The coefficients of variation of critical load parameters are smaller than expected (smaller than many of the input uncertainty ranges), due to a compensation of errors mechanism (Skeffington *et al*., 2006). However, since this mechanism is related to the correlation between parameters, and there is some uncertainty in the value of the correlation coefficients used to connect the various parameters, this area requires further study.
- For policy development, the uncertainty in critical load exceedance is an important parameter. This study enabled a probability of exceedance to be calculated for managed coniferous woodland in the UK, showing that 35 per cent of the habitat has a greater than 95 per cent probability of exceedance for acidity and 98 per cent for nutrient nitrogen.
- The analysis presented in Section 3.1 only applies to critical loads for managed coniferous woodland. Uncertainties for other habitat types, even managed broadleaved woodland whose critical loads are also calculated by the mass balance approach, would have to be re-calculated. This is because the probability of exceedance is a function of how far the deposition is from the critical load function, which will vary both spatially and between habitats.
- A sensitivity analysis showed that where the managed conifer acidity critical load is small (less than 0.2 keq ha^{-1} yr^{-1}), Ca deposition is more important than Ca weathering.

3.6.2 Variance in acidity critical loads by soil type

- Acidity critical loads for England and Wales based on the dominant soil in each 1x1 km grid square are similar to the area-weighted critical load values based on all soil types within each 1x1 km square. The number of 1x1 km squares with critical loads above and below 1.0 keq ha^{-1} yr^{-1} are similar for both maps.
- Acidity critical loads based on the most sensitive soil type have values of ≤ 1.0 keq ha^{-1} yr^{-1} in approximately 70 per cent of the 1x1 km squares in England and Wales. Conversely, about 70 per cent of the squares have critical loads above 1.0 keq ha^{-1} yr^{-1}, when based on the least sensitive soil in each grid square.
- The variance in acidity critical loads by soil type differs by region. For example, in Snowdonia most soil types tend to be very sensitive to acidification and have low critical loads compared to the Brecklands, where the soil types cover a broad range of sensitivities and critical loads.
- The acidity critical load is the same for all soil types within 42 per cent of the 1x1 km squares in England and Wales, and only different for all soil types within four per cent of the squares. The acidity critical load for the dominant soil is the same as that for the most sensitive soil type in 26 per cent of 1x1 km squares in England and Wales.
- The acidity critical load is smaller for the sub-dominant soil in 31 per cent of 1x1 km squares in England and Wales.

3.6.3 Uncertainties in applying national critical loads to designated sites

- The acidity critical loads based on different soils (for the site centroid, or area-weighted for all site squares, or area-weighted by soil and habitat) were similar for 14 out of 17 SACs examined.
- Anomalies in setting appropriate critical load values for habitats within SACs arise for sites that are large with a variety of soil and vegetation types, or where small unusual habitats occur within other habitats, for example, a small area of bog within an area of limestone.

3.6.4 Uncertainties in mapping habitats

- The uncertainties in mapping Biodiversity Action Plan broad habitats for critical loads research can be identified, but not quantified.
- There are uncertainties inherent in the individual data sets used to generate the habitat maps. The data come from many sources, and rarely with any information on their associated uncertainties.
- The data sets used were generated at different times, for different purposes (such as land cover vs land use) and are of differing spatial resolutions.
- Additional uncertainties in the habitat areas arise from the manipulation and combination of the different data sets.
- More than one broad habitat can be mapped in a one km grid square, but the acidity critical loads for all habitats occurring in a square are all based on the dominant soil type, which may or may not be appropriate for the habitat in question. More information is required to link the habitat types to appropriate soil types.
- The mean acidity critical load values by broad habitat show the highest values for the managed woodland habitats, because of the inclusion of the leaching of acid neutralising capacity in the calculations, which is effectively set to zero for the non-woodland habitats.
- Despite the dependence of acidity critical loads on the dominant soil, the mean critical loads nationally appear to be appropriate, with higher critical loads values for calcareous grassland - a habitat found on soil types with a high buffering capacity - and the lowest mean critical loads for the montane habitat occurring on thin base-poor soils.
- The maps provide a national picture of the distributions of broad habitats on a one km grid, suitable for critical loads mapping purposes at the national scale, where the underlying soils data are also at one km resolution. However, due to the uncertainties identified, it is inadvisable to attempt to interpret these maps at a local or site-specific scale.

4 Comparison of national and site-specific uncertainties

Summary

- This section compares the uncertainties based on site-specific and national data for three woodland sites.
- The suitability of criteria, models and data used in both site-specific and national scale assessments is discussed.
- Uncertainty ranges depend on whether site-specific or national data are used.
- With only three sites, it was not possible to establish a clear relationship between the site-specific and national estimates of uncertainty.
- Critical load models and criteria do not exist for all habitats and species.
- Default values derived from very limited measurements have been used for some parameters, due to a lack of site-specific data.

4.1 Input data to uncertainty analysis

In order to provide a measure of the appropriateness of applying the models at the site-specific scale, we incorporated estimates of uncertainty in both national and site-specific data into the calculation of critical load exceedance for individual sites. National scale data were designed for national and regional policy development and may not give accurate answers on individual sites. Use of national data to calculate exceedance may thus lead to vulnerable sites being unprotected, or alternatively to unnecessary expenditure on emission control. This section is an exploration of the extent to which using data measurements from research sites leads to different results from those achieved by using national data applied to the same sites. It uses uncertainty analysis to evaluate the accuracy and precision of exceedance estimates from the two types of data.

This study focused on three coniferous woodland sites across the UK: Liphook in the South East of England, Thetford in the East of England and Aber in Wales for which site-specific data were available. The choice of default values, ranges and distributions for each of these sites is shown in Appendix B.

The distributions and ranges of the input parameters to the Monte Carlo simulations (Appendix B) show large differences depending on whether national or site-specific data are used. High quality measurements are often available for individual sites, whilst at the national scale input data usually have to be derived from other sources, such as regional soil maps, statistics from experimental data sets or expert judgement. For example, national estimates of soil mineral weathering rate are based on the Skokloster classification (Nilsson and Grennfelt, 1998) using national soils databases, whereas site-specific values may be derived from geochemical measurements or modelling techniques applied to the soil present at the site (for example, Langan *et. al.,* 1996). In addition, national critical load calculations use single default values for nitrogen and base cation uptake by coniferous plantation forestry. The values are derived from measurements at a number of sites across the country and the mean value applied to all coniferous forestry, irrespective of tree species. For individual sites, information from harvested trees can be used (for example Reynolds *et. al.,* 1998) to give a more accurate estimate of nutrient removal. Local knowledge of soil may be used to give appropriate ranges for the gibbsite equilibrium constant at the site level, compared to national estimates that use default values based on generalised soil characteristics.

Liphook and Aber both have a large amount of site-specific data. The input parameters differ significantly for Liphook, but less so for Aber. Input values for the Thetford site differ only for weathering rate. The effects these differences have on exceedance are discussed in the next section.

Uncertainty ranges for all sites differ for most input parameters, depending on whether national or site-specific knowledge is used. It would be expected that the uncertainty ranges for input parameters for the site-specific analysis should be narrower than those for the national analysis, as we may have better knowledge based on detailed site measurements in space and time. In general, site-specific uncertainty ranges are narrower.

4.2 Results

Table 4.2.1 below summarises the results of the uncertainty analysis for the critical load exceedance calculations for each site, based on national (N) and site-specific (SS) data. These are compared with the deterministic exceedance values.

Table 4.2.1: Comparison of the deterministic critical load exceedance values and results of the uncertainty analysis for Liphook, Thetford and Aber

Site	Deterministic value	Uncertainty analysis					
		Mean	Standard deviation	Prob. of exceedance	Percentiles		
					5th	50th (median)	95th
Liphook N	2,095	2,177	811	100%	1,009	2,106	3,597
Liphook SS	-15	210	459	68%	-558	217	949
Thetford N	-7,323	-7,089	3,656	0.3%	-14,019	-6,748	-1,913
Thetford SS	-8,061	-8,408	4,138	2%	-16,210	-8,023	-2,662
Aber N	935	1,052	748	93%	-105	1,006	2,354
Aber SS	1,335	1,295	618	96%	213	1,396	2,237

(i) Values are in eq ha^{-1} yr^{-1}.
(ii) N = based on national data; SS = based on site-specific data

4.2.1 Deterministic results

Table 4.2.1 shows the deterministic results of critical load exceedance using national and site-specific input data. For Liphook there is a large difference between predictions, because the measured deposition is much lower for all acidifying species than the national data suggest. For Thetford, there are no site-specific deposition data, and the exceedance differs only slightly because of a small difference in weathering rate. For Aber, measured S deposition is much higher than the national estimate, run-off is considerably less and weathering is somewhat less (Appendix B). Exceedance is therefore greater with the site-specific parameters for this site.

The deterministic results for the Liphook site show that different conclusions would be reached about critical load exceedance, depending on whether national or site-specific data were being used. The national data predict that the site is highly exceeded and the site-specific data predict no exceedance. The Aber analyses also gave quite different deterministic results, although both predicted the site was exceeded regardless of whether site-specific or national data were being used. The results of the Thetford analyses showed good agreement, predicting that the site was not exceeded irrespective of the type of data. This is because the input data sets differ only in weathering rate.

4.2.2 Monte Carlo results

Table 4.2.1 also shows the means, standard deviations, and 5th, 50th and 95th percentiles of the exceedance distributions. For Liphook, the mean for the site-specific analysis (210 eq ha^{-1} yr^{-1}) lies outside the 95 per cent confidence intervals for the national analysis, indicating that the analyses predict very different results. For Thetford and Aber, the mean values for the site-specific analysis (-8,408 eq ha^{-1} yr^{-1} and 1,295 eq ha^{-1} yr^{-1} respectively) are in good agreement with the national analysis (-7,089 eq ha^{-1} yr^{-1} and 1,052 eq ha^{-1} yr^{-1} respectively) and both sets of analyses lie well within one standard deviation of each other. The site-specific analysis for Thetford predicts a slightly higher uncertainty than the national analysis, which is probably due to the higher weathering rate used in the site-specific analysis. In general, the means of the Monte Carlo analysis correspond with the deterministic results apart from the Liphook site-specific result, for which the deterministic value predicts non-exceedance and the mean of the Monte Carlo analysis suggests that the site is exceeded.

The probability distributions of the predicted exceedance at each site were used to calculate the probability that the rate of deposition was lower than the critical load (probability of exceedance statistics in Table 4.2.1). The results of the site-specific and national analyses were very different for the Liphook site, where the national analysis predicted a 100 per cent probability of exceedance; that is, certain risk of acidification. However, the site-specific analysis predicted only a 68 per cent probability that the site is exceeded, indicating less confidence in the exceedance prediction. The national analysis for Aber showed a high level of confidence that the site is exceeded and the site-specific study also predicted a very high level of confidence. For Thetford, the probability of exceedance is less than five per cent for both sets of analyses, suggesting with a high degree of confidence that the critical load is not exceeded at this site.

At Liphook, the national 5th percentile exceedance value is above zero, suggesting it is likely that the site is exceeded. The site-specific study shows the 5th percentile is below zero (and the 50th percentile above zero), indicating greater uncertainty about exceedance. Adopting a highly precautionary approach based on whether the 95th percentile of the distribution is greater than zero results in both analyses concluding that the site is exceeded. For Aber, the national analysis predicts a 50th percentile value above zero and a 5th percentile below zero, so that a policy maker may once again opt to consider that the critical load is exceeded or that further study is required. However, for the site-specific analysis the 5th percentile is above zero, indicating that the site is exceeded. For Thetford, the 95th percentile values are below zero for both analyses, indicating that it is likely that the site is not exceeded.

At Liphook, the 95th percentile of the site-specific analysis (949 eq ha^{-1} yr^{-1}) is smaller than the 5th percentile of the national analysis (1,009 eq ha^{-1} yr^{-1}), so the two distributions hardly overlap, with the national analysis predicting significantly higher exceedance than the site-specific analysis. This demonstrates how differences in the input data manifest themselves in the variability of the exceedance predictions. For Thetford and Aber, the 5th, 50th and 95th percentiles are similar for both distributions, indicating significant overlap between probability distributions, although for Thetford the national data predict higher exceedances whereas for Aber, it is the site-specific data which does so. The distributions for Thetford show that even a site with a large deterministic negative exceedance can have a small tail of exceeded values. No general relationship between national and site-specific critical load exceedance uncertainty probability distributions can be established from the analysis of these three sites.

4.3 Suitability of criteria, models and data

National critical load models are based on steady state assumptions, meaning they set critical loads to protect systems that are in long-term steady state conditions. As such, it may not be possible to validate critical loads in the field, or to find evidence of harmful effects (for example, in areas where critical loads are predicted to be exceeded), unless the system is in steady state. National maps are based on (a) national scale database, and (b) default values for some parameters. The issue of data limitations and resolution is also addressed below (Section 4.3.1).

The critical load models are based on a limited number of chemical criteria, for example the critical calcium to aluminium ratio in soil solution. These criteria may be set to protect specific habitats or species and may not be appropriate for the protection of other habitats or species for which sites are designated. Indeed, criteria and models may not exist to calculate critical loads to protect all features (habitats/species) for designated sites. In addition, the criteria may have been derived from non-UK data, for example the acid neutralising capacity (ANC) threshold used for UK surface waters is based on Norwegian data. Ideally, criteria should be based on UK studies. The following points deal with the different criteria used in the UK:

Weathering rates

Soil associations may contain several soil series and each may have different physical properties. Acidity critical loads for non-peat soils are based on the properties of the dominant series within each soil association, and the critical loads are mapped according to the dominant soil association in each 1x1 km grid square (Loveland, 1991; Hornung *et al.*, 1995). These critical loads are applied to non-woodland terrestrial habitats in the UK and the critical load is set to protect the soil upon which the habitat occurs, that is, it is not habitat-specific.

For the national maps, the assignment of soil types to critical loads class (five class ranges) was performed by the soil survey organisations for England and Wales, Scotland and Northern Ireland. There are unquantified uncertainties in the assignment of critical loads class to the soil types.

A number of methods can be used to determine weathering rates. There are uncertainties associated with the different methods and they can give different results. The weathering rate is not an easy parameter to measure and hence data are rarely available at site-level.

Critical pH

The criteria recommended (Calver *et al.*, 2004) and applied in the calculation of acidity critical loads for peat soils in the UK is a critical hydrogen ion concentration (equivalent to pH 4.4) in soil solution. The critical value of pH 4.4 is precautionary and based on studies of the effects of acidification on *Calluna vulgaris* and on the abundance of bryophytes on peat soils. This value was considered inappropriate for lowland/arable fen peats that are less sensitive to acidification and critical loads for these areas were instead set to 4.0 keq ha^{-1} yr^{-1}. Different critical thresholds may also be needed for specific ecologically important biotic populations.

This method could be applied at an individual site level if information on run-off (precipitation surplus) was also known.

Ratio of calcium to aluminium in soil solution

Molar ratios of base cations or calcium to aluminium in soil solution, set to protect the fine roots of trees, are commonly used criteria within the acidity SMB equation (UBA, 2004). In the UK, a calcium to aluminium ratio equal to one is used in the calculation of acidity critical loads for coniferous and broadleaved woodland habitats (managed and unmanaged). Different values may be more appropriate for individual species or woodland types, but we lack good UK data upon which to base alternative values.

The SMB equation can be applied at the site level, but lots of measurements are required (such as weathering rates, calcium deposition, calcium uptake, run-off) for the input data. National scale data could be used, but this would require an assessment of the site characteristics to determine how applicable national map data were for the site in question.

Acid neutralising capacity (ANC)

The ANC criterion used in the UK is set to protect the populations of brown trout and this may not be appropriate for the protection of other species. This criterion is applied within the Steady State Water Chemistry (SSWC, Henriksen) and First-Order Acidity Balance (FAB) models. Both models require surface water chemistry data and FAB additionally requires catchment-specific input data. This means that the national maps are based on site-specific data (for 1,722 sites). Whilst this is a strength of the approach, critical load values cannot simply be extrapolated and applied to other sites.

Acceptable nitrogen leaching

The mass balance equation for the calculation of nutrient nitrogen critical loads for managed woodlands uses an acceptable nitrogen leaching value as the criterion. At present, in the UK a single value is applied to all managed coniferous woodland, and another value to all managed broadleaved woodland, both based on data published in the literature (Emmett *et al.*, 1993; Emmett & Reynolds, 1996). Other values may be more appropriate for different woodland types in different locations and conditions (such as soil type).

Changes in ecosystem structure/function

The ranges of empirical critical loads of nutrient nitrogen are based on observed changes in the structure or function of ecosystems as determined from experimental data, field observations and/or dynamic ecosystem models. The ranges were the result of a UNECE workshop attended by experts in the nitrogen field (Achermann and Bobbink, 2002). To allow for the application of these critical loads across Europe, ranges were set for habitat classes of the European Nature Information System (EUNIS); therefore, the relationship between these classes and UK habitat types is needed to apply the method in the UK (Hall *et al.*, 2003a; NBN Habitats Dictionary). Critical load values will not necessarily protect all habitats/species within each EUNIS class, as the values are generally based on data for specific habitats or species and data may not be available for all other habitats/species, which may also respond differently to nitrogen loads.

4.3.1 Data limitation/resolution issues

There are two main data issues: resolution and paucity of data. Both contribute to uncertainties in the calculation of critical loads.

Data resolution

Many of the data sets used in the national calculations and mapping of critical loads are at 1x1 km and 5x5 km square resolution. As has previously been mentioned, the soils data are based on the dominant soil association in each 1x1 km grid square and other sub-dominant soils may have different properties. These soils data are used to:

- Determine the dominant soil type (association or map unit) in each 1x1 km square and the selection of the appropriate critical loads calculation method: mineral, organo-mineral soils or peat soils.
- Define the calcium weathering rate inputs to the SMB equation; values are based on a proportion (by soil association) of the base cation weathering rate.
- Define areas of “calcium-rich” and “calcium-poor” soils for the application of different calcium uptake rates in the SMB equation for broadleaved woodland.
- Define the nitrogen immobilisation and denitrification values (set by soil association) required for the calculation of nitrogen critical loads (nutrient nitrogen for managed woodlands, minimum critical load of nitrogen)

The SSMB equation and the calculation of acidity critical loads for peat soils both require long-term (30 year mean) run-off data; 1x1 km square data based on 30-year rainfall data sets are used. The SSMB equation also requires total (marine plus non-marine) calcium deposition data; the national data are on a 5x5 km grid and include separate values assuming acid grassland or woodland cover in each square. However, deposition values may vary across the 5x5 km grid and there are uncertainties in the measurements and modelling of calcium deposition (see Section 2.1.6: Draajers *et al.,* 1997).

Paucity of data

Some input values to the critical load calculations are based on only a few measurements, due to a lack of data nationally. These include:

- Uptake values, such as the removal of base cations, calcium and nitrogen due to harvesting of trees. Data are currently based on UK sites from the ICP Level II Monitoring Survey (http://www.icp-forests.org) number for N, and number for BC and Ca which also have values for Ca-rich and Ca-poor sites.
- Uptake values (base cations) for other habitat types: values are generally taken from the literature and may be based on a single site or few sites in the UK.
- Values for removal of nitrogen by fire (applicable to heathland habitats) are also based on data for a single site.
- Acceptable nitrogen leaching for woodland habitats: values based on literature.

For all of these parameters, the values required may vary by site due to local conditions (such as soils, climate), but the impacts of any uncertainties introduced by using such default values from the literature or survey have not been quantified. However, some assumptions about the data distributions can be made, as used in the national uncertainty analysis (Section 3).

4.4 Conclusions

The analysis, based on example sites, illustrates how the application of national data on a site-specific scale should be treated with care, and shows that:

- Input default values and uncertainty ranges for a research site may not correspond to data derived for national purposes.
- The exceedance probability distributions based on site-specific and national data can vary significantly from one site to another. Liphook varied significantly, although Aber gave good agreement. Thetford also gave good agreement, although more estimates of site-specific data are required to make better judgements for this site.
- The user needs to be aware of the potential for error in both approaches, especially where national data have to be used because the site-specific data do not exist or are too costly to collect.

Users of national critical loads data and maps need to be aware of the following limitations:

- The critical chemical criteria on which the critical load calculations are based are only relevant for specific habitats or species.
- Models and/or criteria may not exist for the habitats/species of interest and it may be inappropriate to use data for other similar habitats or species.
- The resolution of data used to derive national maps and the paucity of data on which some model parameters are based also have limitations.

5 Uncertainties at the regional scale

Summary

- The TRACK model was used to generate predictions of S and N deposition, and these predictions were compared with measured rates derived from wet deposition and air concentration measurements. Agreement was reasonable, but model estimates could be improved by a bias adjustment derived from regression analysis.
- The regression analysis provides a measure of the uncertainty in the modelled deposition rates.
- Monte Carlo analysis of input parameter uncertainty showed it contributed a substantial amount to the uncertainty in deposition model predictions. However, input parameter uncertainty does not explain all the model errors.
- Uncertainty in critical load exceedance was generated for each of the 2,588 one km squares in South East England containing coniferous forest, using probability distributions of critical loads already generated in this study, and a Monte Carlo simulation of TRACK model results.
- Illustrative maps are provided of various metrics of critical load exceedance. Using the median exceedance increases the number of squares exceeded compared to a deterministic estimate, and using the 95th percentile increases it substantially.
- The results depended strongly on which site was used to generate the critical load probability distributions – Liphook, Aber or Thetford.
- Using a bivariate normal distribution for S and N critical loads and a bivariate Student's *t* distribution for deposition gave similar results to direct sampling of the distributions and is therefore recommended for assessment purposes.
- However, in view of the sensitivity of the results to the probability distribution of critical loads used, it is recommended that these functions should be assigned to receptors on a case-by-case basis.

5.1 Introduction

In the calculation of exceedance both the deposition and the critical load estimates are uncertain, because they depend on uncertain estimates of model input parameters. Furthermore, the models used to calculate deposition are also inherently uncertain because of the assumptions made in their derivation and implementation.

The approach taken here was to use Monte Carlo simulations to develop probability distributions of critical loads and deposition rates, taking account of assumed probability distributions for the model inputs. In addition, predictions of sulphur and nitrogen deposition are compared with available measurements in order to estimate the inherent uncertainty in the deposition model. Maps were then prepared based on the following approaches:

- deterministic calculation of the difference between the deposition prediction and the critical load estimates;

- deterministic calculation of the difference between the deposition prediction, adjusted for bias by regression analysis and the critical load estimates;
- calculation of the probability distributions of critical load exceedance by resampling from the joint probability distribution of acid deposition predictions (sulphur and nitrogen) and the joint probability distribution of critical loads ($CL_{max}S$, $CL_{min}N$).
- fitting of multivariate normal distribution functions to acid deposition model outputs and critical load estimates and resampling from the fitted distributions.

The spatial domain of the maps was limited to the South East of England, in order to limit the amount of data generated and to facilitate data handling.

The probability distributions for the critical loads used in this assessment were developed in Subtask 1.3 (Section 3). The methods used are described in Sections 2 and 3 and are not repeated here.

5.2 Deposition modelling

5.2.1 Model

Netcen's long range acid deposition model, TRACK Version 1.7e, was used to predict annual wet and dry deposition of sulphur, oxidised and reduced nitrogen at wet deposition monitoring sites throughout the UK, and at 100, 20 x 20 km square areas covering the South East of England.

The technical specification for the TRACK model is described in a refereed paper by Lee *et al.* (2000). More recently, the predictions of deposition rates and ambient concentrations made by Version 1.7 have been compared with measured values by Abbott *et al.* (2003).

Version 1.7e is set up to facilitate Monte Carlo simulation, with input values selected from feasible ranges. Version 1.7e differs slightly from Version 1.7 used in our earlier work: it includes the wind direction sector averaging algorithm described in Appendix 8 of the annual audit report prepared for the Environment Agency.

Input data for the modelling runs will be found in Appendix C. Table C1 summarises the input parameters used in the model runs. The baseline model run was based on these parameter values. Table C2 summarises the range of input values used for Monte Carlo simulation. Values of each parameter were taken at random from the range, assuming a uniform distribution. Three hundred model runs were carried out. The 2002 50x50 km square EMEP emissions inventory for sulphur dioxide, oxides of nitrogen and ammonia provided the emissions inputs for most of the model domain. UK emissions estimates, taken from the 2003 UK National Atmospheric Emissions Inventory (NAEI), were aggregated onto a 10 km x 10 km grid for the UK. Local deposition of ammonia was calculated from the 2003 NAEI 1 km x 1 km inventory.

Monitoring data used in the assessment were taken from *Management and Operation of the UK Acid Deposition Monitoring Network: Data Summary for 2003*. The data are summarised in Table C3. The data were supplemented by additional data for ammonia and ammonium for 2002 from the UK National Ammonia Monitoring Network for sites included in Table C3.

5.2.2 Comparison of modelled and measured deposition rates

5.2.2.1 Wet deposition of sulphur

The annual wet deposition of sulphur was calculated from the monitoring data as the product of the precipitation-weighted mean sulphur concentration and the annual precipitation.

A background contribution to wet sulphur deposition was added to the modelled wet deposition. The background contribution was calculated as the collected rainfall multiplied by 10 μeq l^{-1}, based on the analysis of data from remote sites carried out by Irwin *et al.(1997)*.

Figure 5.2.1 shows the measured wet deposition plotted against the modelled wet deposition.

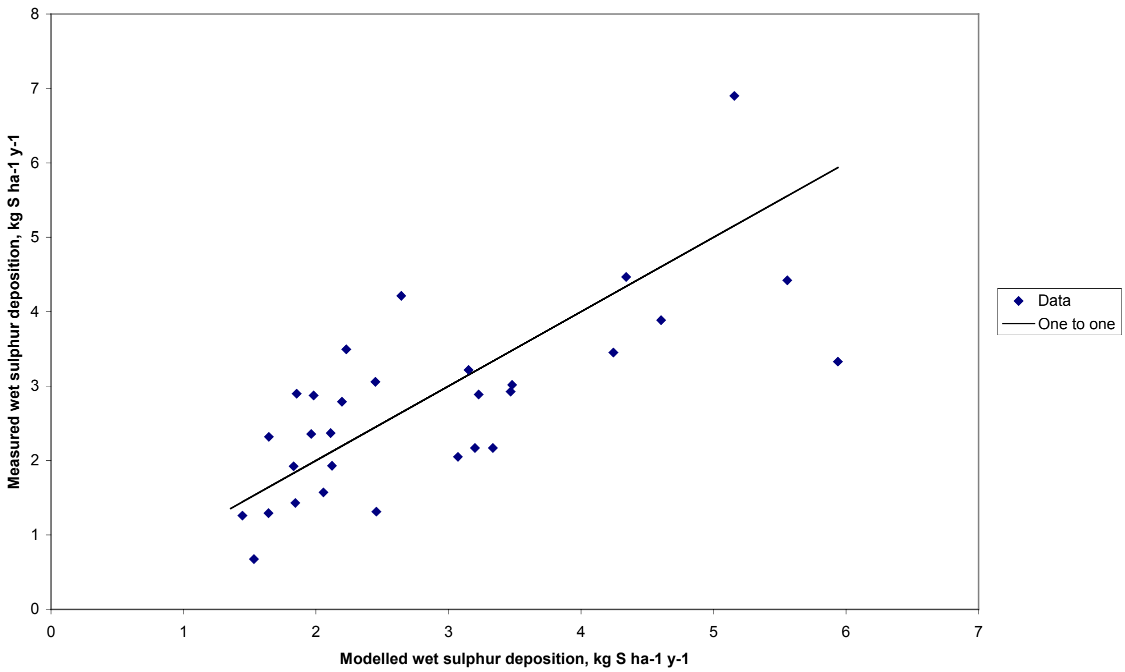

Figure 5.2.1: Comparison of measured and modelled wet sulphur deposition

5.2.2.2 Total deposition of sulphur

The total deposition of sulphur is made up of both wet and dry deposition. Dry deposition of sulphur dioxide is measured continuously at only two sites in the UK: Auchenforth Moss and Sutton Bonington, but wet deposition measurements are not made at these sites, so an estimate of total deposition is not possible. Dry deposition rates for sulphate particulates and sulphuric acid are not routinely made. The rate of dry deposition can only be inferred based on measurements of sulphur dioxide and sulphate concentrations and an assumed dry deposition velocity. A dry deposition velocity of 13.6 mm s^{-1} for sulphur dioxide was used in the assessment, corresponding to the upper bound of measurements made at Sutton Bonington. A dry deposition velocity of 0.5 mm s^{-1} was used for sulphate, corresponding to particulate deposition.

Model predictions of sulphur dioxide and sulphate concentrations are reviewed elsewhere in this project, in the annual audit report prepared for the Environment Agency. Figure 5.2.1 shows the total sulphur deposition calculated from the measurements plotted against the modelled deposition and the modelled one-to-one line. A least squares analysis was performed on the data, assuming that the error was proportional to the modelled deposition. The slope of the least squares line through the origin was 0.72. Figure 5.2.2 shows the least squares line and the upper and lower bounds of the 90th percentile prediction interval (the 5th and 95th percentiles). The prediction interval is the interval that will contain the measured values with the specified degree of confidence. The prediction interval was calculated assuming Student's *t* distribution with five degrees of freedom. This distribution was used because of the limited number of data points. It approximates to a normal distribution at over 30 degrees of freedom.

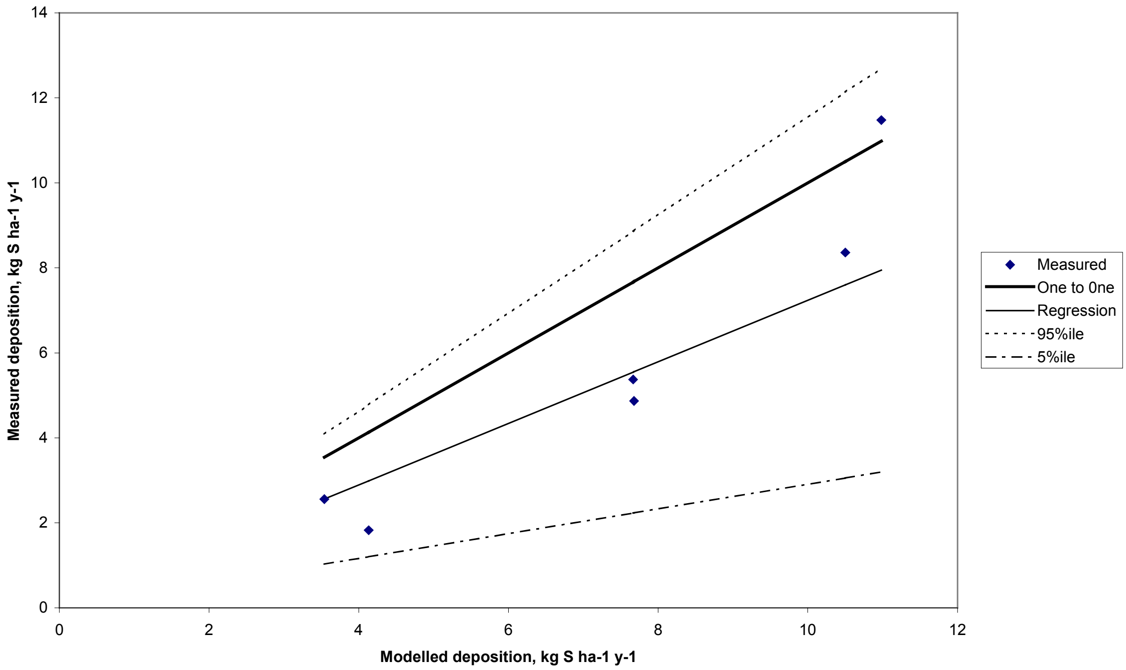

Figure 5.2.2: Comparison of modelled and measured total sulphur deposition

Table 5.2.1 shows the results of the Monte Carlo simulation. It shows the results of the baseline model run, the estimated sulphur deposition based on measurements, the 5th and 95th percentiles and mean of the modelled estimates from the Monte Carlo simulation. In several cases, the measurement-based estimate is outside the 5th and 95th percentile confidence limits. It may be concluded that there are sources of uncertainty in the model predictions other than input parameter uncertainty.

Table 5.2.1: **Sulphur deposition estimates from TRACK Monte Carlo simulation**

Site	Sulphur deposition estimates, kg S ha^{-1} y^{-1}				
	Baseline modelled	Measuremen t-based estimate	5th percentile of modelled estimates	95th percentile of modelled estimates	Mean of modelled estimates
Yarner Wood	7.7	4.9	7.1	8.2	7.8

Stoke Ferry	10.5	8.4	9.2	13.7	11.7
High Muffles	11.0	11.5	9.7	15.8	12.9
Lough Navar	3.5	2.6	3.3	3.6	3.4
Eskdalemuir	7.7	5.4	6.8	8.3	7.7
Strathvaich	4.1	1.8	3.4	4.3	3.9

5.2.2.3 Wet deposition of nitrogen

The annual wet deposition of nitrogen was calculated from the monitoring data as the product of the precipitation-weighted mean nitrogen concentration (oxidised plus reduced) and the annual precipitation. Figure 5.2.3 shows the measured wet deposition plotted against the modelled wet deposition.

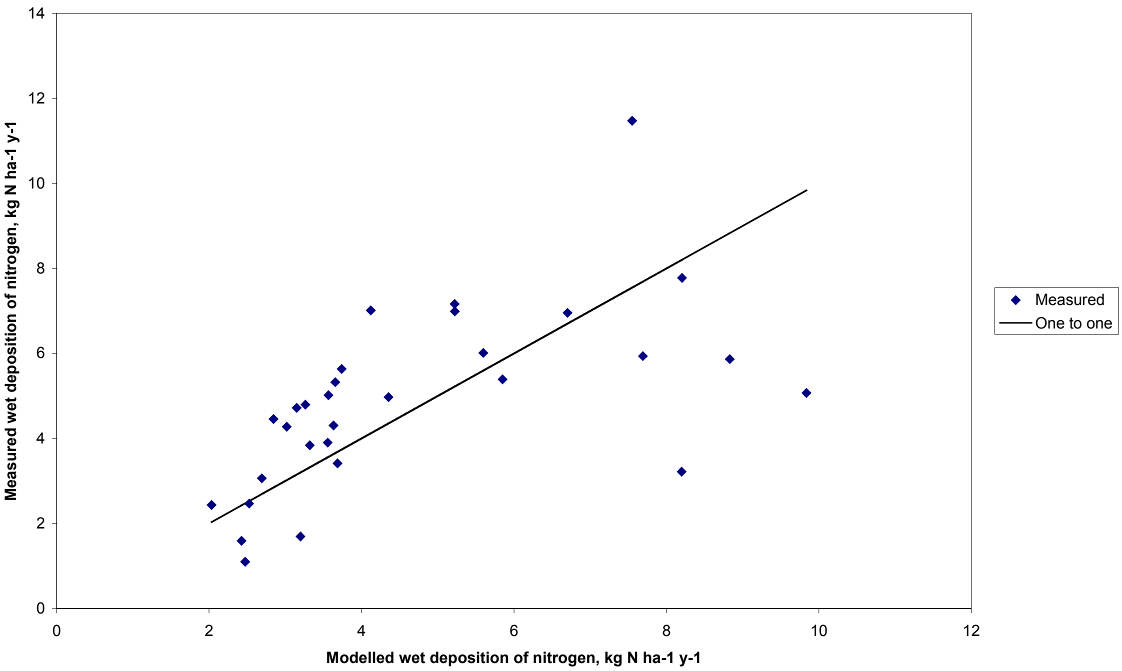

Figure 5.2.3: Comparison of modelled and measured wet deposition of nitrogen

5.2.2.4 Total deposition of nitrogen

The total deposition of nitrogen is made up of both wet and dry deposition of oxidised and reduced nitrogen species. Dry deposition of nitrogen species is not measured continuously in the UK. The rate of dry deposition can only be inferred based on measurements of nitrogen dioxide, particulate nitrate, nitric acid, ammonia and ammonium concentrations and assumed dry deposition velocities. The assumed dry deposition velocities were as follows:

Nitrogen dioxide	1.0 mm s^{-1}
Particulate nitrate	0.5 mm s^{-1}
Nitric acid	22.7 mm s^{-1}
Ammonia	17.2 mm s^{-1}
Ammonium particulate	0.5 mm s^{-1}

Most of the ammonia emitted in the UK is emitted near ground level from various agricultural sources. A considerable part of the ammonia emitted is rapidly deposited to the ground. The TRACK model is a long-range model that considers the net ammonia emission (total emission less locally deposited emission). The local deposition was estimated from the NAEI 1 km x 1 km ammonia emission inventory, assuming the ammonia is released 1 m above ground as:

$$D_l = \frac{(R_a - R_{a1})}{(R_{a1} + R_b + R_c)} E_{net}$$

where R_a is the total resistance in the atmospheric surface stress layer;
R_{a1} is the resistance in the lowest one metre of the surface stress layer;
R_b is the laminar sub layer resistance;
R_c is the surface resistance;
E_{net} is the net ammonia emission per unit area.

For most surface types, for neutral atmospheric conditions D is approximately 0.44 $E_{net.}$ The estimated local deposition was added to the long-range deposition modelled using TRACK.

Figure 5.2.4 shows the total nitrogen deposition calculated from the measurements plotted against the modelled deposition. Figure 5.2.4 also shows the modelled one-to-one line. A least squares analysis was performed on the data assuming that the error was proportional to the modelled deposition. The slope of the least squares line through the origin was 1.05. Figure 5.2.4 shows the least squares line and the upper and lower bounds of the 90th percentile prediction interval. The prediction interval was calculated assuming Student's t distribution with five degrees of freedom.

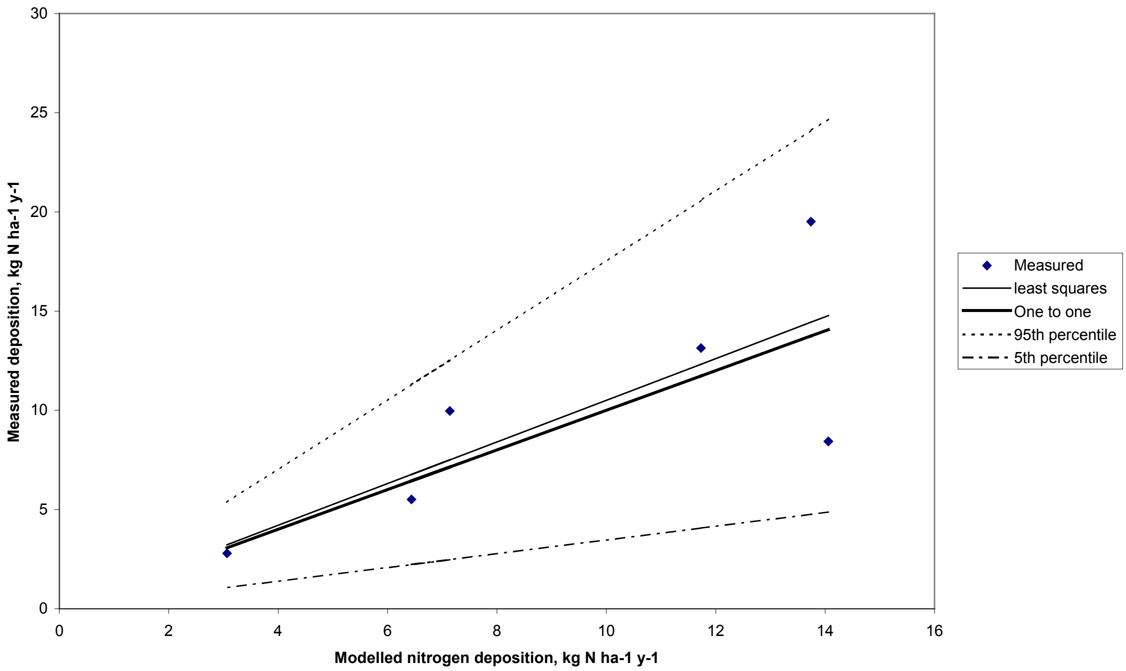

Figure 5.2.4: Comparison of modelled and measured total nitrogen deposition

Table 5.2.2 shows the results of the Monte Carlo simulation at reference sites. It shows the results of the baseline model run, the estimated nitrogen deposition based on measurements, the 5th and 95th percentiles and mean of the modelled estimates from the Monte Carlo simulation. In several cases, the measurement-based estimate is outside the 90th percentile range of the modelled estimates. It may be concluded that that there are sources of uncertainty in the model predictions other than input parameter uncertainty.

Table 5.2.2: **Nitrogen deposition estimates from TRACK Monte Carlo simulation**

Site	**Nitrogen deposition estimates, kg N ha^{-1} y^{-1}**				
	Baseline modelled	**Measurement-based estimate**	**5th percentile of modelled estimates**	**95th percentile of modelled estimates**	**Mean of modelled estimates**
Yarner Wood	14.1	8.4	11.4	17.7	14.9
Stoke Ferry	13.7	19.5	12.0	21.9	17.0
High Muffles	11.7	13.1	10.1	17.5	13.8
Lough Navar	6.4	5.5	5.3	7.2	6.2
Eskdalemuir	7.1	10.0	5.0	7.9	6.4
Strathvaich	3.1	2.8	2.3	3.2	2.8

5.3 Critical load exceedances

5.3.1 Introduction

The predicted rates of sulphur and nitrogen deposition were compared with critical loads for coniferous forest for the South East of England provided by CEH. The critical loads data used for this part of the report are not based on CEH's most recent evaluations of the critical loads: the values used here are intended to demonstrate the methodology only. The critical load data contains values of the critical loads for each one km square where there is coniferous forest present. These include:

- $CL_{max}S$, the critical load for sulphur deposition;
- $CL_{min}N$, the minimum (threshold) critical load for nitrogen.

The exceedance of the critical load was calculated from the sulphur deposition rate, S and the nitrogen deposition rate, N.

For $N < CL_{min}N$: $E = S - CL_{max}S$

For $CL_{min}N < N < CL_{max}S + CL_{min}N$: $E = S + N - CL_{max}S - CL_{min}N$

For $N > CL_{max}S + CL_{min}N$: $E = S$

CEH and Skeffington Consultants carried out a Monte Carlo analysis for the critical loads at three coniferous forest sites (Liphook, Aber and Thetford) as part of Subtask 1.3 (See Sections 2, 3 and 4). The analysis provided 5,000 paired estimates of the critical loads for each of the three sites. These three sites were used to estimate the uncertainty at other sites in the preparation of the maps.

5.3.2 Baseline estimates of exceedance

Rates of deposition calculated by the baseline model run were compared with the critical load estimates to provide deterministic exceedance estimates at each one km square with coniferous forest in the South East of England. Figure 5.3.1 shows a map of the predicted exceedance. There are 235 out of 2,588 one km square areas with coniferous forest where the predicted deposition rate exceeds the critical load.

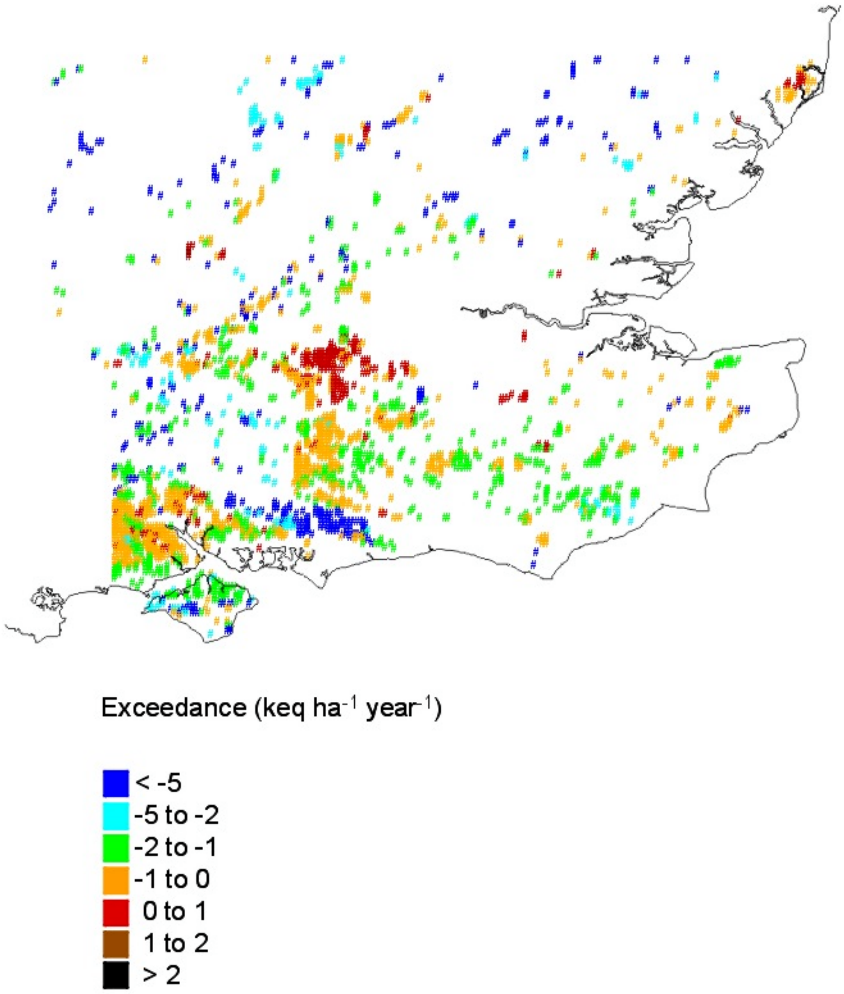

Figure 5.3.1: Predicted exceedance of critical loads for coniferous forest, baseline model run

The baseline model predictions were then adjusted for model bias on the basis of the regression analysis shown in Figure 5.2.2 and Figure 5.2.4. Modelled rates of sulphur deposition were multiplied by 0.72: modelled rates of nitrogen deposition were multiplied by 1.05. The resulting predictions of critical load exceedance are shown in Figure 5.3.2. There are 59 out of 2,588 one km square areas with coniferous forest where the predicted deposition rate exceeds the critical load.

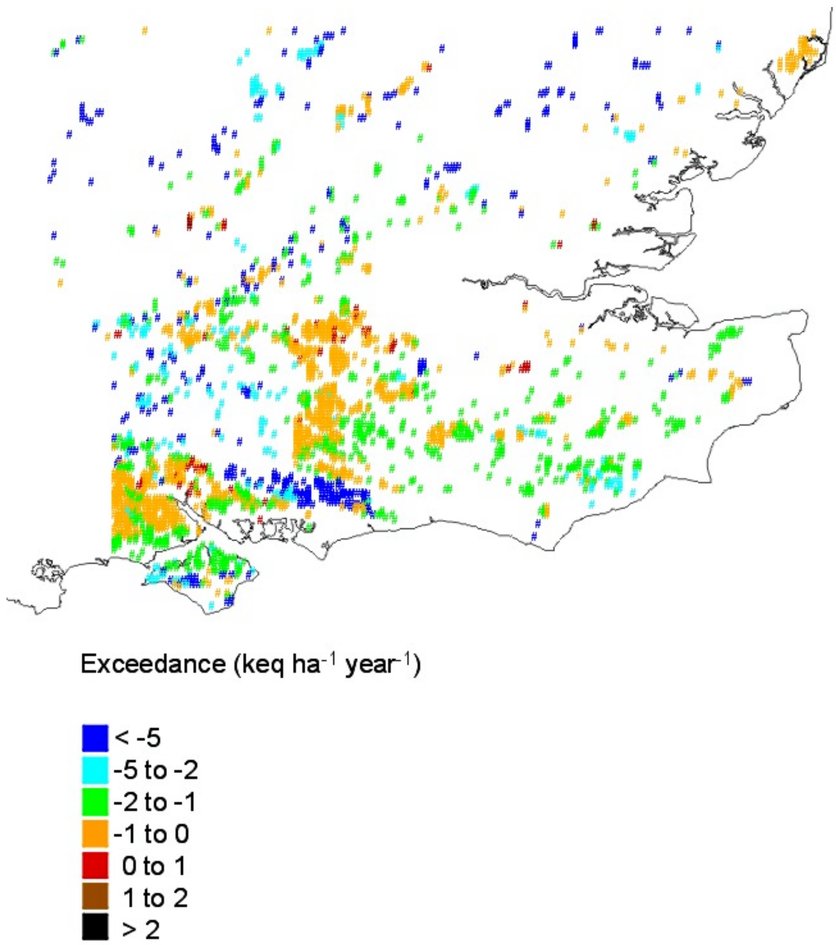

Figure 5.3.2: Predicted exceedance of critical loads for coniferous forest, baseline model run adjusted for deposition model bias

5.3.3 Monte Carlo simulation – statistical parameterisation

5.3.3.1 Uncertainty in critical loads

Estimates of uncertainty in the exceedance of critical loads in coniferous forests in South East England were then prepared to contrast with the deterministic estimates presented in Section 5.3.2. To achieve this, a Monte Carlo simulation to predict exceedance was carried out assuming that the error in the critical loads at each one km square has a bivariate normal distribution. The coefficients of variation (standard deviation divided by mean) for $CL_{max}S$ and $CL_{min}N$ and the correlation coefficient were determined from the analysis of the Monte Carlo simulations for Liphook, Aber and Thetford carried out by CEH and Skeffington Consultants.

lists the coefficients of variation and correlation coefficients determined from the Skeffington Consultants' simulations. Separate simulations were carried out assuming that all the one km squares had "Liphook type", "Aber type" or "Thetford type" probability distributions described by the parameters in Table 5.3.1. The deposition predictions were based on the baseline model run adjusted for model bias. One hundred model runs were used in the simulation.

Table 5.3.1: Coefficients of variation of critical loads

Site	Coefficient of variation		Correlation coefficient
	$CL_{max}S$	$CL_{min}N$	
Liphook	0.77	0.11	-0.25
Aber	0.26	0.05	-0.15
Thetford	0.35	0.19	-0.03

Table 5.3.2 shows the predicted numbers of one km squares with coniferous forests where the predicted deposition exceeds the critical loads. The numbers shown are for the median and 95th percentile predictions of exceedance, that is, for each square the 50th and 95th highest predictions of the exceedance in each square were taken.

Table 5.3.2: Number of one km squares where the specified percentile of predicted exceedances exceeds zero

Assumed probability distribution type	Median	95th percentile
Liphook	74	2346
Aber	64	540
Thetford	69	972

The number of one km squares where the median estimate of the exceedance was more than zero was similar to the estimate based on the bias-adjusted baseline predictions. The number of one km squares where the 95th percentile estimate of the exceedance was more than zero was much greater. The calculated numbers of one km squares where the 95th percentile exceedance is more than zero depends to a great deal on the assumed distribution of errors in the critical load estimates.

The number of one km squares where the 95th percentile of calculated exceedance values exceeds zero is not the same statistic as the 95th percentile of estimates of the number of exceedances. Table 5.3.3 shows the median and 95th percentile estimates of the number of one km squares where the predicted deposition exceeds the critical load. Table 5.3.2 was calculated as follows:

> For each site, calculate the 95th percentile or median of the predicted exceedances, then count the number of sites where the specified percentile exceeds zero.

Table 5.3.3. was calculated as follows:

> For each iteration of the Monte Carlo simulation, count the number of sites where the predicted exceedance exceeds zero, then calculate the 95th percentile or median of the number of sites over all the iterations.

In the derivation of Table 5.3.3, it has been assumed that the critical loads at each site have been estimated independently; that is, it has been assumed that there is no correlation between the critical load estimates at different sites. (It is not clear to what extent the estimates of critical loads are correlated between sites.)

Table 5.3.3: **Median and 95th percentile estimates of the number of one km squares exceeding the critical load**

Assumed probability distribution type	Median	95th percentile
Liphook	549	584
Aber	147	163
Thetford	236	257

The differences between the results of the two calculations are clear and emphasise the difference between the two statistics. Table 5.3.2 would be useful in assessing the extent of exceedance at individual sites.

Table 5.3.3 would be useful in assessing the extent of exceedance over the whole region. The 95th percentile shown in Table 5.3.3 is slightly greater than the median. Statistical theory suggests that 95th percentile of the estimates would converge towards the median as the number of sites increases.

Figure 5.3.3 shows the 95th percentiles of exceedances at each one km square, assuming a Thetford type distribution of critical load errors and the baseline predictions of deposition adjusted for model bias.

We have considered whether the probability of exceedance of the critical load is a more robust statistic. The calculated probability appears to be less sensitive to the choice of the probability distribution of the critical loads. Figure 5.3.4 shows the probability of exceedance at each of the one km squares, based on the Aber and Thetford probability distributions plotted against the probability of exceedance calculated using the Liphook probability distribution. However, comparison of the exceedance probability maps for Thetford type and Liphook type distributions highlights the differences resulting from the choice of distribution.

In conclusion, the choice of the assumed probability distribution for the critical loads has a significant effect on the apparent uncertainty in exceedance prediction.

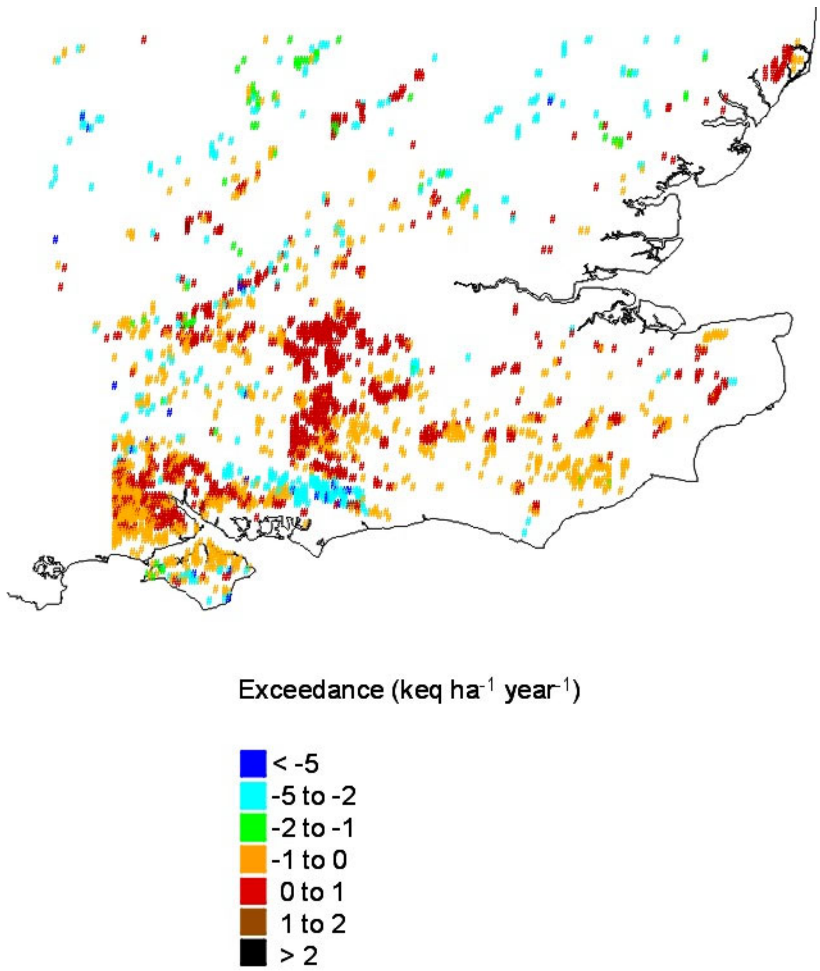

Figure 5.3.3: 95th percentile of predicted exceedances assuming Thetford type distribution of critical loads and baseline predictions of deposition adjusted for model bias

Figure 5.3.5 and Figure 5.3.6 show the probability of exceedance assuming Thetford and Liphook type distribution of critical loads and baseline predictions of deposition adjusted for model bias

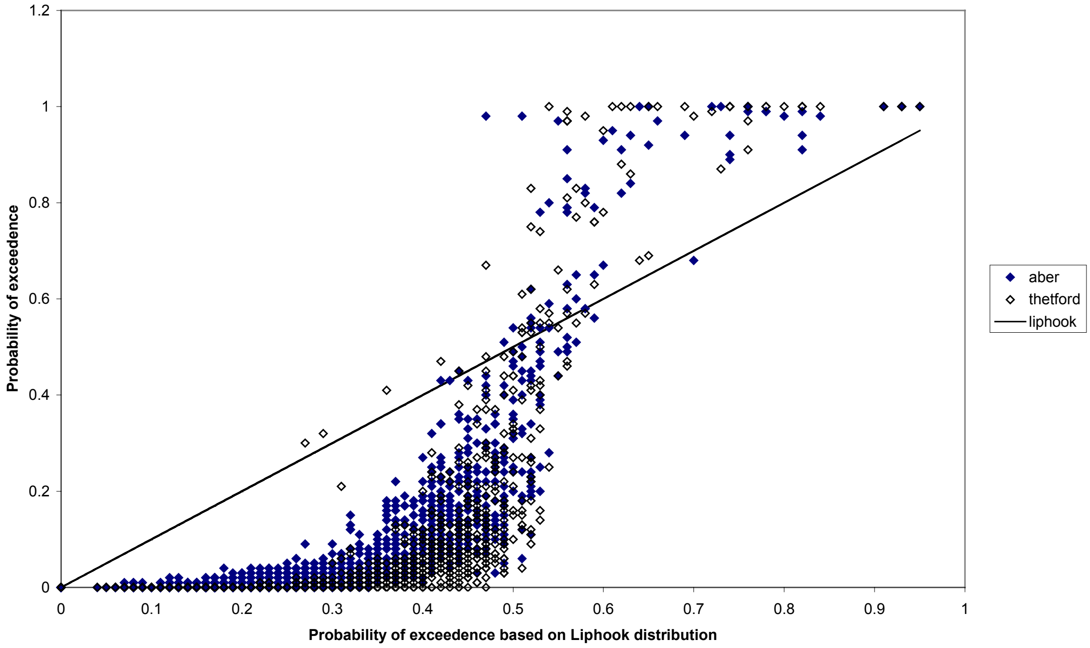

Figure 5.3.4: Comparison of the probability of exceedance at one km squares for different assumed probability distributions of the critical loads

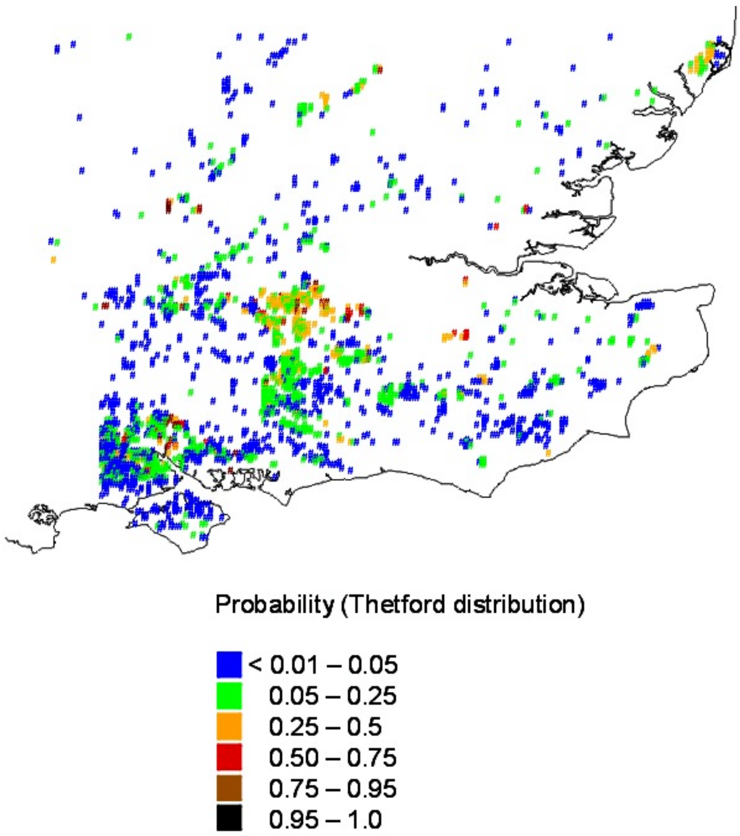

Figure 5.3.5: **Probability of exceedance assuming Thetford type distribution of critical loads and baseline predictions of deposition adjusted for model bias**

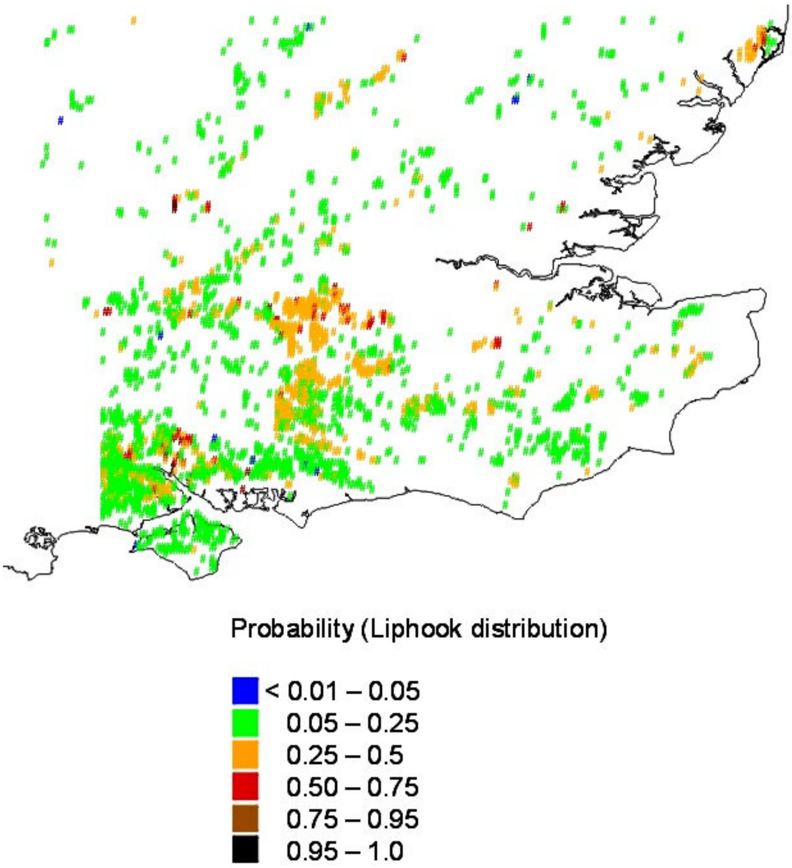

Figure 5.3.6: **Probability of exceedance assuming Liphook type distribution of critical loads and baseline predictions of deposition adjusted for model bias**

5.3.3.2 Uncertainty in Deposition Predictions

A Monte Carlo simulation to predict exceedance was carried out, assuming that the error in the predicted deposition has Student's t distribution with five degrees of freedom, with the coefficient of variation derived from the regression analysis against measured values (Figure 5.2.2 and Figure 5.2.4). The model bias adjustment and coefficients of variation for sulphur and nitrogen deposition are shown in Table 5.3.4. Examination of the data suggested a weak correlation between the errors in the sulphur and nitrogen deposition predictions. It was assumed that they were independent for this analysis, and fixed values for the critical loads for each one km square were used.

Table 5.3.4: Deposition uncertainty distribution parameters

	Sulphur	Nitrogen
Model bias adjustment	0.72	1.05
Standard deviation	0.198	0.323

The median prediction of exceedance was greater than zero for 70 of the 2,588 one km squares. The 95th percentile of predicted exceedances was greater than zero for 678 of the one km squares. These values may be compared with those in Table

5.2.4. It was concluded that the uncertainties in the critical loads and the uncertainties in the deposition contribute approximately equally to the overall uncertainty in exceedance estimates.

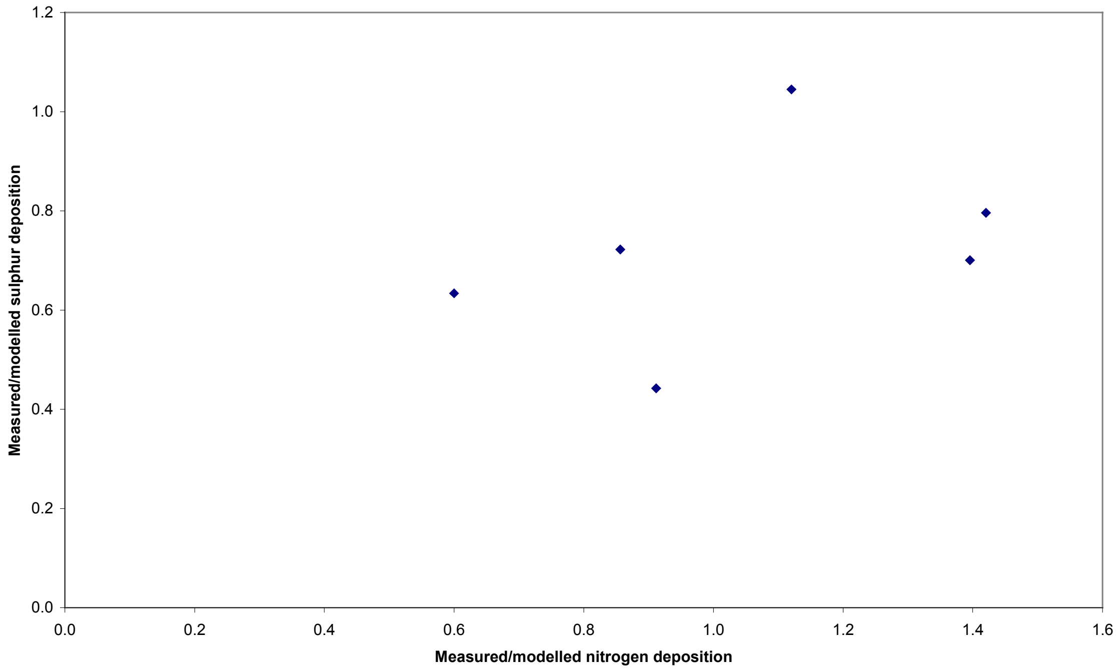

Figure 5.3.7: Errors in measured/modelled sulphur and nitrogen deposition

A separate simulation was carried out assuming that the errors in the nitrogen deposition estimates were correlated with the errors in the sulphur deposition estimates, with a correlation coefficient of 0.3. The median prediction of exceedance was greater than zero for 77 of the 2,588 one km squares. The 95th percentile of predicted exceedances was greater than zero for 744 of the one km squares. It was concluded that the correlation results in a relatively small increase in the predicted numbers of exceedances. It was therefore assumed that the errors were independent for the remaining simulations.

5.3.3.3 Combined uncertainties from deposition estimates and critical load estimates

A Monte Carlo simulation was carried out assuming that the critical loads had a bivariate normal distribution (5.3.3.1) and the deposition estimates had Student's t distribution (5.3.3.2). Table 5.3.5 shows the predicted numbers of one km squares where the median and 95th percentile exceedance predictions were greater than zero. Table.5.3.6 shows the median and 95th percentile estimates of the numbers of squares exceeding the critical loads.

Table 5.3.5: Number of one km squares where the specified percentile of predicted exceedances exceeds zero - combined uncertainties

Assumed probability distribution type	Median	95th percentile
Single value	70	678
Liphook	105	2,338
Aber	86	955
Thetford	87	1,240

Table 5.3.6: Median and 95th percentile estimates of the number of one km squares exceeding the critical load - combined uncertainties

Assumed probability distribution type	Median	95th percentile
Liphook	607	634
Aber	257	277
Thetford	329	358

Comparing Table 5.3.2 with Table 5.3.5 and

Table 5.3.3 with Table 5.3.6 confirms that consideration of the uncertainties in both the critical loads and the deposition estimates increases the uncertainty in the estimated exceedance.

Figure 5.3.8 shows the predicted 95th percentile of exceedance estimates at each one km square with coniferous forest (Thetford distribution).

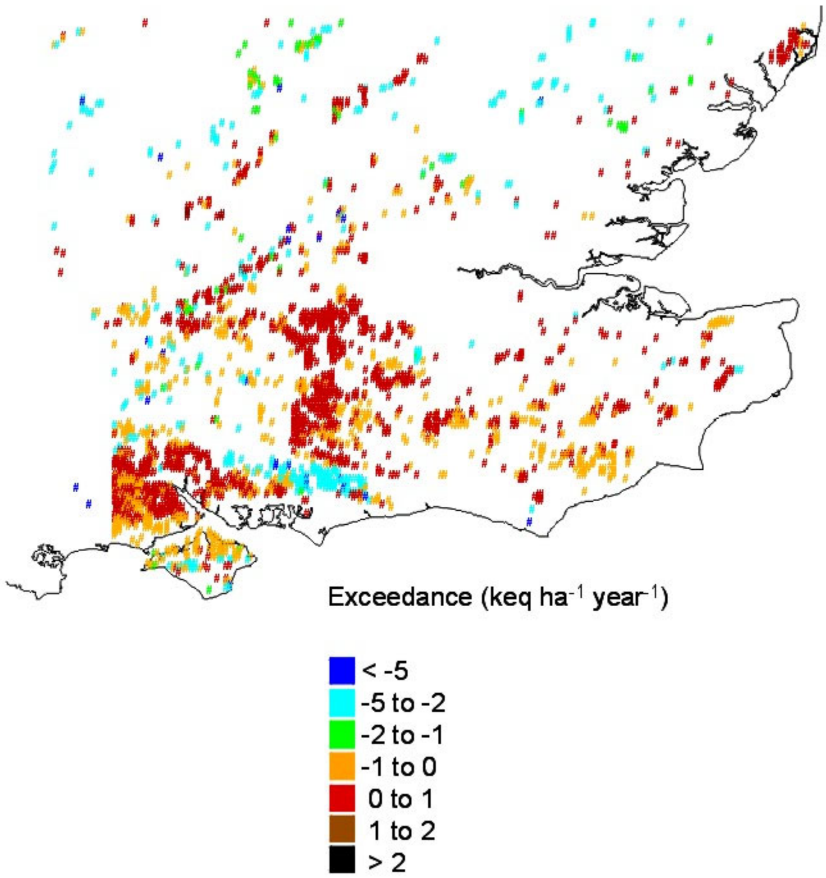

Figure 5.3.8: **Predicted 95th percentile of exceedance estimates at each one km square with coniferous forest (Thetford).**

5.3.4 Monte Carlo simulation – direct sampling

5.3.4.1 Uncertainty in critical loads

A simulation was carried out to compare the baseline deposition estimates, adjusted for model bias, with the critical loads at each one km square. The critical load values for each square were assumed to have Liphook type, Aber type or Thetford type probability distributions. Paired estimates of the critical loads $CL_{max}S$ and $CL_{min}N$ were taken at random from the probability distributions provided by Skeffington Consultants and CEH and normalised by division by the mean value.

Thus for each one km site (i):

$$CL_{k,i} = CL_i \frac{CL_{k,Liphook}}{CL_{mean,Liphook}}$$

where k is the iteration number.

Table 5.3.7 shows the numbers of one km squares where the specified percentile of predicted exceedances exceeds zero. Table 5.3.8 shows the median and 95th percentile estimates of the number of one km squares exceeding the critical load.

Table 5.3.7: Number of one km squares where the specified percentile of predicted exceedances exceeds zero

Assumed probability distribution type	Median	95th percentile
Liphook	74	2,546
Aber	77	252
Thetford	85	767

Table 5.3.8: Median and 95th percentile estimates of the number of one km squares exceeding the critical load

Assumed probability distribution type	Median	95th percentile
Liphook	585	615
Aber	101	113
Thetford	236	236

Comparing Table 5.3.2 with Table 5.3.7 and

Table 5.3.3 with Table 5.3.8 indicates that the use of the bivariate normal distribution for critical loads provides an adequate description of the probability distributions for the critical loads.

5.3.4.2 Uncertainty in the deposition

A simulation was carried out to compare the baseline critical load estimates with the modelled deposition estimates. Estimates of the deposition were taken from the deposition model Monte Carlo simulation, adjusted for model bias. The deposition model run outputs were correlated between receptor sites: a set of input parameters that results in a high prediction in the deposition at a receptor is also likely to produce a high estimate at an adjacent receptor. Sulphur and nitrogen deposition rates at each site were also correlated. These features were retained in this simulation. Predictions of the simulation are summarised as the 'single value' probability distribution type in the second line of Table 5.3.9 and Table 5.3.10.

The median of predicted exceedances is greater than zero in more of the one km squares for the direct simulation than for the parameterised simulation in Section 5.3.3. This is because median estimates of nitrogen and sulphur depositions predicted by the TRACK Monte Carlo simulations of deposition are larger than the baseline estimates.

5.3.4.3 Combined uncertainties from deposition estimates and critical load estimates

Simulations were carried out using the results of the Monte Carlo simulations of both deposition estimates (Section 5.3.4.2) and critical load estimates (Section 5.3.4.1). Separate simulations were carried out for Liphook type, Aber type and Thetford type critical load probability distributions.

Table 5.3.9 shows the predicted numbers of one km squares where the median and 95th percentile exceedance predictions were greater than zero. Table 5.3.10 shows the median and 95th percentile estimates of the numbers of squares exceeding the critical loads.

Table 5.3.9: Number of one km squares where the specified percentile of predicted exceedances exceeds zero - combined uncertainties

Assumed probability distribution type	Median	95th percentile
Single value	344	803
Liphook	397	2,543
Aber	373	890
Thetford	417	1,323

Table 5.3.10: Median and 95th percentile estimates of the number of one km squares exceeding the critical Load - combined uncertainties

Assumed probability distribution type	Median	95th percentile
Single value	367	803
Liphook	766	1,032
Aber	411	781
Thetford	508	886

5.4 Conclusions and recommendations

The TRACK model was used to predict sulphur and nitrogen deposition at monitoring sites across the UK. Modelled estimates of sulphur and nitrogen deposition were compared with estimates derived from measured rates of wet deposition and air concentrations of deposited species. The following conclusions were drawn:

- The TRACK model provides reasonable estimates of the deposition rates for sulphur and nitrogen when compared with estimates of deposition based on measured values of wet deposition and air concentrations.
- The modelled estimates of deposition can be improved by small bias adjustments based on regression analysis.
- The regression analysis provides a measure of the uncertainty in the modelled deposition rates.

The uncertainty in deposition model estimates resulting from the uncertainty in model inputs can be estimated by means of Monte Carlo simulation of model input parameters. The 5-95th percentile range of model predictions was compared with the estimates derived from measured data. It was concluded that the input parameter uncertainty provides a substantial contribution to the overall uncertainty in deposition model predictions. However, input parameter uncertainty does not explain all the model errors.

Skeffington Consultants and CEH have used Monte Carlo simulation to determine the uncertainty in critical load estimates, taking account of the uncertainty in the input parameters to the critical load equations. The joint probability distribution of the critical load for sulphur, $CL_{max}S$, and the minimum critical load for nitrogen, $CL_{min}N$, was determined at three coniferous woodland sites: Liphook, Aber and Thetford. The uncertainty in critical load estimates is not small compared with the critical loads.

The results of the Monte Carlo simulation of critical loads were used in this study in the further simulation of critical load exceedances, where the exceedance was calculated as the difference between the modelled deposition and the critical load. The critical load exceedance was calculated at each of the 2,588 one km squares in the area of South East England selected for this feasibility study. The joint probability distribution for the critical loads at each site was assumed to take one of three forms:

- single value for the critical loads for sulphur and nitrogen;
- bivariate normal distribution;
- direct sampling from the joint probability distribution developed from the Monte Carlo simulation of critical loads.

The joint probability distributions were assumed to be Liphook type, Aber type or Thetford type depending on the critical load data set used.

The joint probability distribution for the deposition rate at each site was also assumed to take one of three forms:

- single values for sulphur and nitrogen deposition derived from the TRACK model runs with the baseline input parameters;
- bivariate Student's t distribution derived from the regression analysis of model predictions and measurement estimates of sulphur and nitrogen deposition;

- direct sampling from the TRACK Monte Carlo simulation of sulphur and nitrogen deposition.

The following combinations of probability distributions were investigated:

- single-single
- single-bivariate
- bivariate-single
- bivariate-bivariate
- single-direct
- direct-single
- direct-direct

Four metrics were used to compare the model predictions. These were:

1. The number of one km square coniferous forest areas where the median of the predicted exceedances was more than zero.
2. The number of one km square coniferous forest areas where the 95th percentile of the predicted exceedances was more than zero.
3. The median of the estimates of the number of squares where the deposition exceeds the critical load.
4. The 95th percentile of the estimates of the number of squares where the deposition exceeds the critical load.

Metrics 1 and 2 provide a measure of the likelihood of exceedance for individual one km squares. Metrics 3 and 4 provide a measure of the overall sensitivity of the region to acid deposition.

The data set chosen to represent the critical load joint probability distribution did not affect Metric 1 significantly. However, the choice of distribution had a substantial effect on the other metrics. It was concluded that the probability distribution functions should be assigned to receptors on a case-by-case basis.

The bivariate probability distribution used to represent the critical loads data set produced similar results to direct sampling from the original distribution. It was concluded that a bivariate normal distribution may be used to represent the joint probability distribution of critical loads at receptor sites.

The use of single value estimates for the critical loads leads to a substantial underestimation of Metric 3. It is essential to consider the joint probability function of the critical loads when assessing the extent of exceedance of the critical loads over a region.

Direct sampling from the TRACK model deposition probability distribution leads to higher estimates of Metric 1 than the single value model predictions, or those using the bivariate t-distribution based on the regression analysis. This situation arises because the median of the TRACK model Monte Carlo simulation predictions of deposition at most sites is larger than the baseline model output used for the regression analysis. It is recommended that the bivariate t distribution based on the regression analysis is used in critical load assessments, because it provides a direct link to the observed values and takes account of the errors in the model predictions.

In summary, the bivariate-bivariate combination of probability distributions is recommended for assessment and mapping of critical load exceedance. It is important that the joint probability distribution used to represent critical loads at each receptor site should be representative of the soil conditions at that site. This may be a difficult condition to fulfil, as the three known distributions (Liphook, Thetford and Aber) give rather different results (see above), and other sites may produce other distributions. There is indeed little basis at the moment for associating any square without measurement data with a given distribution.

6 Frameworks for the Environment Agency's regulatory role

Summary

- This section describes the process for carrying out critical load, exceedance and uncertainty assessments at the local (point, site), regional and national scales, and illustrates these with flow charts.
- Uncertainties depend on whether site-specific or national data are used; values differ between sites and habitat types, and values for a site are not necessarily applicable regionally or nationally. Different approaches are needed at each scale.
- Sensitivity analyses to determine the key input parameters that influence the critical load and exceedance values, together with calculations of the probability of exceedance, are the key tools for carrying out an assessment at any scale.
- Uncertainties may be reduced at the point- or site-specific scale, if more site-specific data for sensitive parameters (such as base cation and calcium weathering rates and deposition) are collected.
- The points, sites or areas that have a very high or very low probability of exceedance can be identified using the methods proposed. For those areas where the probability of exceedance is uncertain, more information to improve the estimates of the key input parameters is required.

6.1 Introduction

Critical loads are often represented as a dose-response function (Figure 6.1.1) and defined as the deposition load below which significant harmful effects on specified elements of the environment (such as soil, vegetation, water) do not occur according to present knowledge (Nilsson and Grennfelt, 1988). Exceedance of critical loads is defined as the amount of excess deposition above the critical load.

Sections 1 to 5 of this report describe the uncertainties in the calculation of critical loads and their exceedances, both at the site-specific and regional/national scales. The results of this work show that there is no simple value of uncertainty that can be assigned to critical loads or exceedances. The critical (most sensitive) parameters in the calculations differ depending on whether national or site-specific data are used and differ between habitat types. This is indicated in Figures 6.1.2 and 6.1.3 which show that when calculations are based on national data (as in Section 3), the key uncertainties are for base cation and calcium weathering, critical Ca:Al ratio, nitrogen uptake and leaching, and deposition.

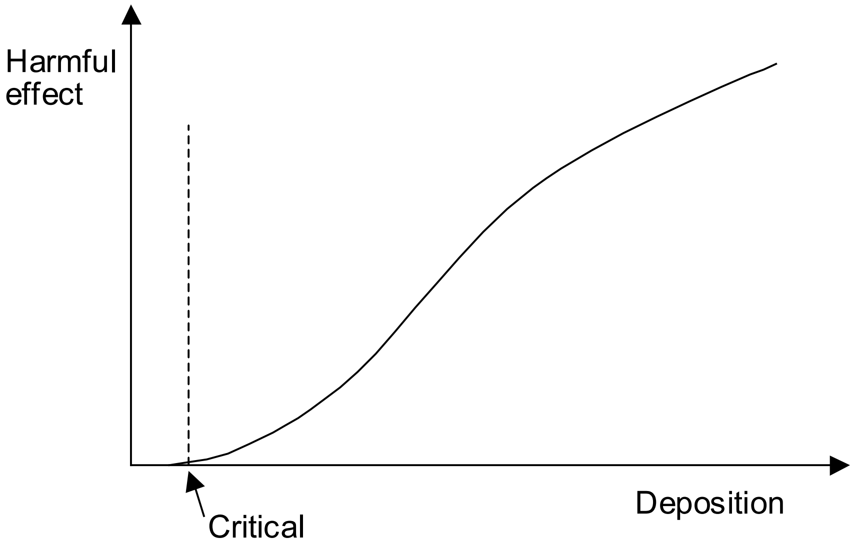

Figure 6.1.1: Critical load represented by a dose-response function

The analysis based on site-specific data (as in Section 2) showed that calcium and base cation deposition were the most sensitive parameters, but nitrogen leaching and sulphur and nitrogen deposition were also important. At the site level, the sensitive parameters varied from one site to another (Section 2). Uncertainties at the point or local scale can be reduced if all or most of the input data are site-specific, rather than derived from the national databases, since in general site-specific uncertainty ranges are narrower and so there is higher confidence that the input parameters are appropriate. However, there are remarkably few sites in the UK where many parameters have been measured; this is especially true for the critical parameters of calcium and base cation weathering and deposition, as these are both difficult and expensive to measure reliably. The main focus of critical loads research over the last 10-15 years has been the development of a national database to produce maps and assessments at the national scale. At this scale, there is little that can be done to reduce uncertainties because of the scale and cost of collecting sufficient data.

Analyses carried out under this study have shown how local conditions (soil type, habitat type, deposition) can vary from information derived from national maps/data sets, such that critical loads, exceedances and their uncertainties based on site-specific data can be lower or higher than those based on national data. There are only three sites with contrasting characteristics for which it was possible to compare the uncertainties based on site-specific and national data (Section 4); the results for the probability of exceedance are repeated in Table 6.1.1 below.

Table 6.1.1: Comparison of probability of exceedance results based on national and site-specific data for three woodland sites

Site	Probability of exceedance based on:	
	Site-specific data	**National data**
Thetford	2%	0.3%
Aber	96%	93%
Liphook	68%	100%

Site-specific and national results are similar at Thetford and Aber, but differ markedly for Liphook. With only three sites it is difficult to draw definitive conclusions, but the national data should provide an acceptable prediction of the status of a site. Hence, the results are currently insufficient to derive a relationship between national and site-specific results, or to provide information on when national data may be appropriate (or not) for use at the site level.

The Environment Agency's regulatory role is more focused on assessments at the point- or site-specific scales, rather than at the national scale, which is the remit of Defra. The questions to be answered will include:

- What is the probability that the critical load is exceeded?
- Is this probability acceptable?
- If not, to what level should deposition be reduced to make it acceptable, and what are the implications for Environment Agency-regulated sources?
- If the probability of exceedance is acceptable, then to what level can deposition be increased before it becomes unacceptable?
- What confidence do we have that a site will recover by a defined time, with a given deposition scenario?

The probability of exceedance will help answer the first two questions. Dynamic models, already developed for acidified waters in the UK (such as Cosby *et al.*, 2001) are currently being developed for both acidity and nitrogen for terrestrial habitats (see Defra-funded project: *UK Research on the Eutrophication and Acidification of Terrestrial Ecosystems*) and these will enable the latter questions to be addressed in due course. The probability of exceedance can be calculated for a single point, a site (such as a designated area) or regionally/nationally (see below). Using the probability of exceedance raises further questions:

- Above what percentage probability of exceedance is a site considered to be at high or very high risk of exceedance?
- Below what percentage probability of exceedance is a site considered to be at a low or very low risk of exceedance?
- How can or should these thresholds be set?

Scientists typically use the 95 per cent (or five per cent) levels to indicate significance, but this probability of one-in-twenty is just a long-standing convention. For a regulator, the level of significance should be related to the "cost" of an inappropriate decision (where a "cost" can be financial, such as preventing a development or requiring ameliorating activities, or it might be environmental, such as the loss of a particular species or habitat). If conventional thresholds are applied - greater than 95 per cent for a site with a very high risk of exceedance and less than five per cent for a very low risk of exceedance - this leaves a large category of 'uncertain exceedance'. This approach could be applied when using national maps of the probability of exceedance, as a tool to screen individual points or sites. It will identify the extremes (very low/high risk of exceedance) and highlight sites that fall between thresholds where additional site-specific data should be collected to reduce the uncertainty of the results. Alternatively, another expert, for example a conservation officer, may simply prefer to accept a threshold of greater than 50 per cent probability of exceedance to define a site as exceeded and therefore be more certain of action being taken to protect a site. Additionally, one could argue that the probability of exceedance could be defined in descriptive terms only (Table 6.1.2), not quoting the corresponding percentage values.

Table 6.1.2: Example of descriptive labels for percentage probability of exceedance

Probability of exceedance	Descriptive term
< 5%	Very low risk of exceedance
5-25%	Low risk of exceedance
25-75%	Medium risk of exceedance
75-95%	High risk of exceedance
> 95%	Very high risk of exceedance

The problems with assigning descriptive terms are: (a) they depend on who defines the categories; (b) used alone, they are not transparent and thus may not be defensible. It must always be possible to explain exactly what they refer to and how they have been defined.

The issue of acceptable thresholds therefore needs further consideration. But in general, the probability of exceedance will help the Environment Agency make decisions about the status of a site in terms of critical loads exceedance. How the results are then used within the Environment Agency's regulatory role is another issue, since the next step may be, if the probability of exceedance is below x per cent, is it okay for industrial plant A to emit y kT of sulphur or nitrogen? Or if the probability of exceedance is already very high (greater than 95 per cent), will additional emissions of S or N have any further impact on the site? In the latter case, an increased load is likely to slow down the (chemical and biological) recovery of the site. Dynamic models will help in addressing the issue of time scales for recovery.

The sections below outline the steps for calculating critical loads, exceedances and the probability of exceedance at three scales:

- Points: individual locations of negligible spatial extent.
- Sites: areas where it can usually be assumed that there is uniform deposition, for example, an SSSI might constitute a site.
- Regions/National: areas where deposition is expected to vary spatially, for example, East Anglia, England, UK.

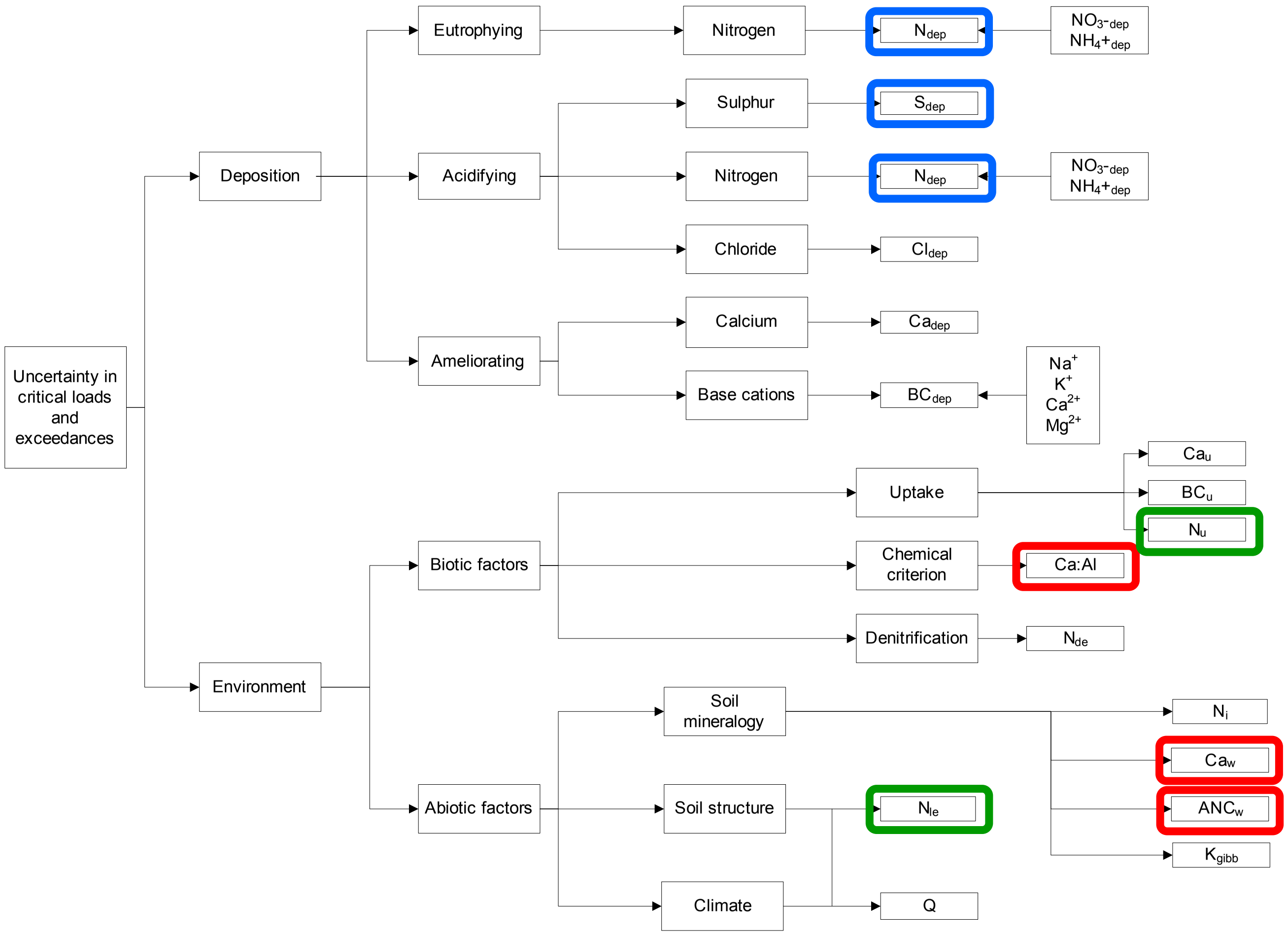

Figure 6.1.2 Uncertainties at the national scale for managed coniferous woodland. Boxes are colour coded to show parameters most sensitive for calculation of acidity critical loads (red), nutrient nitrogen critical loads (green) and exceedances (blue); these may vary for other habitat types. Refer to glossary of terms for definitions of parameters.

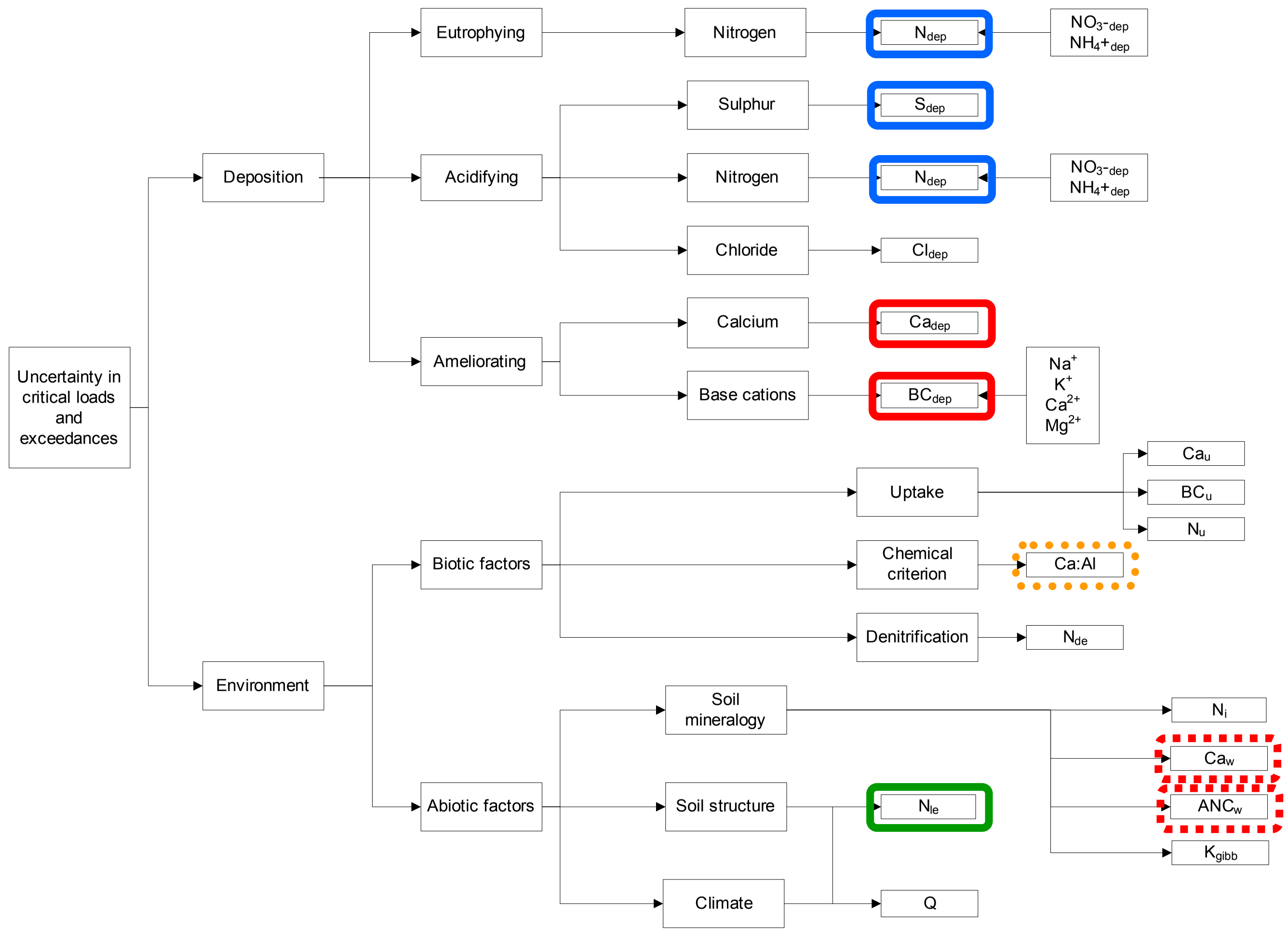

Figure 6.1.3: Uncertainties at the site-specific scale for managed coniferous woodland. Boxes are colour coded to show parameters most sensitive for calculation of acidity critical loads (red and orange), nutrient nitrogen critical loads (green) and exceedances (blue); these may vary between sites and habitats. Refer to glossary of terms for definitions of parameters.

The assessments below follow a slightly different sequence of operations depending on the spatial scale of interest. The methods for each are only loosely coupled, so that aggregation and disaggregating may yield different results.

The first step for all three assessments is to determine if the point location, site features (habitats/species), or region (habitat) is sensitive to acidification or eutrophication. Databases such as the UK Air Pollution Information System (APIS: http://www.apis.ac.uk/) can assist in making this decision at the outset. The methods are applicable for both acidity and nutrient nitrogen assessments.

6.2 Point assessments

A point assessment assumes there is a single habitat at a single location. The process of assessment is provided in Figure 6.2.1. Decisions and processes are annotated a) to j) and consist of:

a) If critical loads, exceedances and probability of exceedance have previously been determined for the point location, proceed to stage (h).
b) If no assessment has previously been carried out, obtain any existing point-specific data (such as critical load input data, deposition) for that location. The data may be measured directly or modelled in some form. In most cases, only a few point specific parameters will be available and in many cases there will be no specific information.
c) Extract the remaining parameters required from national databases or other data sources (for example, deposition models such as TRACK).
d) Calculate deterministic critical load and exceedance values (normal calculations). Appendix 1 of this report describes two of the commonly used critical load methods; other detailed descriptions of the agreed UK methods can be found in Hall *et al.* (2003a, 2004a).
e) Decide if deterministic results are acceptable; if yes, go ahead and use values. Note here the term "acceptable" is used to denote how certain we are of the value, not whether the value is environmentally "good" or "bad". The more point-specific data, especially for the most sensitive parameters, that are used in the calculations, the more confident one can be of the results.
f) If the deterministic values are not acceptable, perform an uncertainty analysis using a Monte Carlo simulation method; Crystal Ball™ provides a convenient 'off-the-shelf' platform to perform calculations within MS Excel, as do other commercial packages.
g) Crystal Ball can also be used to process the Monte Carlo simulation results and produce a probability distribution of exceedance values and a record of which parameters have the largest effect on predicted values (sensitivity analysis results).
h) Compare the probability of exceedance with pre-defined threshold values (refer to discussion above regarding the setting of threshold values). If the probability is greater than the upper threshold, then the critical load for the point is considered to be "definitely exceeded" and the process can be terminated.
i) If the probability of exceedance is below the lower threshold, the critical load for the point is considered to be "definitely not exceeded" and the process can be terminated.
j) If the probability of exceedance is below the upper threshold and above the lower threshold, exceedance of the critical load is uncertain. In these circumstances, it may be justifiable to revise the range or distribution of one or more key parameters as identified by the sensitivity analysis. Alternatively,

a point-specific estimate of a parameter may be outside any published values and therefore considered to be suspect. If it is justifiable to revise the parameters, the Monte Carlo simulation (stage (f) above) and subsequent stages need to be re-run; if not, the process terminates and the assessment for the point is inconclusive.

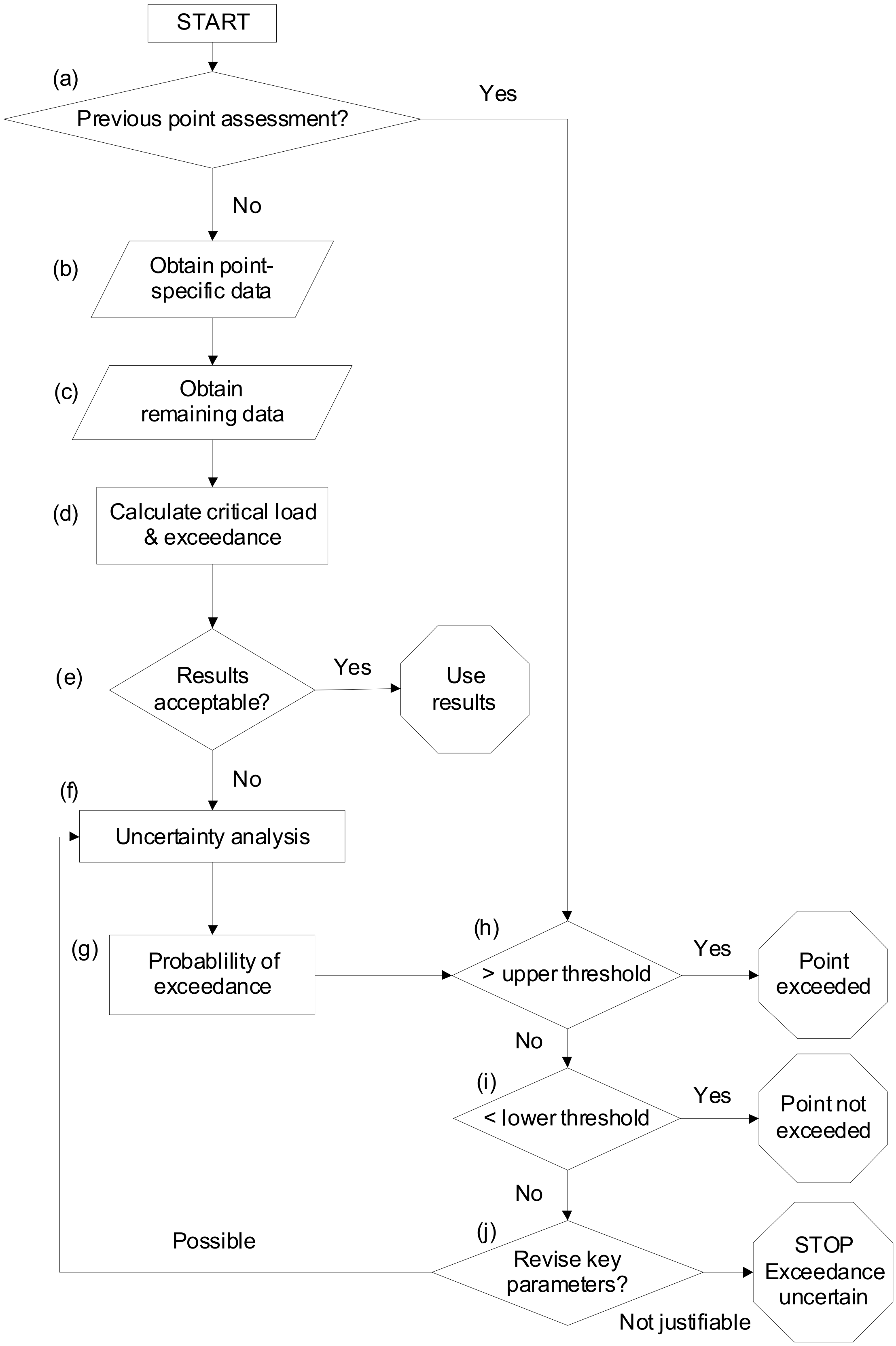

Figure 6.2.1: Framework for a point-specific assessment

6.3 Site-specific assessments

It is envisaged that this scale of assessment will be performed on a designated site such as a Natura 2000 site (SACs: Special Areas of Conservation; SPAs: Specially Protected Areas) or a Site of Special Scientific Interest (SSSI). Sites like SACs range in size from around one hectare to more than 100,000 hectares, and may be composed of several land parcels which may or may not be adjacent. Sites are designated to protect one or more features, which may be species or habitats, but where the location of the protected habitat (or species) within the site is usually unavailable.

This assessment method focuses on habitats as vegetation communities, rather than species, and was developed under commissioned research for the Environment Agency (R&D Technical Report SC030310, Wadsworth and Hall, 2005). This work assumes that site-specific and national estimates contain useful information, but that both are uncertain. To optimally combine both strands of evidence, some theory of probability is required: here, we adopt a variation of Bayesian statistics called Dempster-Shafer statistics (Dempster, 1967; Shafer, 1976). In classical statistics, probability is the frequency with which something occurs, given a multitude of repetitions under identical conditions. Outside of situations like picking numbers for the National Lottery, the identical conditions necessary for a classical interpretation of probability are rarely, if ever, encountered. In Bayesian statistics, probability is defined as the degree of belief in a proposition and Bayes' Rule is used to revise the belief in the light of new evidence. Those that hold to classical statistics criticise Bayesians because of their willingness to include subjective (expert) opinion, but there are many apparently objective situations where subjectivity is critical but at least partially hidden from view.

For example, in Monte Carlo simulations a decision has to be made as to what factors to include, how they are correlated, what range of values a random variate can have and what the shape of its frequency distribution is (uniform, triangular, Gaussian). When we adopt a Bayesian point of view, we think we know something about a situation and we seek evidence that will increase or decrease our initial opinion or belief. Dempster-Shafer's Theory of Evidence is essentially an extension of the Bayesian view that is formulated in a way that makes uncertainty explicit and open to scrutiny, so that upper and lower bounds on the belief in a proposition (idea, hypothesis and so on) are generated rather than a single value.

To begin with, we consider the process of calculating or setting critical loads for a designated site. The uncertainties in applying the national critical loads data to designated sites are discussed in Section 3.4 of this report. We therefore suggest the following approaches, highlighting where national data could be used, but offering alternatives where the national data may be inappropriate. The methods are aimed at sites where the designated features are habitats rather than species, though the same approach could be applied if the habitats on which the species depend are known. Current research for JNCC includes consideration of appropriate methods for assigning critical loads (both acidity and nutrient nitrogen) to designated site (principally SSSI) features, both for habitats and species (Hall, Bealey and Wadsworth, 2005, in preparation).

6.3.1 Site-specific critical load assessments

Figure 6.3.1 outlines the proposed stages in setting critical loads for acidity; text in italics refers to the sources of information to help make the decision at each stage.

Stage 1 is the initial step in determining if the feature of interest is sensitive to acidification before proceeding further.

Stage 2 suggests consulting the map of critical load variance (see Section 3.3 and Figure 3.3.5) for the site location. If the variance in critical loads is low (where all soils within a 1 km^2 have the same value) and the site is small (less than 1 km^2), then the critical load value from the national map could be used. Otherwise proceed to Stage 3.

The method proposed for acidity is based on relating critical loads data for different soil types to the soil information available for vegetation communities of the National Vegetation Classification (NVC, Rodwell 1990 *et seq*).

Stage 3 deals with the conversion from feature habitats to NVC communities. It should be noted that more than one NVC class may be associated with a habitat type. The designated habitats of the UK's Natura 2000 sites are described by JNCC in terms of both the EU Habitats Directive Annex 1 habitat types and also the corresponding NVC classes (http://www.jncc.gov.uk/). Where this information is not readily available, the conversion to NVC class(es) can be done using (i) the NBN Habitats Dictionary (http://www.nbn.org.uk/habitats/), or (ii) Modular Analysis of Vegetation Information System (MAVIS) that assigns NVC classes to groups of species (http://www.ceh.ac.uk/products/software/CEHSoftware-MAVIS.htm).

Stage 4 applies the Endorsement Theory methodology to set acidity critical loads to each terrestrial NVC community. Queries in an Access database (supplied to the Environment Agency) provide an endorsement for each of the six soil acidity empirical critical load classes (five classes of critical load values for non-peat soils, and a separate category for peat soils). In the current implementation, five levels of endorsement are used in descending order: "definitive", "confident", "likely", "weak" and "very weak". A definitive endorsement is possible only where there is a great deal of robust evidence for a particular option (hypothesis/class) and no evidence for any other alternative, while a very weak endorsement means that there is very little evidence for that choice. Queries within the database provide reports that describe why particular endorsements were made in each case. However, the database requires further validation and quality assurance prior to its widespread application to designated sites.

Stage 5 deals with uncertainty by combining the local (Endorsement Theory) critical load with the estimate from the national maps/databases. This is explained in more detail below.

The framework for incorporating uncertainty in the site-specific acidity assessments is shown in Figure 6.3.2. Steps (a) and (b) are described under Stages 3 and 4 above. We begin this process with the endorsements for the critical load classes; the remaining steps in the assessment are described below. See Wadsworth and Hall (2005) for a detailed description of how to apply the proposed Dempster-Shafer methodology.

c) For comparison with the national critical loads data (or other data), the endorsements are converted into numeric values for belief and uncertainty required for stage (d). For example, a definitive endorsement could be translated to give a belief value of 0.9 and an uncertainty value of 0.1.
d) Site-specific critical load values from the endorsement, and national estimates from the national critical load maps, are combined using the Dempster-Shafer Theory of Evidence (DS) which is an extension to Bayesian statistics that allows the explicit representation of uncertainty. The main advantage of this formulation is that weak evidence for a proposition does not have to imply strong evidence for something else.
e) From the combined estimate of the appropriate critical load, the probability of exceedance can be calculated (for example, using Crystal Ball as for the point-specific assessments).
f) If the probability of exceedance is above the pre-defined upper threshold, the critical load for the site is considered to be definitely exceeded. If the probability of exceedance is below the pre-defined lower threshold, the critical load for the site is considered to be definitely not exceeded. If the probability of exceedance is below the upper threshold and above the lower threshold, exceedance of the site critical load is uncertain. If the latter is the case, then the above processes need to be re-examined to see if the uncertainties in any of the inputs can be reduced, for example, whether more detailed information of the soil-vegetation relationships can be obtained.

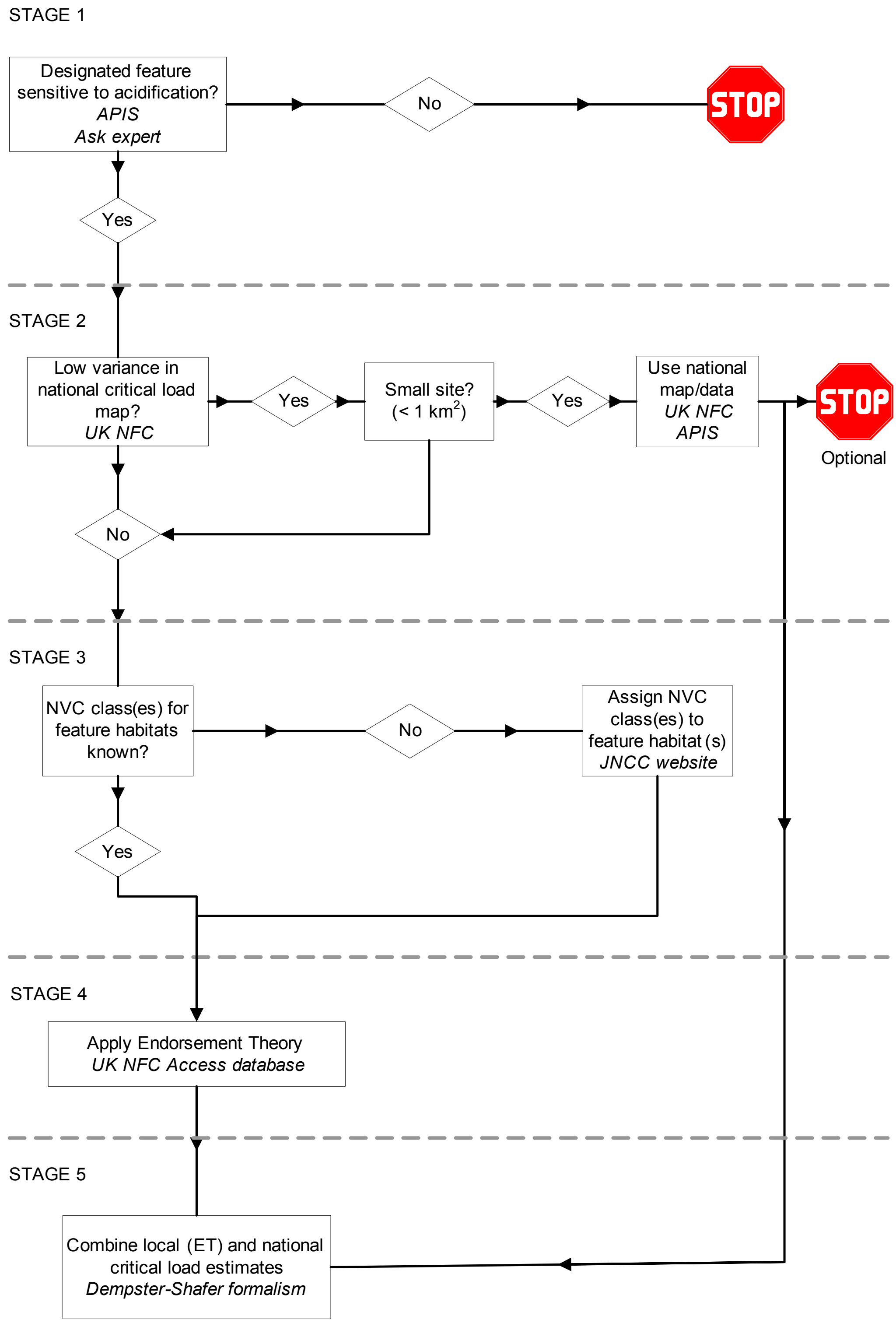

Figure 6.3.1: Framework for setting acidity critical loads for designated sites

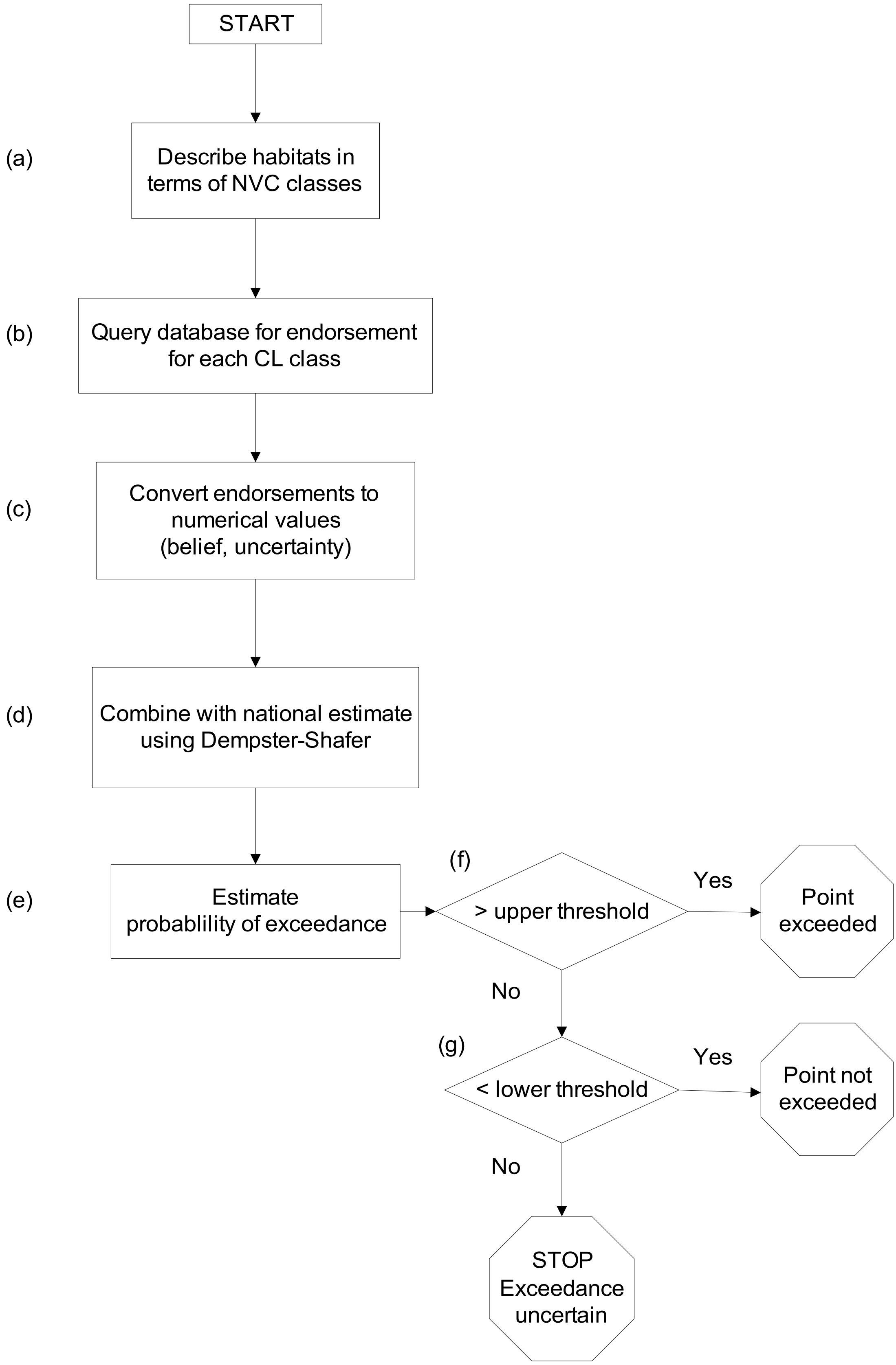

Figure 6.3.2: Framework for site-specific assessment

6.4 Site-specific nutrient nitrogen critical load assessments

Critical loads for nutrient nitrogen are based on empirical and mass balance approaches. The latter is applied in the UK to managed forest habitats and these will not be considered under this section. Empirical critical loads are estimated for different habitat types based on experimental or field evidence of thresholds for changes in species composition, plant vitality or soil processes. These critical loads are expressed as a range and have been initially set for habitat classes of the European Nature Information System (EUNIS, Davies and Moss, 2002) to enable their application across Europe (Achermann and Bobbink, 2003). In the UK, the main sensitive habitat classes have been translated into Biodiversity Action Plan broad habitats (Hall *et al*., 2003a) and single 'mapping value' critical load values set for each habitat to enable exceedances to be calculated (Table 6.4.1). Therefore, if either the broad habitat or EUNIS class is known, the appropriate critical load values can be assigned from Table 6.4.1. However, this table only lists the habitat types that it has been possible to map nationally; nutrient nitrogen critical loads may exist for other habitat types (such as neutral grassland) and appropriate values for these can be found in Hall *et al.* (2003a), Achermann and Bobbink (2003) or on the UK Air Pollution Information System (http://www.apis.ac.uk). Alternatively if only the NVC classes were known these could be translated to either broad habitats or EUNIS classes using the NBN Habitats Dictionary as above for acidity.

There is limited information available to estimate the uncertainties associated with empirical nutrient nitrogen critical loads. The range of critical load values for each EUNIS class indicates the variation in sensitivity within an ecosystem. The uncertainty is expressed qualitatively as "reliable", "quite reliable" and "expert judgement" (see Table 6.4.1). Hall *et al.* (2003b) therefore proposed extending the range of critical load values according to the reliability category as follows:

##	"reliable"	use range as published
#	"quite reliable"	±5 kg N ha^{-1} yr^{-1} beyond the range
(#)	"expert judgement"	±10 kg N ha^{-1} yr^{-1} beyond the range

The only exception to this rule was the critical load for bogs (EUNIS class D1), where the UK is using the upper limit of the range as its mapping value; to cover for this, the maximum was increased to 12 kg N ha^{-1} yr^{-1} to provide an estimate of uncertainty.

Table 6.4.1: Empirical nutrient nitrogen critical loads by habitat for the UK

Broad habitat[1]	EUNIS class	Critical load range[2] (kg N ha^{-1} yr^{-1})	UK mapping value (kg N ha^{-1} yr^{-1})
Acid grassland	Dry acid and neutral closed grassland (E1.7)	10-20 #	15
	Moist or wet oligotrophic grassland (E3.5)	10-20 #	15
Calcareous grassland	Semi-dry calcareous grassland (E1.26)	15-25 ##	20
Dwarf shrub heathland	(Lowland) dry heaths (F4.2)	10-20 ##	12
	(Lowland) *Erica* wet heaths (F4.11)	10-25 #	15
	(Upland) *Calluna* wet heaths (F4.11)	10-20 (#)	15
Bogs	Raised and blanket bogs (D1)	5-10 ##	10
Montane	Moss and lichen dominated mountain summits (E4.2)	5-10 #	7
Unmanaged woodland	Broadleaved woodland (effects on ground flora)	10-15 #	12
Broadleaved woodland (Atlantic oak woods)	Broadleaved woodland (effects on epiphytic lichens and algae)	10-15 (#)	10
Broadleaved woodland (managed)	Broadleaved woodland	Mass balance approach applied	
Coniferous woodland (managed)	Coniferous woodland	Mass balance approach applied	
Supralittoral sediment	Shifting coastal dunes (B1.3)	10-20 #	15
	Stable dune grassland (B1.4)	10-20 #	15

[1] The broadleaved, mixed and yew woodland broad habitat is separated into the following classes for the purposes of mapping nutrient nitrogen critical loads: broadleaved woodland (managed); broadleaved woodland (Atlantic oak woods); and unmanaged (ancient and semi-natural) coniferous and broadleaved woodland (excluding Atlantic oak woods), abbreviated to 'unmanaged woodland' above.

[2] The reliability of the recommended range of critical load values is indicated as:

(#) expert judgement;
quite reliable;
reliable.

6.4 Regional/national assessments

Regional assessments rely on national data to maintain consistency across the entire region. The process of performing a regional or national uncertainty assessment and aggregating the uncertainty information is set out in Figure 6.4.1. The assessment consists of the following stages:

a) If the national scale uncertainty analysis has previously been determined for the region/habitat, proceed to stage (d).
b) Obtain national input data, such as critical loads, deposition, uncertainty ranges and correlations specific to the habitat (see Section 3.1.1).
c) Perform uncertainty analysis for all one km grid squares within the region using a Monte Carlo simulation method; ArcInfo Arc Macro Language (AML) provides a convenient platform to perform the calculations at this scale and resolution.
d) Calculate the probability of exceedance for every one km grid square (see Sections 3.1.3 and 7.2).
e) Compile a one km database of the probability of exceedance and associated habitat areas. Map the probability of exceedance (see Figure 3.1.2) to visualise the risk of exceedance spatially.
f) Generate summary statistics from the one km database, for example, the area of habitat falling within different classes of exceedance probability (see legend to Figure 3.1.2). These statistics are useful for comparing the results of different scenarios, such as the probability of exceedance based on current deposition and on a future forecast of deposition.

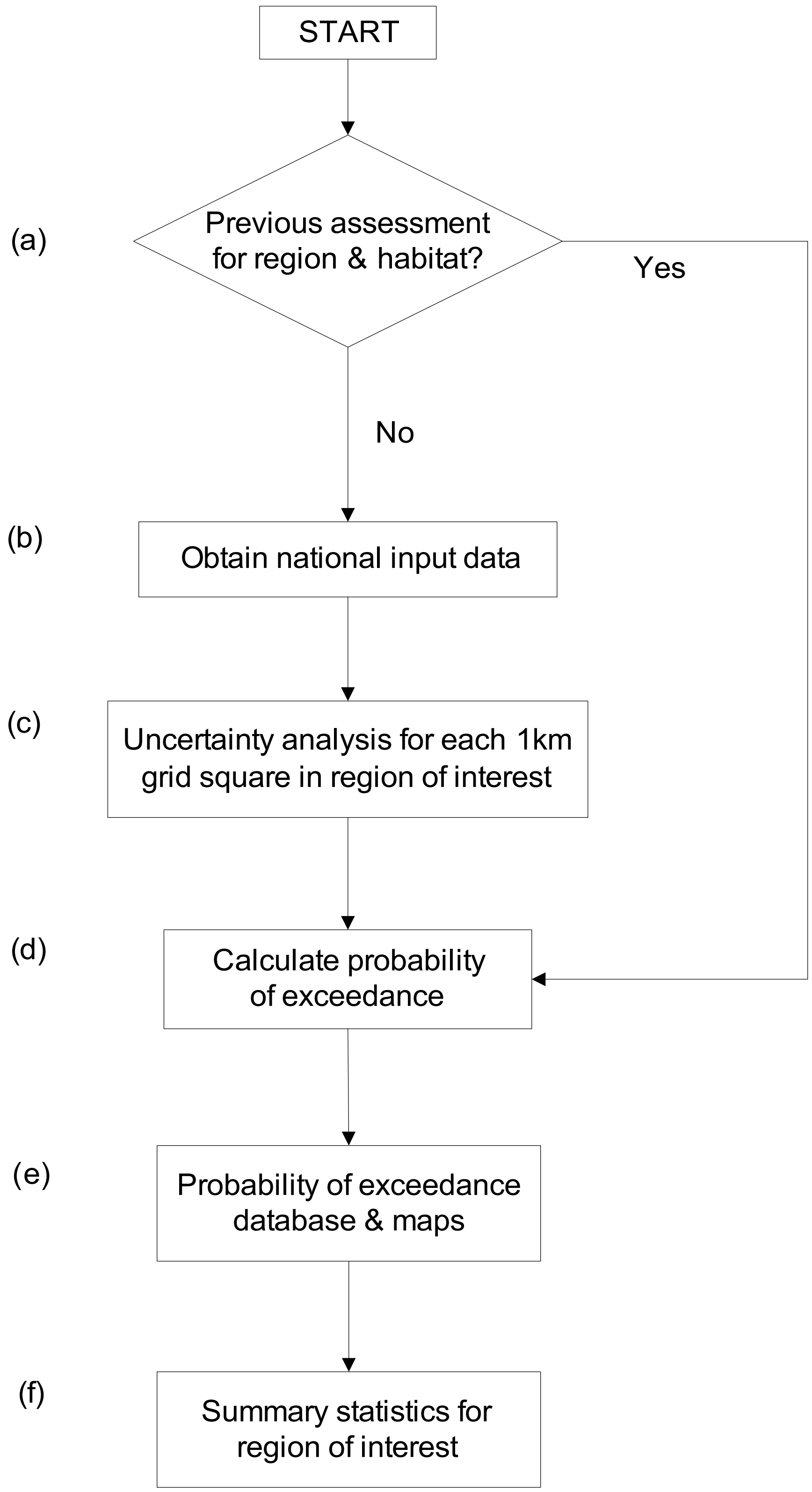

Figure 6.4.1: Framework for regional/national assessment

6.5 Discussion and conclusions

This study has focused on uncertainties in the calculations of critical loads and exceedances. It should be remembered that the critical loads maps and data presented are based on empirical or steady-state mass balance methods, used to define long-term critical loads for systems at steady state. Therefore, exceedance of these critical loads is an indication of the potential for harmful effects to systems at steady state. The challenge then is how to relate the probability of exceedance to the probability of damage. It is not possible to validate these steady state exceedances in the field unless we know the system is at steady state, and even then we may not know what indications to look for. Common Standards Monitoring in England identifies only seven per cent of SSSIs (by area) in unfavourable condition that have air pollution cited as the reason for their adverse condition. Other factors (such as stresses from climate, species competition, disease) may simultaneously be contributing to harmful effects. However, the interpretation of damage, whether caused by air pollution or other factors, is beyond the scope of this contract. This work deals only with the potential impacts from atmospheric pollution, their uncertainties and how the policymaker can use this information.

Critical load and exceedance assessments are carried out to answer a number of different questions. Depending on the type of question being asked, analysis may be required for a single point, a small site, a region or the whole country. Consistent detailed data are not available everywhere, but rather than resorting to a lowest common denominator approach, we have proposed a number of different methods depending on the spatial scale. These methods are represented by the flow charts above. The drawback of advocating different approaches at different scales is that the models are of necessity only loosely coupled; the advantage is that the best use is made of any existing data. It would be useful to be able to trial the alternative approaches within an active decision-making context, to confirm their utility.

7 Presentation of uncertainties in critical loads exceedances

Summary

- This section describes methods for presenting uncertainty in critical loads exceedances. The suitability of each method depends in part on the scale of presentation.
- Cumulative frequency charts of the exceedance distribution are a useful way of presenting uncertainty for a specified habitat for a 1x1 km grid square.
- Cumulative frequency charts of the area exceeded are useful for summarising results at the UK and European scales.
- Probability of exceedance maps provide a spatial means of communicating uncertainty at 1x1 km resolution for the whole of the UK, or for specific regions.

Deterministic methods of presenting critical load exceedance, which treat the critical load concept as a set criterion, are well established. Spatially explicit uncertainties in critical load exceedance estimates are now available and new ways of presenting information on critical load exceedance are possible. Previous studies have attempted to present this uncertainty in a number of different ways. Gascoigne and Wadsworth (1999) presented best and worst-case exceedance maps for grid squares across Wales, which were exceeded under low and high deposition scenarios respectively. Suutari *et al.* (2001) presented a series of maps depicting the percentage of protected ecosystems in Europe using 5, 50 and 95 percent probabilities. Barkman *et al.* (1999) calculated and mapped the probability of exceedance in all grid squares in the Svalöv municipality in Sweden. Syri *et al*. (2000) showed probabilities associated with different estimates of area exceeded for Finland. Similar approaches have been applied in this report to explore the most appropriate method for representing uncertainty within the exceedance calculation.

The aims of this section are to:

- summarise different methods for the communication of uncertainties in critical load exceedances;
- demonstrate the advantages and disadvantages of each method and suggest under which circumstances each method may be adopted.

7.1 Deterministic exceedance

When exceedance values are calculated deterministically (uncertainty in input data is not taken into account), they are generally conveyed to the policy maker as summary statistics, or in map form, as:

1. Exceedance of critical loads for individual habitats at one km resolution.
2. Exceedance of percentile critical loads; these combine critical loads for different habitats to protect a specified percentage of the total habitat area, for example 95 per cent. This method can also be used to aggregate the data to a coarser resolution.

The policy maker often finds it easier to have a single exceedance map, rather than one for each habitat, to summarise the exceedance information. Hence the data can be used to produce an acidity critical load exceedance map of the UK, which combines all the habitat information into one map at any desired resolution using a specified statistic (for example, minimum, 5th

percentile). Other parameters can be calculated to convey exceedance information to the policy maker, including:

- the area of sensitive habitats for which the critical load is exceeded;
- the accumulated exceedance (AE) which integrates both the area of habitat exceeded and the magnitude of the exceedance: AE(eq yr^{-1}) = ∑ exceedance (keq ha^{-1} yr^{-1}) x exceeded area (ha);
- where exceedance for non-exceeded areas is taken as zero;
- the average accumulated exceedance (AAE), which is the AE normalised by area.

Exceedance results have generally been required by UK policy makers in terms of area exceeded and AE. This information is presented in two ways: a table of statistics and maps.

1. Statistics of exceeded area or AE can be derived for each habitat separately and for all habitats combined and summed across grid squares to give regional or national statistics as required.
2. Maps of area exceeded and AE are typically derived for all habitats combined and aggregated to the same resolution as the deposition data.

7.2 Methods for presenting uncertainties in exceedances

This section describes the methods used in this project to present the results of exceedance calculations when uncertainty is incorporated. Some of the methods below have also been applied in a case study for Wales, carried out by Heywood *et al.* (2006b).

7.2.1 Frequency chart and statistical measures

This method has been used in Section 3 (Figure 3.1.1(a)) to present the probability of exceedance distribution. The frequency chart shows the number, or frequency of values, occurring in a given exceedance interval (bin). Statistical measures (Table 4.2.1) such as mean, standard deviation and percentile information are descriptive statistics of the probability distribution.

7.2.2 Cumulative frequency chart and inverse cumulative frequency chart

Cumulative frequency charts (Figure 3.1.1(b)) show the number or percentage of values less than or equal to a given amount. Inverse cumulative frequency charts (Figure 3.1.1(c)) show the number or percentage of values greater than or equal to a given amount.

7.2.3 Percentiles of predicted exceedance

The 95th percentile estimates of exceedance were used to map the exceedance (positive or negative) for each one km grid square in the South East of England for an individual habitat (for example, Figure 5.3.3). Four different statistics were used to compare the model predictions (see tables within Sections 5.3.3 and 5.3.4). These were:

1. The number of one km square coniferous forest areas where the median of predicted exceedances was more than zero.
2. The number of one km square coniferous forest areas where the 95th percentile of predicted exceedances was more than zero.

3. The median of the estimates of the number of squares where deposition exceeds critical load.
4. The 95th percentile of estimates of the number of squares where deposition exceeds critical load.

Statistics 1 and 2 were obtained by calculating the 95th percentile or median of the predicted exceedance, then counting the number of sites where the specified percentile exceeds zero. Statistics 3 and 4 were obtained by counting the number of sites where the predicted exceedance exceeds zero for each iteration of the Monte Carlo simulation, and then calculating the 95th percentile or median of the number of sites over all iterations.

7.2.4 Probability of exceedance

In summary, the probability of exceedance is the probability of achieving exceedance values above zero. The resulting probability values for each one km grid square can be classified and mapped to give a spatial representation of which grid squares have a high/low probability of exceedance for the whole of the UK (Figure 3.1.2). The legend to Figure 3.1.2 also includes the area of coniferous woodland in each exceedance probability class.

7.2.5 Cumulative frequency chart of habitat exceedance

The cumulative frequency chart of habitat exceedance was derived by calculating the area exceeded within various levels of exceedance probability and can be presented in tabular form (the legend of Figure 3.1.2) or graphical form (Figure 3.1.3). Figure 3.1.3 gives the percentage of coniferous woodland in the UK that will be exceeded with a probability of x per cent or less.

7.3 Discussion

A deterministic critical load assessment is based on the principle that the deposition of sulphur, oxidised and reduced nitrogen deposition should not exceed the critical load. Several methods of displaying uncertainty in critical load exceedances to both scientists and policy makers have been presented. No single method is recommended, but a choice should be made depending on a number of factors including:

- ease and speed of producing the statistics/maps;
- ease of interpretation of the data/maps;
- whether spatial or non-spatial information is required;
- the size of area to be mapped;
- whether it is important to the decision maker that the shape of the probability distribution function, or at least some of its characteristics be retained.

Frequency charts condense much information into a small space. It is possible to display the same information in several different ways, both using cumulative frequency charts and inverse cumulative frequency charts. The advantage of displaying these charts for individual one km grid squares is that it gives the decision maker information on the entire shape of the exceedance distribution, enabling them to read off the probability of any exceedance value (positive or negative). The disadvantage is that the decision maker may find the amount of information overwhelming and difficult to interpret for many one km grid squares.

To overcome the problem of situations where more than a few grid squares need to be presented, colour-coded maps have been used to communicate the level of risk of exceedance of critical loads for sulphur and nitrogen at specific geographical areas. The percentile of predicted exceedance maps used to present the exceedance data generated from the deposition and critical load uncertainty data are similar to the methods used to present the deterministic data, and hence have the advantage that the decision maker is already familiar with the format. However, this could also be seen as a disadvantage since it may not be obvious to the decision maker that they are reading different information. Statistics 1 and 2 (see section 7.2.3) provide a measure of the likelihood of exceedance for individual one km squares and are useful in assessing the extent of exceedance at individual sites. Statistics 3 and 4 (see section 7.2.3) provide measures of the overall sensitivity of the region to acid deposition and are useful in assessing the extent of exceedance over the whole region. Disadvantages of these statistics are that they give no information on the area of habitat exceeded, and they use the concept of an absolute criterion, where the critical load is either exceeded or not exceeded at set percentiles.

Probability of exceedance maps can be also be used to represent risk of exceedance at one km resolution (Figure 5.3.5 and Figure 5.3.6 and Section 3.1.2). However, this method retains the probabilistic nature of the exceedance information. The legend in Figure 3.1.2 shows the area that falls into each of the probability classes. At the national scale, these maps can become hard to interpret and only general spatial trends can be recognised.

The data can be aggregated by country or county using the cumulative frequency of habitat exceedance charts (Figure 3.1.3). The advantage of this method is its relative simplicity, in that it summarises both the area and probability of exceedance information for the whole of the UK, but in doing so it loses any spatial element. This method provides national area statistics for any desired range of probability of exceedance and as such could be very useful for making national policy decisions, for example on the potential benefits of different emission abatement strategies.

To summarise, we suggest that:

- the cumulative frequency chart is particularly suited to presenting uncertainty information for a one km grid square for a specific habitat;
- percentile of predicted exceedance statistics and maps are used for specific percentile exceedance scenarios, for example 50^{th} and 95^{th} percentiles.
- probability of exceedance maps provide one km resolution risk of exceedance information spatially;
- cumulative frequency of habitat exceedance charts summarise the area within ranges of probability of exceedance classes and are suitable for large areas, such as the UK.

7.4 Conclusions

By using exceedance distributions, absolute criteria are avoided. Presenting uncertainties in critical load exceedances helps identify areas with different levels of risk for harmful effects and allows the policy maker to decide on acceptable levels of risk, such as a 50 per cent or five per cent probability of exceedance.

Each approach of presenting uncertainty explored in this report is applicable at different scales. The area cumulative frequency chart can be used at the UK and European scales for national and international policy development. Probability of exceedance maps are useful at the regional scale. Exceedance of specified percentile statistics are useful in assessing the extent of exceedance for the whole region or at individual sites. Cumulative frequency charts of one km data are more appropriate for local assessments.

8 Source sector contributions to exceedance

Summary

- The TRACK model was used to calculate the contribution of various source sectors to critical load exceedance at Liphook. Monte Carlo analysis was used to generate cumulative distribution functions for exceedance with and without each sector contribution.
- In this case study, there was a 41 per cent probability of exceedance given 2002 deposition. Excluding power stations and oil refineries made a negligible difference. Excluding all large point sources reduced the exceedance probability to 30 per cent, and excluding local ammonia sources to 35 per cent.
- This case study demonstrates the methodology: the proportions attributable to each source sector are likely to vary across the country.

8.1 Liphook case study

The aim of this task was to make recommendations on the source sectors requiring the greatest attention in critical load assessment. A limited number of TRACK model runs were conducted to ascertain the contribution of various Environment Agency-controlled source sectors to a sensitive coniferous forest receptor at Liphook. Predicted deposition rates were compared with the critical loads for coniferous forest.

The TRACK model was used to predict sulphur and nitrogen deposition rates at a sensitive receptor site in the coniferous forest area near Liphook in Hampshire (GR 4805 1295). The following model runs were carried out:

- baseline for 2002 including all emission sources;
- baseline excluding all UK large point sources;
- baseline excluding power stations;
- baseline excluding refineries;
- baseline excluding local ammonia emissions.

A Monte Carlo simulation of the exceedances of the critical loads was then carried out for each case, based on the statistical parameterisation of model errors and critical load uncertainty as described in Section 5. One thousand iterations of the Monte Carlo simulation were done in each case. A Liphook type probability distribution for the critical loads was assumed to apply. Figure 8.1.1 shows the cumulative distributions for the calculated exceedances.

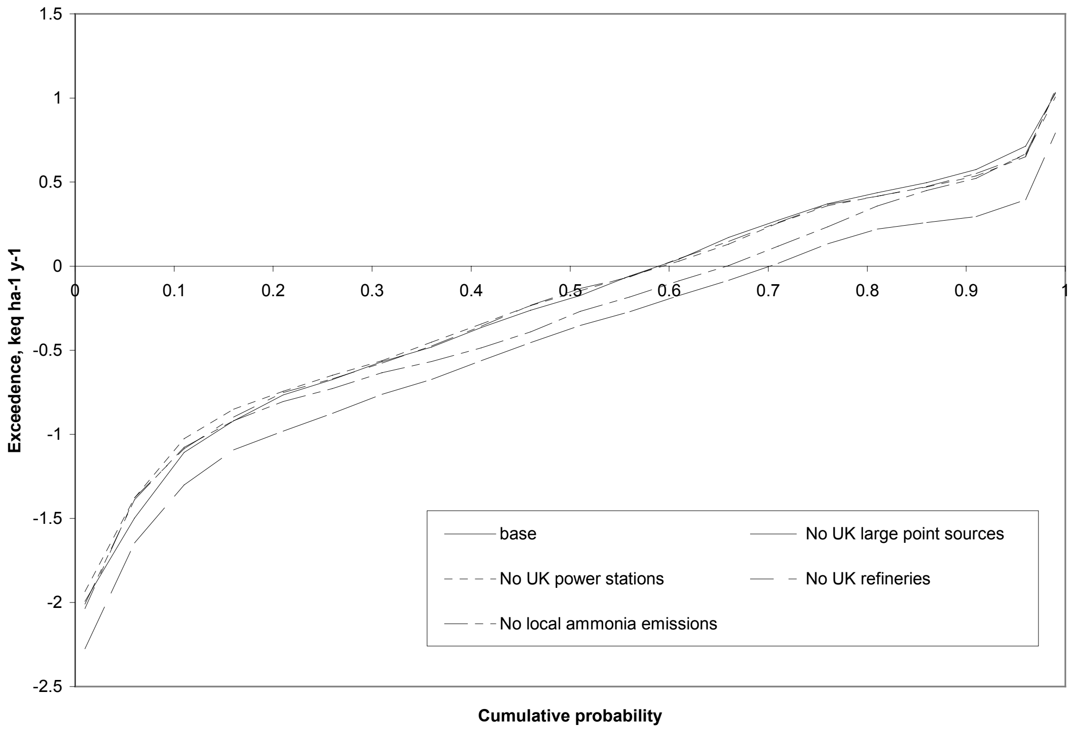

Figure 8.1.1: Cumulative probability distributions of predicted exceedances of coniferous forest critical load for 2002.

The baseline model run for 2002 with all source sectors showed that the predicted deposition rate was slightly less than the critical load. However, the Monte Carlo simulation indicated that there was a 41 per cent chance of exceedance.

Excluding the contributions from power stations only and refineries only leads to a negligible reduction in deposition at the Liphook site. Excluding all UK large point sources reduces the chance of exceedance to 30 per cent. Excluding the contribution from local agricultural emissions of ammonia reduces the chance of exceedance from the baseline 41 per cent to 35 per cent . It is concluded that reducing the emissions from power stations and refineries alone will not reduce the chance of exceedance of the critical load for coniferous forest at Liphook. Reducing the emissions generally from all Environment Agency-regulated sources would have some effect on reducing the chance of exceedance: the effect is comparable with that associated with local ammonia emissions from agriculture.

8.2 Conclusion

This case study shows that power stations and refineries make a negligible contribution to critical load exceedance at Liphook. The contribution of local agricultural sources and Environment Agency-regulated sources in general are comparable, but the maximum effect is to reduce the chance of exceedance from 41 per cent to 30 per cent. These conclusions are, however, site-specific. Repeating the analysis elsewhere in the country may lead to different conclusions, depending on the proximity and strength of the different source sectors. It may be possible to produce a map showing the relative contributions of each source sector in different places. With more difficulty, this could be developed into a tool to calculate the reduction in exceedance probability which would result from action on any given source sector.

9 Suggestions for further work

Site-specific uncertainty

More examples of critical load methods which are under-represented in this work should be run, such as acidity critical loads for heathlands and unmanaged forests, and nutrient nitrogen critical loads for managed forests. The work could be extended to broad habitats, which use the same methods (broadleaved woodland, acid grassland and calcareous grassland), though these would not be different in principle. Some consideration could be given to the habitats which use different critical load methods which are less amenable to error propagation uncertainty analysis, because the uncertainty is largely in model definition or habitat definition (such as peatlands). Work on more sites would produce a more complete set of probability distributions for critical loads and enable the investigation of appropriate distributions to use in regional applications such as those described in Section 5.

Comparison of site-specific and national data

Further sites need to be analysed before any general conclusions can be drawn on the likely effects of using national or site-specific data on critical loads or critical load exceedance.

National scale uncertainty

The national scale analysis presented in this report only applies to critical loads for managed coniferous woodland and exceedances based on mean deposition data for 1999-2001. Data for other habitat types and other years may be calculated and presented using the methods described.

To provide risk information for different deposition scenarios the uncertainty analysis would need to be redone, since Skeffington *et al*. (2006) and other work in this report shows that a reduction in deposition over time reduces uncertainties in deposition and makes the critical load uncertainties relatively more important.

Practical regulation

It would be very useful to work through some examples of critical load and exceedance assessments using real situations. Such situations might be an assessment of a designated site, for example. This would test the real world applicability of the methods proposed in the earlier sections, and enable further progress towards a practical methodology.

10 Overall conclusions

The overall aim of the project was “to examine the uncertainties in critical load assessments and develop a practical methodology for such assessments within the Environment Agency’s regulatory role”. We have undertaken an extensive examination of uncertainties in critical loads and exceedances in the UK, and in Section 6 proposed some frameworks for critical load and exceedance assessment at various spatial scales. Since the Environment Agency’s regulatory functions are diverse and dissimilar, however, it seems worthwhile in this section to consider the project as a whole and its implications for the Environment Agency’s activities.

Does uncertainty matter?

In its original form, the critical load is a ‘hard’ concept – a threshold deposition below which harmful effects do not occur, and above which they do. Uncertainty is not considered. This corresponds well with traditional environmental standard setting, where a single value can be set for an environmental limit which is designed to protect some aspect of the environment. This approach has the advantage of being relatively easy to apply, it appears authoritative and objective, and it is easy to communicate to policy makers. Acknowledging that the critical load is in fact uncertain reduces the authority of the concept, makes it harder to explain and apply, and requires difficult decisions and value judgements to be made.

So what are the advantages of introducing uncertainty? Firstly, it is more honest. Critical loads, like other environmental quantities, are intrinsically uncertain. Environmental scientists have a professional responsibility to communicate the degree of confidence they have in the environmental effects they predict and the standards they propose; the most objective way to assess this degree of confidence is by an uncertainty analysis, such as the one carried out for this report. Secondly, it should increase the quality (fitness for purpose) and accuracy of the proposed critical loads. Critical loads, like other standards, often have a precautionary element built in, but without an explicit acknowledgement and calculation of uncertainty, it is impossible to assess how precautionary a given limit is. Even if the critical load is treated ultimately as a simple limit value, uncertainty analysis should provide a better estimate of its magnitude. Thirdly, the assessment of uncertainty, and in particular sensitivity analysis, is essential in evaluating and improving the models which lie behind critical load calculations. It can indicate which parameters would be the best targets for more accurate measurement, whether the inevitable model simplifications are reasonable representations of reality, and can help evaluate whether model additions are worthwhile. Thus, it can lead to more robust models. Fourthly, introducing uncertainty can help identify and avoid potential surprises due to inadequacies in data and model structures. For these reasons, it seems worthwhile to try to evaluate critical load uncertainties. More detailed discussion of these points can be found in the literature (Barkman, 1997; Skeffington, 2006).

How to conduct an uncertainty analysis of critical loads

Our investigation of critical load uncertainties explored to some extent the best ways to characterise input parameters for the critical load and exceedance models; that is, what values, ranges, statistical distributions and intercorrelations are most appropriate. In most cases these can be derived only by expert judgement, though we have tried to use objective methods wherever possible. Our investigations showed for a limited set of examples that inclusion of certain features made a significant difference to the results (such as intercorrelations between deposition parameters), whereas others were of limited importance (statistical distribution of catchment parameters) and some of no importance at all. The characteristics we chose are set out in the report and its appendices, and can serve as a guide for future assessments, since the frameworks described in Section 6 require such assessments in some situations.

Other sources of uncertainty

Uncertainty analysis as used in this report only evaluates uncertainties in the input parameters and how these propagate through the models. Other forms of uncertainty are harder (or impossible) to quantify even when they can be identified, such as the uncertainties in mapping broad habitats mentioned in Section 3. Unknown processes may be involved, though these cannot of course be incorporated into the models (epistemological uncertainty) and there may be systematic errors of various sorts and approximation uncertainties due to model simplifications. All these must be borne in mind in specific applications, because they can sometimes be taken into account even if they cannot be quantified.

How uncertain are critical loads?

A meta-analysis of the entire set of critical load data (Section 2.3) showed coefficients of variation (CVs) generally between 5% and 60%, with different values applicable to different nodes on the critical load function: 30-60% for *CL(A)*, 25-50% for $CL_{max}S$, 10-35% for $CL_{max}N$, 5-25% for $CL_{min}N$, and (based on a very limited sample) about 15% for $CL_{nut}N$. To those familiar with our knowledge of the underlying processes, these values seem remarkably small, especially given that individual input parameters typically have wider limits. The 'compensation of errors' phenomenon reduces the calculated uncertainties of the critical load models, as described in the text. Environmental regulators may in contrast feel that these uncertainties are uncomfortably large. For instance, at no site can we be 100% confident of either exceedance or non-exceedance. Some suggestions as to how to deal with this level of uncertainty are given in the following sections. There is substantial variation in uncertainty between sites and types of site, but these CVs could be used as a rough guide as to the uncertainty in individual cases in the absence of other information. However, more data are really required. The level of uncertainty indicated does, however, imply that it is worthwhile continuing to use critical loads for environmental assessment.

Which input parameters have most influence on uncertainty?

If certain input parameters were consistently important for uncertainty, this would suggest that research into the values of these parameters would be most valuable in reducing it. Unfortunately, for coniferous forests at least, it appears that almost any parameter can be important depending on site characteristics. There are indications that there may be classes of site; for instance, for low weathering rate sites, uncertainty in calcium deposition appears likely to be important for critical load estimation. This illustrates also that deposition of calcium (or base cations) can be important for critical load as well as exceedance calculations. For aquatic ecosystems, one parameter, the present-day non-marine base cation concentration, is most important for uncertainty. This implies that a better estimate of this parameter will reduce the uncertainty in the critical load estimates. This could be obtained by more sampling; many UK aquatic critical loads are based on a single sample taken over 10 years ago. However, it also implies that there are limits to this process, as such concentrations are intrinsically variable.

How to deal with uncertain critical loads

Instead of a single value for a critical load or exceedance, the product of an uncertainty analysis is a probability distribution. This can be used in a number of ways, but some decisions have to be made in order to do so. For a given site, the uncertainty analysis may indicate that with current deposition, the probability that a critical load is exceeded is (say) 62 per cent. A statistical distribution showing probability of exceedance versus deposition can be used to read off the probability for any given deposition level, so for this site it might indicate that a reduction in deposition of 200 eq ha^{-1} yr^{-1} is required to reduce the probability of exceedance to 50 per cent, and of 800 eq ha^{-1} yr^{-1} to reduce it to 10 per cent. This raises the question: what probability of exceedance is considered acceptable?

It might be tempting to use the conventional probabilities used in statistical significance testing (where the probability must be less than five or one per cent to be considered acceptable), but this should only be done with due consideration. There is nothing sacred about the conventional significance levels. They were proposed by R. A. Fisher in the early twentieth century for analysing agricultural experiments. The appropriate probability level should be *a decision by the regulator*, which raises the question of how the level should be chosen. In Section 6 we suggest that probability level should be related to the "cost" of an inappropriate decision (where a "cost" can be financial, such as preventing a development or requiring ameliorating activities, or environmental, such as the loss of a particular species or habitat). This may make those who have to choose these levels somewhat uncomfortable, but it means that the choices are explicit and transparent, whereas using non-probabilistic methods they tend to be implicit and opaque. The width of the uncertainties of critical loads and exceedances revealed by this report implies that use of excessively precautionary criteria will be very expensive, requiring large reductions in deposition with concomitant costs.

The importance of spatial scale

Critical loads are calculated for several habitats on each of over a quarter of a million kilometre squares covering the UK. Measured data are available for only a tiny proportion of these habitats; the rest must rely on modelled data which depend largely on accurate identification of the physical and biological characteristics of each mapped habitat, and accurate estimates of the deposition of acidifying and basic substances. Section 4 showed that using national data for sites for which measured data are available can lead to very different results to those which use the measurements. This does not mean that the national default data are wrong, just that they were generated for national-scale assessment and are thought adequate for that purpose. We show in Section 3 that applying the national critical loads data to individual sites can give rise to anomalous values, particularly for larger sites with variable soils and habitats. Similarly, the habitat critical load maps provide a national picture of their distribution and sensitivity to acidification and eutrophication, appropriate for assessments at the national scale. They may be inappropriate for site-specific assessments. The procedures for critical load assessment should therefore differ depending on scale. In Section 6 we provide a set of procedures for critical load assessment at the local, regional and national scales.

How to present uncertainties

There are numerous ways of presenting critical load and exceedance uncertainties. Which is chosen will depend on the purpose of the presentation, the sophistication of the user, and the amount of information which is desired or is possible to communicate. It should be clear to the viewer what sort of information they are seeing. Section 7 makes some suggestions for presenting exceedance uncertainties at different spatial scales. The cumulative frequency chart loses least information and is particularly suited to presenting uncertainty information at large scales, for example for a one km grid square for a specific habitat. Maps and statistics showing the value of a given percentile of predicted exceedance can be used at regional or local scales. Probability of exceedance maps provide spatial information at one km resolution about risk of exceedance. Cumulative frequency of habitat exceedance charts summarise the area within ranges of probability of exceedance classes and are suitable for large areas, such as regions or the UK, but provide no spatial information. Other presentations may also be appropriate in other circumstances.

How to use uncertainties in regulation

Section 6 provides flow charts and descriptions to guide critical load, exceedance and uncertainty assessments at the local (point, site), regional and national scales. A point is a single habitat in a single location, whereas a site will typically be a designated site (a SSSI or Natura 2000 site) which may occupy quite a wide area and consist of several habitats. The flow charts suggest a series of actions depending on the circumstances, and should be consulted for details. One clear message from the charts is that if the uncertainty is considered too large, it can be reduced by using site-specific data. These may be data that already exist, and include checking that the critical load models are appropriately parameterised, for example, that a designated feature of a site (such a certain habitat) is present on the soil type the critical load model is assuming. In the absence of existing data, new data can be obtained if it is considered that the cost and importance of the site warrants it. Data from well-studied sites described in this report show clearly that uncertainties can be reduced by collecting more data, but it should be understood that this could take several years and that many of the quantities required are intrinsically difficult to measure.

In general, the following scheme might be adopted:

- Identify a probability of exceedance below which the risk of significant harmful effects is acceptably low, and therefore further emission reductions are not required.
- Identify a probability of exceedance above which the risk of significant harmful effect is unacceptably high, and therefore further emission reductions are required.
- For probabilities of exceedance between these two extremes,a precautionary approach could be adopted, in which case the deposition would be reduced depending on the cost-effectiveness of the measure. Alternatively, further study could be recommended on those parameters that dominate the overall uncertainty.

How to determine the influence of different emission sources

Section 8 describes a case study showing how separate modelling of different source sectors can be used to assess their influence on critical load exceedance at a given point. In principle this could be applied everywhere in the country, though this would be rather laborious. There are liable to be spatial patterns due to the uneven distribution of different types of source, but the method could be used at any given site.

To conclude

Overall, we believe this report gives an indication of the uncertainties likely to be found in the most widely used critical load assessment models. Some models are better covered than others, however, and more data would be valuable. We also believe we have shown how the Environment Agency could use uncertainty assessment to improve the quality of its regulation of emission sources. The uncertainty analyses contained in this report should give an indication of the reliability of critical load models, and the weight which should be put on them during the Environment Agency's regulatory activities.

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List of abbreviations

ANC	acid neutralising capacity
CV	coefficient of variation: standard deviation divided by the mean, expressed as a percentage
FAB	first order acid balance
SD	standard deviation
SMB	simple mass balance
SSMB	steady state mass balance

Critical loads function parameters – see Appendix A1.2

$CL_{max}S$	critical load for acid deposition when nitrogen deposition is zero
$CL_{max}N$	critical load for acid deposition when sulphur deposition is zero
$CL_{min}N$	critical load of acidity due to nitrogen removal processes alone
$CL(A)$	critical load for acidity (taking into account base cation uptake and deposition)

Critical load equation parameters:

*	refers to the non-marine contribution
dep	deposition of the substance in question
S_{dep}	sulphur deposition
$NO_3^-{}_{dep}$	nitrate deposition
$NH_4^+{}_{dep}$	ammonium deposition
N_{dep}	$NO_3^-{}_{dep} + NH_4^+{}_{dep}$
BC	base cations ($Na^+ + K^+ + Ca^{2+} + Mg^{2+}$)
Bc	originally, base cations other than Na^+. In the UK implementation of the equations, however, Bc is equivalent simply to Ca^{2+}
BC_{dep}	base cation ($Na^+ + K^+ + Ca^{2+} + Mg^{2+}$) deposition
ANC_w	acid neutralising capacity generated by weathering
Ca_w	calcium released from soil minerals by weathering
Ca_{corr}	proportion of ANC_w which is Ca weathering
BC_u	base cation uptake into plants
Ca_u	uptake of calcium into plants
Q	run-off or effective rainfall
$[BC]_l$	limiting concentration below which plants are considered to be unable to take up base cations
Ca/Al_{crit}	critical ratio of Ca to Al in the soil solution, defining the damage threshold
K_{gibb}	gibbsite equilibrium constant, defining the relationship between H^+ and Al concentrations in soil solution
N_i	immobilisation of nitrogen (into soil organic matter)
N_u	uptake of nitrogen into plants
N_{de}	denitrification (conversion of inorganic N into N_2 or N_2O)

Critical load parameters specific to FAB:

f	fraction of catchment area which is forested
r	ratio of lake area to total (lake + catchment area)
N_{ret}	in-lake retention of N
S_{ret}	in-lake retention of S
BC_{le}	leaching of base cations ($Na^+ + K^+ + Ca^{2+} + Mg^{2+}$)
ANC_{le}	acid neutralising capacity leaching

Appendices

Appendix A – Methods Section 2

Appendix A contains details and methods for Section 2 of the main report site-specific uncertainty.

1.1 Monte Carlo analysis

Monte Carlo analysis is a well-established method for assessing the effects of uncertainty in model parameters on model outputs. Parameter values are sampled from known or assumed distributions, and the model is run repeatedly to generate a distribution of output values. For the applications described in this report, the critical load model being used (see below) was run 5,000 times using a commercial software package, Crystal Ball (Decisioneering UK Ltd), which operates as an add-on to Microsoft Excel. It uses pseudorandom numbers produced by a multiplicative congruential generator with a repeat period of 2,147,483,646 to randomly sample input distributions. The program allows the covariance structure of the inputs to be taken into account. The results of the Monte Carlo analysis depend critically on the means, ranges and distribution types chosen for each parameter. Attempts were made to produce the best possible estimates given knowledge of the Liphook site as described in the report.

Our Monte Carlo analysis included a sensitivity analysis, in which the sensitivity of each output parameter to variation in the input parameters is assessed. The sensitivity analysis is based on rank correlation coefficients between individual input parameters and the given output parameter. A high value for the rank correlation coefficient for an input parameter implies a high sensitivity of the output to that parameter. The use of rank rather than parametric correlations allows the analysis of non-linear equations provided the relationship is monotonic (which it is). As it is easier to interpret, data are expressed as a percentage contribution to variance by squaring the original rank correlations and normalising to 100%. The technique of using rank correlations is well-established and described in textbooks such as Saltelli *et al.* (2000).

1.2 The critical load function

Deposition of both sulphur and nitrogen compounds can contribute to exceedance of the acidity critical load. The critical load function was developed under the UNECE Convention on the Long Range Transport of Air Pollutants and 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 exceed the critical load, while all areas below or on the line represent an 'envelope of protection' where critical loads are not exceeded (Figure A1).

$CL_{max}S$ represents the critical load for acidity when nitrogen deposition is zero. Conversely, $CL_{max}N$ is the critical load for acidity when S deposition is zero. $CL_{min}N$ is the deposition-independent critical load of acidity due to nitrogen removal processes alone (nitrogen uptake, immobilization, and denitrification). The critical load is equal to $CL_{max}S$ when all nitrogen deposition is taken up by the catchment, hence the horizontal portion of the critical load function. Not shown in Figure A1 is the critical load for nutrient nitrogen, which may be any value on the N-axis greater than or equal to $CL_{min}N$. This represents the eutrophying, as opposed to acidifying, effects of nitrogen.

Formulae for calculating these quantities are given in the explanations of the various critical load models below.

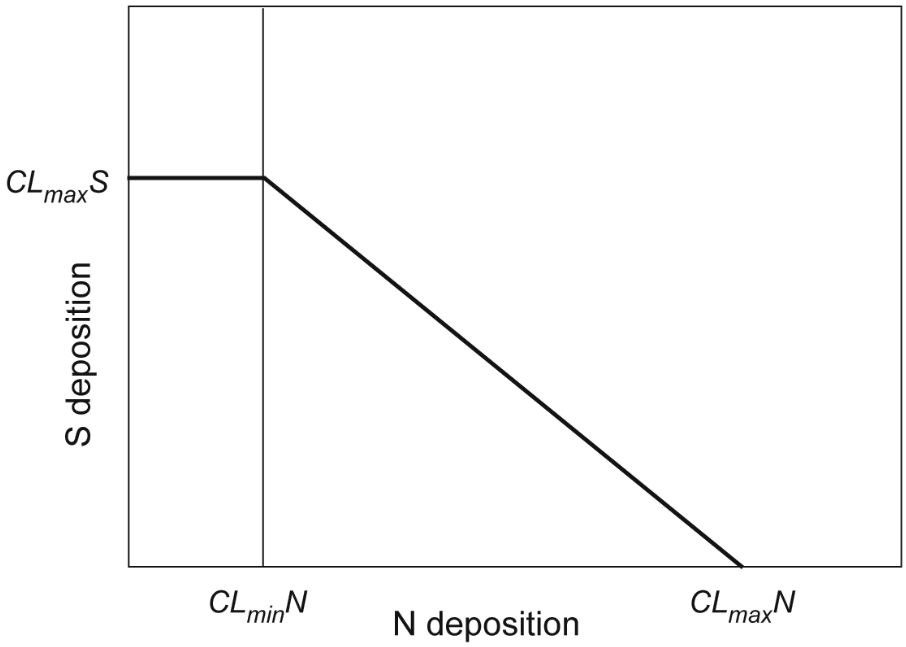

Figure A1: The critical load function

1.3 SSMB model

1.3.1 Detailed model description

The steady state mass balance (SSMB) model is the most commonly used model in Europe for the calculation of acidity critical loads for woodland ecosystems. The model is based on balancing the acidic inputs to and outputs from a system, to derive a critical load that ensures a critical chemical limit (related to effects on the ecosystem) is not exceeded (Sverdrup and De Vries, 1994). The equation has been derived from a charge balance of ions in leaching fluxes from the soil compartment, combined with mass balance equations for the inputs, sinks, sources and outputs of sulphur and nitrogen (Posch *et al.*, 1995). The SSMB equation can be written as:

$$CL_{max}(S) = BC^*_{dep} - Cl^*_{dep} + BC_w - BC_u + (1.5 \times \frac{Bc_{dep} + Bc_w - Bc_u}{(Bc/Al)_{crit}}) + Q^{2/3} \times (1.5 \times \frac{Bc_{dep} + Bc_w - Bc_u}{(Bc/Al)_{crit} \times K_{gibb}})^{1/3} \quad \text{..................} (A1)$$

where:

- concentrations and fluxes are expressed in equivalence units, such as eq ha^{-1} yr^{-1} (mol_c ha^{-1} yr^{-1});
- $CL_{max}S$ is as defined in Section A1.2;
- * refers to the non-marine contribution;

the suffixes represent:

- *dep* – deposition;
- *w* - weathering (release of base cations from soil or rock minerals);
- *u* - uptake by plants into perennial tissues;
- *BC* is the flux of base cations ($Na^+ + K^+ + Ca^{2+} + Mg^{2+}$);
- *Bc* is the flux of base cations other than Na;
- $(Bc/Al)_{crit}$ is the critical Bc /Al ratio defined by the user;
- *Q* is effective rainfall/run-off;
- K_{gibb} is the gibbsite equilibrium constant, defining the relationship between H^+ and Al^{3+} concentrations in the soil solution.

UK practice (Hall *et al.,* 2003) is to use a variant of this equation in which the *Bc* terms represent only Ca^{2+} rather than $K^+ + Ca^{2+} + Mg^{2+}$. If all three base cations are used, critical loads calculated for the UK are high, largely because the deposition of marine Mg^{2+} elevates the numerator of the equation. There is then little exceedance even in areas known to be sensitive to acidification (Reynolds, 2000). It is also assumed that plants cannot take up Ca^{2+} at concentrations below 2 μeq L^{-1}, (known as the limiting base cation concentration) and this value, converted into a flux by multiplying by *Q*, is subtracted from the numerators of the two expressions involving *Bc* in Equation A1.

The minimum critical load for nitrogen $CL_{min}N$ (Figure A1) represents the critical load of acidity due solely to nitrogen removal processes in soil:

$$CL_{min}N = N_u + N_i + N_{de} \qquad \text{(A2)}$$

where:

- N_i immobilisation of nitrogen (into soil organic matter);
- N_u uptake of nitrogen into plants;
- N_{de} denitrification (conversion of inorganic N into N_2 or N_2O);

The maximum critical load for N $CL_{max}N$ in the UK is:

$$CL_{max}N = CL_{max}S + CL_{min}N \qquad \text{(A3)}$$

The practice recommended in the Mapping Manual (UBA, 2004) is to make the denitrification rate dependent on available nitrogen, which would introduce N_{de} as a factor in Equation A3, but the UK does not follow this practice. Instead, it uses a fixed rate of denitrification dependent on soil type (Hall *et al.*, 2003) which simplifies calculations considerably. The UK also defines a critical load for acidity *CL(A),*which takes into account base cation fluxes, as:

$$CL(A) = CL_{max}S + {}^*BC_{dep} - {}^*Cl_{dep} - {}^*BC_u \qquad \text{(A4)}$$

This was also calculated and reported in this study.

Exceedance is defined as the amount by which deposition exceeds the critical load. Posch (1999) provided four equations for calculating exceedance of the critical load function, each of which applied to one of four regions in the zone of exceedance. In the UK implementation of critical loads, where the relationship $CL_{max}N = CL_{max}S + CL_{min}N$ (Equation A3) holds, this can be simplified into a unified expression depending on whether nitrogen deposition exceeds $CL_{min}N$ or not:

$$\begin{aligned} Ex &= S_{dep} - CL_{max}S && \text{where } N_{dep} < CL_{min}N \\ Ex &= N_{dep} + S_{dep} - CL_{max}N && \text{where } N_{dep} \geq CL_{min}N \end{aligned} \qquad \text{(A5)}$$

1.3.2 Input parameter sets used

Table A1: Base case - Liphook (Section 2.1.2)

Parameter	Units	Mean	Lower	Upper	SD	Distribution	Correlations
BC_w	eq $ha^{-1}yr^{-1}$	100	0	200		Rectangular	None
BC_u	eq $ha^{-1}yr^{-1}$	250	125	375		Rectangular	None
Ca_{dep}	eq $ha^{-1}yr^{-1}$	430			215	Normal	None
Ca_{corr}	unitless	0.1	0	0.2		Rectangular	None
Ca_w	eq $ha^{-1}yr^{-1}$	10				Calculated	None
Ca_u	eq $ha^{-1}yr^{-1}$	160			43.2	Normal	None
Q	m^3 $ha^{-1}yr^{-1}$	4,100			943	Normal	None
$[BC_l]$	μeq L^{-1}	2	2	2		None	None
Ca/Al_{crit}	mol mol^{-1}	1	0.5	1.5		Rectangular	None
K_{Gibb}	m^6 eq^{-2}	950	760	1140		Rectangular	None

Table A2: Effects of correlation of deposition terrestrial parameters - Liphook

Parameter	Mean	Lower	Upper	SD	Distribution	Correlations
BC_w	150	50	250	-	Rectangular	
BC_u	270	135	540	-	Rectangular	0.9, Bc_u; 0.75, N_u
Ca_{corr}	0.57	0.47	0.67		Rectangular	
Bc_w	85.5	-			Calculated	
Bc_u	125	80	320		Rectangular	0.9, *BCu*
Bc/Al_{crit}	1	0.5	1.5		Rectangular	
K_{gibb}	950	300	3,000		Rectangular	
Q	4,690	4,221	5,159		Rectangular	
N_u	500	400	600		Rectangular	0.75, *BCu*
N_i	214	107	321		Rectangular	
N_{de}	71	35.5	105		Rectangular	

Bc is equivalent to *Ca* in the UK application of the SSMB. Units are as in Table A1. See Section 2.1.5.

Table A3: Deposition parameters - Liphook

Parameter	Mean	Lower	Upper	SD	[1]Distribution	Correlations
BC_{dep}	1,965	1,367	2,543	289	TL	S_{dep}, 0.63; NO_{3dep}, 0.17; NH_{4dep}, 0.45; Cl_{dep}, 0.97.
Bc_{dep}	175	109	241	33	TL	S_{dep}, 0.76; NO_{3dep}, 0.47; Cl_{dep}, 0.51.
Cl_{dep}	1,300	820	1,780	240	TL	BC_{dep}, 0.97; Bc_{dep}, 0.51; S_{dep}, 0.56; NO_{3dep}, 0.30; NH_{4dep}, 0.52.
$*Cl_{dep}$	55	35	75	10	TL	
S_{dep}	781	437	1,125	172	TN	BC_{dep}, 0.63; Bc_{dep}, 0.76; NO_{3dep}, 0.44; Cl_{dep}, 0.56.
NO_{3dep}	392	150	634	121	TL	BC_{dep}, 0.17; Bc_{dep}, 0.47; S_{dep}, 0.44; NH_{4dep}, 0.51; Cl_{dep}, 0.30.
NH_{4dep}	513	193	833	160	TL	BC_{dep}, 0.45; NO_{3dep}, 0.51; Cl_{dep}, 0.52

Units are eq $ha^{-1}yr^{-1}$. [1]Distribution types: T indicates truncated at ± 2 standard deviations; N - normal, L - lognormal. See Section 2.1.5.

Table A4: Input parameters – comparison of distribution types

		Limits			
Parameter	**[1]Mean**	**[2]Lower**	**[3]Upper**	**[4]SD**	**[1]Correlations**
BC_w	150	50	250	50	
BC_u	270	135	405	67.5	0.9, Bc_u; 0.75, N_u
Ca_{corr}	0.57	0.47	0.67	0.05	
Bc_w	85.5	-			
Bc_u	160	80	240	40	0.9, *BCu*
Bc/Al_{crit}	1	0.5	1.5	0.25	
K_{gibb}	1,650	300	3,000	675	
Q	4,690	4,221	5,159	234.5	
N_u	500	400	600	50	0.75, *BCu*
N_i	214	107	321	53.5	
N_{de}	71	35.5	107	17.75	

Units are eq $ha^{-1}yr^{-1}$. [1]Applies to all distributions. [2]Applies to all except normal, where it is zero or a very small number in the case of Bc/Al_{crit} or K_{gibb}. [3]Applies to all except normal, where it is infinity. [4]Applies to normal and truncated normal. See Section 2.1.7.

Table A5: Terrestrial input parameters for the three site comparison

	Liphook			Thetford			Aber		
Parameter	**Mean**	**Lower**	**Upper**	**Mean**	**Lower**	**Upper**	**Mean**	**Lower**	**Upper**
BC_w	150	50	250	4,600	2,300	6,900	163	120	300
Fertiliser	-			-			177	133	221
BC_u	270	135	540	270	135	540	316	284	348
Ca_{corr}	0.57	0.47	0.67	0.90	0.85	0.95	0.092	0.042	0.142
Bc_w	85.5	-		4140			15		
Bc_u	125	80	320	160	80	240	160	80	320
Bc/Al_{crit}	1	0.5	1.5	1	0.5	1.5	1	0.5	1.5
K_{gibb}	950	300	3,000	950	300	3,000	100	10	300
Q	4,690	4,221	5,159	1,620	1,458	1,782	5,930	5,337	6,523
N_u	500	400	600	210	105	315	246	221	271
N_i	214	107	321	71	35.5	107	150	120	180
N_{de}	71	35.5	107	71	35.5	107	35.5	28	43

Units are as in Table A1. Distributions are all rectangular, and correlations are as in Table A2.

Table A6: Deposition input parameters for the three site comparison

Site	Thetford					Aber				
Parameter	**Mean**	**Lower**	**Upper**	**SD**	**[1]Dist.**	**Mean**	**Lower**	**Upper**	**SD**	**[1]Dist.**
$*BC_{dep}$	190	100	340	[2]	L	397	197	597	100	TN
Bc_{dep}	220	117	415	[2]	L	485	243	327	121	TN
Cl_{dep}	-					-				
$*Cl_{dep}$	0					0				
$*S_{dep}$	490	260	925	[2]	L	1,481	881	2,081	300	TN
NO_{3dep}	1,240	657	2,340	[2]	L	940	658	1,222	141	TN
NH_{4dep}	1,640	870	3,095	[2]	L	1,190	590	1,790	300	TN

Units are eq ha^{-1} yr^{-1}. [1]Distributions: L – lognormal, TN - normal truncated at two standard deviations. [2]For Thetford, the distributions described in Section 2.1.6 were used, and the range shown is from the 0.5^{th} to 99.5^{th} percentile; Liphook values were as in Table A3. Note that in Table A3, the BC_{dep} and S_{dep} parameters are total and not non-marine, and chloride was entered as an independent parameter, whereas in Table A6, $*BC_{dep}$ and $*S_{dep}$ are non-marine and there is no independent estimate of chloride.

Table A7: Terrestrial input parameters for critical loads for heathland

Parameter	**Mean**	**Lower**	**Upper**	**Distribution**
BC_w	497	120	980	Rectangular
BC_u	-			
Ca_{corr}	0.023	0.074		Rectangular
Bc_w	11			
Bc_u	-			
Bc/Al_{crit}	0.8	0.3	1.3	Rectangular
K_{gibb}	1,650	300	3,000	Rectangular
Q	16,000	14,400	17,600	Rectangular
N_u	-			
N_i	214	107	321	Rectangular
N_{de}	71	35.5	107	Rectangular

Units are as in Table A1. For the empirical critical load, only BC_w was used.

Table A8: Deposition input parameters for critical loads for heathland (Climoor Site)

Parameter	Mean	[1]Lower	[2]Upper	[3]Distribution	Correlations
$*BC_{dep}$	135	72	255	L	S_{dep}, 0.63; NO_{3dep}, 0.17; NH_{4dep}, 0.45;
Bc_{dep}	114	60	215	L	S_{dep}, 0.76; NO_{3dep}, 0.47;
$*S_{dep}$	1,300	690	2,453	L	BC_{dep}, 0.63; Bc_{dep}, 0.76; NO_{3dep}, 0.44.
NO_{3dep}	570	192	1,688	L	BC_{dep}, 0.17; Bc_{dep}, 0.47; S_{dep}, 0.44; NH_{4dep}, 0.51.
NH_{4dep}	680	360	1284	L	BC_{dep}, 0.45; NO_{3dep}, 0.51.

Units are eq ha^{-1} yr^{-1}. [1,2]The distributions described in Section 2.1.6 were used, the lower limit is the 0.5th percentile and the upper the 99.5th percentile. [3]L – lognormal.

1.4 Critical loads for nutrient nitrogen

1.4.1 Detailed model description

In order to be at long-term steady state, nitrogen inputs should be equal to nitrogen outputs. This is the basis of the steady state mass balance approach to setting critical loads for nutrient nitrogen. To avoid N accumulation, the critical load must be equal to the sum of all the relevant N sinks in a catchment. In the UK implementation of critical load methods, these are assumed to be the same as those used in calculating $CL_{min}N$, that is, N uptake (N_u), N immobilisation (N_i) and denitrification (N_{de}), plus a fourth term, the acceptable N leaching ($N_{le(acc)}$).

Hence

$$CL_{nut}N = N_u + N_i + N_{de} + N_{le(acc)} \qquad \text{(A6)}$$

Acceptable N leaching is defined as the leaching flux at which no damage occurs to the terrestrial ecosystem or linked (aquatic) ecosystems. This is a rather tenuous concept and there is no straightforward method of connecting leaching with damage. The UK approach is simply to use 4 kg N ha^{-1} yr^{-1} for managed conifers and 3 kg N ha^{-1} yr^{-1} for managed deciduous forests. The latest Mapping Manual (UBA, 2004) recommends using a nitrate concentration of 14.3 μeq L^{-1} to calculate acceptable N leaching. This will produce a lower value than the UK method except at very wet sites (run-off > 2,000 mm yr^{-1}). For the Monte Carlo analysis, we used a rectangular distribution for $N_{le(acc)}$ with a mean at the UK default value (286 eq ha^{-1} yr^{-1}), a minimum at the value recommended in the Mapping Manual, and a maximum of the same amount above the mean as the minimum is below. The equation was applied to the Liphook and Aber sites, the parameters being derived as above.

1.4.2 Input parameter sets used

Table A9: Terrestrial input parameters for nutrient N critical loads

	Liphook				Aber			
Parameter	**Mean**	**Lower**	**Upper**	**Distrib.**	**Mean**	**Lower**	**Upper**	**Distrib**
N_u	500	400	600	Rect.	210	105	315	Rect.
N_i	214	107	321	Rect.	71	35.5	107	Rect.
N_{de}	71	35.5	107	Rect.	71	35.5	107	Rect.
$N_{le(acc)}$	286	67	505	Rect.	286	86	486	Rect.

Units are eq ha^{-1} yr^{-1}. See Section 2.1.10.

Table A10: Deposition input parameters for nutrient N critical loads

	Liphook				Aber			
Parameter	**Mean**	**0.5%ile**	**99.5%ile**	**Distrib.**	**Mean**	**0.5%ile**	**99.5%ile**	**Distrib**
NO_{3dep}	392	132	1,160	L	940	318	2,784	L
NH_{4dep}	513	272	969	L	1,190	632	2,245	L

1.5 FAB model

1.5.1 Detailed model description

The first order acidity balance model (FAB model) is now the method of choice for calculating freshwater critical loads. In various variants it is used in the UK and in continental Europe. The model is derived from a combination of charge and mass balance approaches as described in Posch *et al.* (1997), Henriksen and Posch (2001) and UBA (2004). Implementation of the model in the UK is described in Hall *et al.* (2003, 2004a). Various routines and assumptions are used to calculate the sinks of deposited S and N in terrestrial catchments and lakes. The critical load criterion variable is the acid neutralising capacity (ANC), see below.

The FAB model can be written as:

$$*S_{dep} + N_{dep} = fN_u + (1 - r)(N_i + N_{de}) + rN_{ret} + rS_{ret} + *BC_{le} - ANC_{le} \qquad \text{(A7)}$$

where:

- * - indicates the non-marine fraction
- *dep* - deposition
- *f* - fraction of catchment area which is forested
- N_u - N uptake into plants
- *r* – ratio of lake area to total (lake + catchment area)
- N_i - N immobilisation in soil organic matter
- N_{de} - denitrification in the catchment soils
- N_{ret} - in-lake retention of N
- S_{ret} - in-lake retention of S
- BC_{le} - leaching of base cations ($Na^+ + K^+ + Ca^{2+} + Mg^{2+}$)
- ANC_{le} - acid neutralising capacity leaching

The sinks for nitrogen in the terrestrial catchment are uptake, immobilization and denitrification. Nitrogen uptake is calculated only for forest vegetation, as it is assumed that uptake by other species is only temporary, whereas the nitrogen in trees may be removed by harvesting. Hence, the value used is meant to represent the long-term average over the whole forest rotation. In the UK, two values for uptake are used, for managed coniferous forest (0.21 keq $ha^{-1}yr^{-1}$) and managed broadleaved forest (0.42 keq $ha^{-1}yr^{-1}$). Unmanaged woodland and other vegetation is assumed to have zero uptake. In practice, forest N uptake will vary over a considerable range. Similarly, N immobilization is the long-term sustainable immobilization, not the immobilization currently observed which is normally much higher. Immobilisation in the UK system is a fixed value dependent on soil type, and the value applied to the catchment is an area-weighted mean of all soil types present. Denitrification is treated in the same way as immobilisation in the UK, unlike in continental Europe where it is made dependent on N availability, with consequent complications.

The parameters rN_{ret} and rS_{ret} in Equation A7 calculate nitrogen and sulphur retention respectively in any lakes present and are discussed further below. The last two parameters in Equation A7 represent SO_4^{2-} and NO_3^- fluxes through surface waters, and are derived from the definition of ANC:

$$[ANC] = [BC] - [SO_4^{2-}] - [NO_3^-] - [Cl^-] \qquad \text{(A8)}$$

where brackets represent concentrations expressed in equivalence units. The chloride term can be eliminated by using non-marine values for base cation and sulphate fluxes, as:

$$[ANC] \approx [*BC] - [*SO_4^{2-}] - [NO_3^-] \qquad \text{(A9)}$$

Equation A7 can be translated into a critical load for acid deposition by calculating the terms on the right hand side, and specifying a criterion concentration for ANC. The idea is that catchments should be able to supply enough ANC to maintain the criterion ANC concentration indefinitely. The most common criterion value is 20 μeq L^{-1}, and although the UK originally used zero, it now uses 20 μeq L^{-1} except where there is evidence that the pre-industrial ANC was less than this (Hall *et al.*, 2004a: Section 5). All the other terms in Equation A7 are calculated at their steady state values - the only acceptable sinks for S and N, and sources for base cations, are those which are sustainable indefinitely. The critical load is then calculated by assuming a critical concentration for ANC, and multiplying by water run-off to obtain an ANC flux. The sustainable base cation leaching BC_{le} is essentially that derived from mineral weathering and pre-industrial non-marine base cation deposition, and is calculated in the form of a concentration $[*BC]_0$ by a method derived from an earlier model, the Steady State Water Chemistry Model, as follows. The current non-marine base cation concentration $[*BC]_t$ is used as a basis for the estimate. It is recognised, however, that $[*BC]_t$ may incorporate base cations leached from the soil ion exchange complex by acid deposition. These elements have therefore to be subtracted from $[*BC]_t$ to obtain $[*BC]_0$. The relationship between changes in SO_4^{2-} + NO_3^- and non-marine base cations in surface waters is expressed by the F-factor:

$$F = \Delta [*BC] / \Delta ([*SO_4^{2-}] + [NO_3^-]) \qquad \text{(A10)}$$

F is estimated by an empirical relationship with current base cation concentrations observed in Norwegian lakes (Brakke *et al*., 1990).

$$F = \sin\left(\frac{\frac{\pi}{2}[BC]_t}{S}\right) \qquad \text{(A11)}$$

This formula is merely a fitted curve giving roughly the right shape: S is set to 400 µeq L^{-1}, at which concentration F = 1. If $[*BC]_t > 400$ µeq L^{-1}, F is set to 1.

If the pristine sulphate concentration $[*SO_4^{2-}]_0$ in a lake is known, the pristine base cation concentration $[*BC]_0$ can be calculated. The pristine nitrate concentration is usually assumed to be zero. The pristine sulphate concentration is estimated by yet another empirical relationship observed in Norwegian lakes, in areas unaffected by acid deposition:

$$[*SO_4^{2-}]_0 = 15 + 0.16 \times [*BC]_t \qquad \text{(A12)}$$

It has long been thought that this equation gives values for $[*SO_4^{2-}]_0$ which are too high, and Henriksen and Posch (2001) give a set of alternative equations.

$[*BC]_0$ can now be calculated as:

$$[*BC]_0 = [*BC]_t - F \times ([*SO_4^{2-}]_t + [NO_3^-]_t - [*SO_4^{2-}]_0 - [NO_3^-]_0) \qquad \text{(A13)}$$

This procedure generates the last two terms in Equation A7 from a knowledge of $[*BC]_t$ and $[*SO_4^{2-}]_t$; $[NO_3]_0$ is assumed to be zero. If a criterion ANC concentration is chosen to protect aquatic organisms, the last two terms are abbreviated to L_{crit}, where:

$$L_{crit} = Q([*BC]_0 - [ANC]_{crit}) \qquad \text{(A14)}$$

and Q is run-off expressed in meters.

Lakes are powerful generators of alkalinity, and they cannot be ignored when acidification effects are being considered. The principal processes are S and N reduction in the lake sediments, and the rates of these two processes are estimated as follows. In-lake retention of N is assumed to be proportional to the net input of N, that is, N deposition less immobilisation, uptake and denitrification:

$$rN_{ret} = \rho_N(N_{dep} - fN_u - (1-r)(N_i + N_{de})) \qquad \text{(A15)}$$

where the factor ρ_N is modelled by an equation derived from work on North American lakes:

$$\rho_N = S_N / (S_N + (Q / r)) \qquad \text{(A16)}$$

The factor S_N is a mass transfer coefficient for N with units of velocity (m yr^{-1}).

Sulphur retention is treated in an exactly analogous way:

$$rS_{ret} = \rho_S S_{dep} \qquad \text{(A17)}$$

where the factor ρ_S is modelled by an equation based on the same North American lakes:

$$\rho_S = S_S / (S_S + (Q / r)) \quad \text{(A18)}$$

where the factor S_S is a mass transfer coefficient for S with units of velocity.

The FAB critical load equation can now be defined because all terms in Equation A7 can be calculated. From this, the nodes on the critical load function can be calculated: $CL_{max}S$ (the critical load for S deposition when N deposition is zero); $CL_{max}N$ (the critical load for N deposition when S deposition is zero) and $CL_{min}N$ (the sum of the three N sink parameters).

Henriksen and Posch (2001), however, recognised that this formulation did not take into account N falling directly into lakes, and produced a revised version of FAB. This now means that N deposition always makes some contribution to critical load exceedance if there is a lake in the catchment. It causes considerable complication, with three different equations for $CL_{max}N$ depending on the relative sizes of N deposition and the catchment N sinks. The UK formulation (Hall *et al.*, 2004a: Section 5.6) simplifies this situation by assuming that forest uptake can be averaged over the whole terrestrial catchment. The physical meaning of this is that forests can take up N from the non-forested parts of catchments, which are typically upslope. This is a sensible simplification, but even so there are two equations for $CL_{max}N$.

When $N_{dep} \leq CL_{min}N$ (no terrestrial nitrate leaching):

$$CL_{max}N = L_{crit}/r(1-\rho_N) \quad \text{(A19)}$$

When $N_{dep} > CL_{min}N$ (terrestrial nitrate leaching occurs):

$$CL_{max}N = L_{crit}/(1-\rho_N) + CL_{min}N \quad \text{(A20)}$$

The first of these recognises the theoretical possibility (C. Curtis, pers. comm.) that direct deposition of N to the lake surface can result in exceedance before terrestrial N leaching occurs, due perhaps to a high value of *r* and a very high terrestrial N sink ($CL_{min}N$). This will lead to the apparently bizarre situation that $CL_{max}N$ may be less than $CL_{min}N$. The definition of $CL_{min}N$ is now the minimum N deposition load that results in terrestrial N leaching. The formulae for $CL_{max}S$ and $CL_{min}N$ are always:

$$CL_{max}S = L_{crit} / (1-\rho_S) \quad \text{(A21)}$$

$$CL_{min}N = (1-r)(N_u + N_i + N_{de}) \quad \text{(A22)}$$

1.5.2 Implementation of FAB for Monte Carlo analysis

This new version of FAB proved quite difficult to implement in MS Excel. Results from all the AWMN sites were compared with the official submissions produced by ENSIS (which are calculated using MS Access). This time-consuming but valuable exercise ensured the uncertainty analysis below reflected the use of the FAB model in the UK.

$CL_{max}N$ was the most awkward value to calculate because of the alternative formulations (Equations A19 and A20). An initial test was made to determine whether N_{dep} was greater than $CL_{min}N$. If so, Equation A20 was used. If, however, N_{dep} was less than or equal to $CL_{min}N$, (Equation A19), the results are only meaningful under certain circumstances. If there is no lake in the catchment, and $N_{dep} < CL_{min}N$, there is no meaningful value for $CL_{max}N$, as all deposition is absorbed by the catchment. Monte Carlo runs with this combination of parameters were thus discarded from the simulations. If the catchment did contain a lake, and $N_{dep} < CL_{min}N$, then the value of $CL_{max}N$ calculated by Equation A19 would normally be much higher than that calculated by Equation A20. If, however, by raising N deposition to this level it became greater than $CL_{min}N$, the lower (Equation A20) value would apply. $CL_{max}N$ would then be less. To resolve this paradox, runs for which $CL_{max}N < CL_{min}N$ were only accepted if there was a lake in the catchment and $N_{dep} < CL_{min}N$. These two conditions resulted in discarding up to 25 per cent of the Monte Carlo runs in certain sites. Uncertainty statistics for $CL_{max}N$ have been calculated using this reduced number of runs. For the other critical load parameters, statistics have been calculated using the full set of runs since the runs discarded will tend to be those of lower N deposition, thus affecting exceedance. Exceedance calculations are fortunately independent of $CL_{max}N$.

Exceedance was defined as:

$$Ex = (S_{dep} + N_{dep}) - L_{crit} \qquad \text{(A23)}$$

This is *not* the amount by which S and N deposition has to be reduced to reach the critical load. The different adsorption capacities of catchments and lakes for S and N mean there is no unique value for this quantity.

1.5.3 Fish health functions

The distributions of healthy and reduced populations of all fish and brown trout as a function of ANC were read from the graphs in Lien *et al.* (1996). These were approximated by functions suitable for implementation in MS Excel. Functions and original data are shown in Figures A2 to A5. The distribution of reduced fish populations fits well to a normal distribution with mean 1.9 μeq L^{-1} and standard deviation 16.2 μeq L^{-1} (Figure A2). The distribution of reduced brown trout populations also fits a normal distribution, though not quite so well at the extremes (Figure A3). The parameters were mean, –1.0 μeq L^{-1} and standard deviation, 16.0 μeq L^{-1}. The distributions for extinct populations are asymmetric, but fit reasonably well to four straight lines (Figures A4 and A5).

The probability of extinct brown trout populations was zero if ANC > 10 μeq L^{-1}; and expressed as a percentage, -0.84 ANC + 8.4 for ANC between 0 and 10 μeq L^{-1}; -2.03555 ANC + 8.4 for ANC between 0 and -45 μeq L^{-1}; and 100% for ANC < -45 μeq L^{-1}.

For fish in general, the probability of extinct populations was zero if ANC > 1 μeq L^{-1}; and expressed as a percentage, -1.026 ANC + 1.3636 for ANC between 1 and -10 μeq L^{-1}; -2.5 ANC - 13.3781 for ANC between -10 and -45 μeq L^{-1}; and 100% for ANC < -45 μeq L^{-1}.

The current UK critical load of 20 μeq L^{-1} thus corresponds to a probability of reduced fish and brown trout populations of about 10-13%, but a virtually zero probability of population extinction. The previous criterion of 0 μeq L^{-1} had a probability of reduction of around 50%, and an extinction probability of 9-12%, always assuming these functions are valid for UK waters.

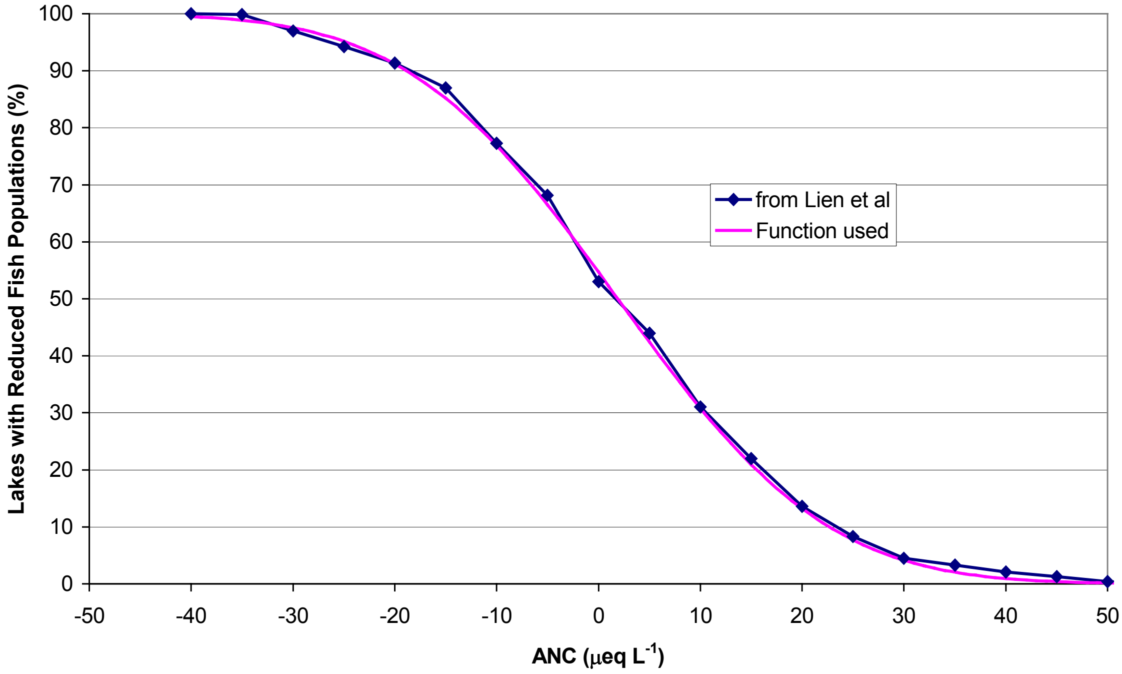

Figure A2: Relationship between ANC and percentage of lakes with reduced fish populations. The blue points are literature data; the purple line is a fitted function.

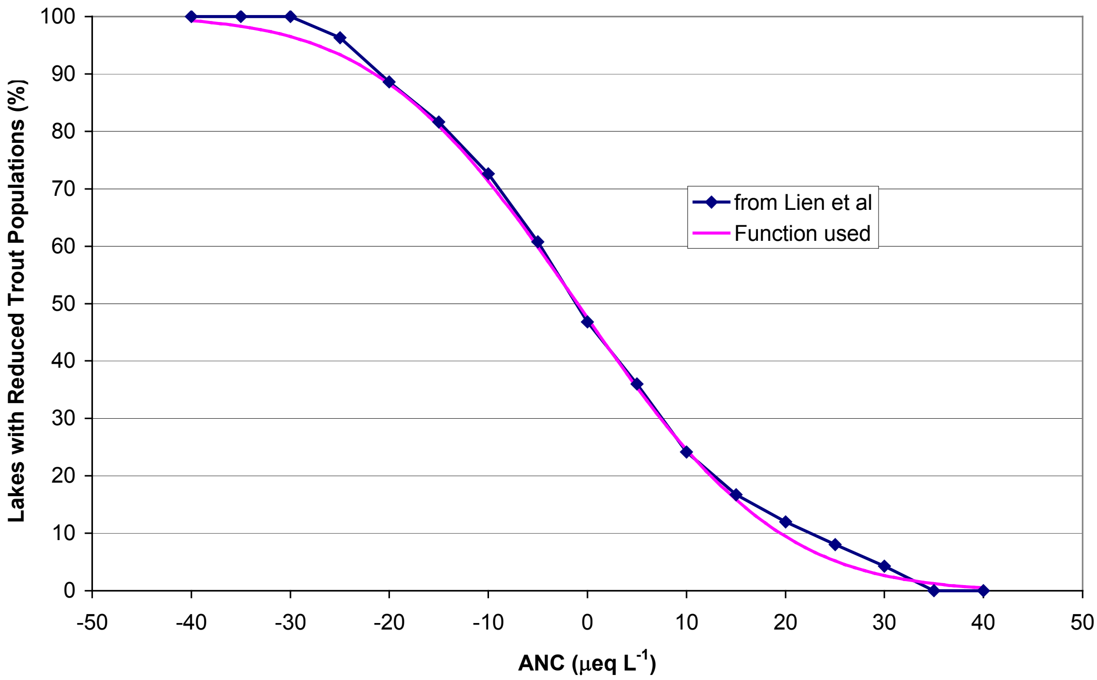

Figure A3: Relationship between ANC and percentage of lakes with reduced brown trout populations. The blue points are literature data; the purple line is a fitted function.

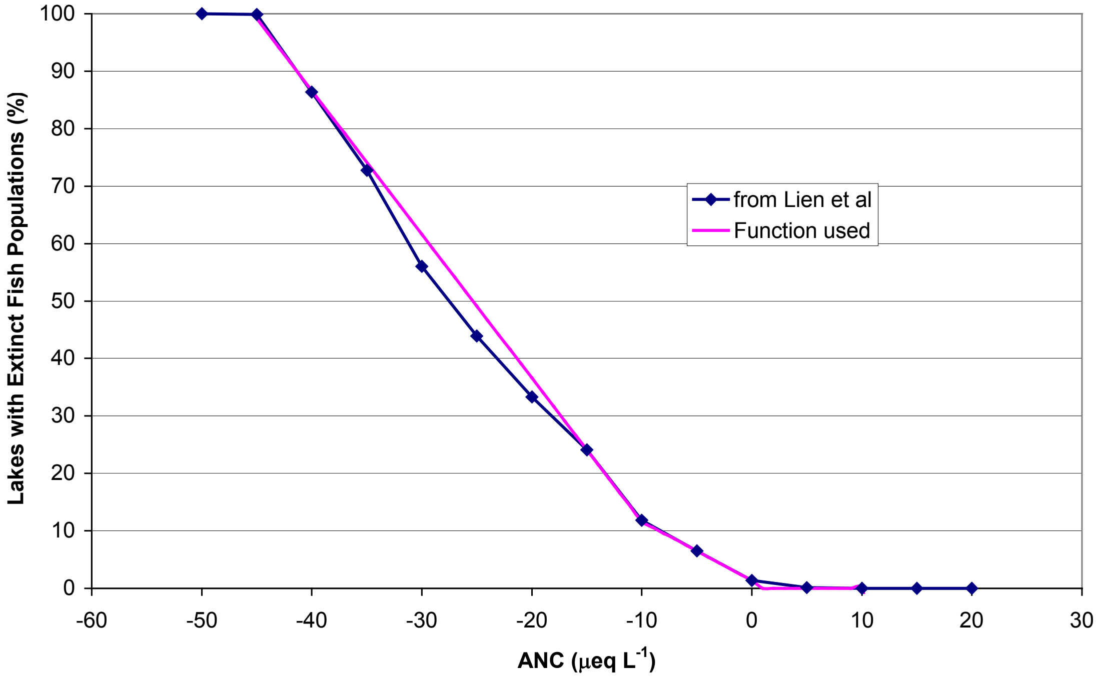

Figure A4: Relationship between ANC and percentage of lakes with extinct fish populations. The blue points are literature data; the purple line is a fitted function.

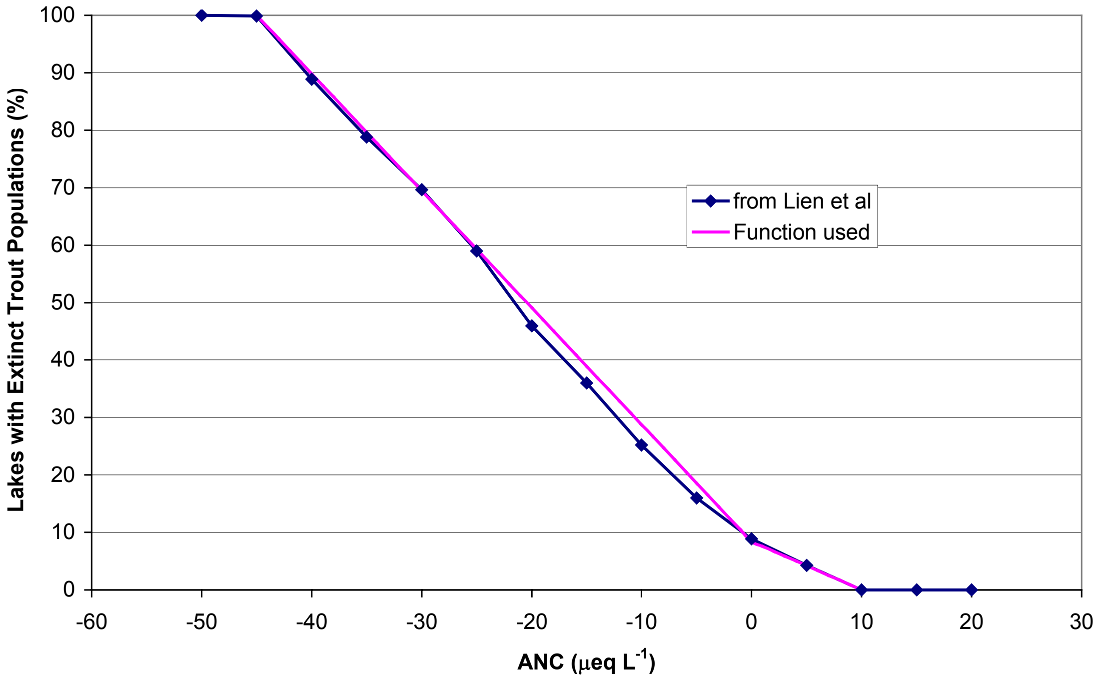

Figure A5: Relationship between ANC and percentage of lakes with extinct brown trout populations. The blue points are literature data; the purple line is a fitted function.

Appendix B – Methods Section 4

Appendix B contains details of input parameters for Section 4 of the main report: comparison of national and site-specific uncertainty.

Table B1 shows default values and uncertainty ranges for input data parameters for Liphook, Thetford and Aber comparison of national and site-specific data. “N” is national data and “SS” site-specific.

Table B2 shows default deposition parameters used in this study.

Table B1: Input parameters for the site-specific / national comparison

Parameter	Site/Scale	Input	Lower	Upper	SD	Distribution
Base cation weathering (eq $ha^{-1}yr^{-1}$)	Liphook N	100	0	200		Rectangular
	Liphook SS	150	50	250		Rectangular
	Thetford N	4,000	2,000	6,000		Rectangular
	Thetford SS	4,600	2,300	6,900		Rectangular
	Aber N	350	200	500		Rectangular
	Aber SS	163	120	300		Rectangular
Base cation uptake (eq $ha^{-1}yr^{-1}$)	Liphook N	270			62	Rectangular
	Liphook SS	270	135	540		Rectangular
	Thetford N	270			62	Normal
	Thetford SS	270	135	540		Rectangular
	Aber N	270			62	Normal
	Aber SS	316	284	348		Rectangular
Calcium correction factor (dimensionless)	Liphook N	0.1	0	0.2		Rectangular
	Liphook SS	0.57	0.47	0.67		Rectangular
	Thetford N	1	0.8	1		Rectangular
	Thetford SS	0.9	0.85	0.95		Rectangular
	Aber N	0.1	0	0.2		Rectangular
	Aber SS	0.092	0.042	0.142		Rectangular
Calcium uptake (eq $ha^{-1}yr^{-1}$)	Liphook N	160			43.2	Normal
	Liphook SS	125	80	320		Rectangular
	Thetford N	160			43	Normal
	Thetford SS	160	80	320		Rectangular
	Aber N	160			43	Normal
	Aber SS	160	80	320		Rectangular
Run-off (m^3 $ha^{-1}yr^{-1}$)	Liphook N	4,100			943	Normal
	Liphook SS	4,690	4,221	5,159		Rectangular
	Thetford N	1,620			373	Normal
	Thetford SS	1,620	1,458	1,782		Rectangular
	Aber N	13,800			3174	Normal
	Aber SS	5,930	5,337	6,523		Rectangular
Ca/Al_{crit} (mol mol^{-1})	All sites	1	0.5	1.5		Rectangular
Gibbsite coefficient (m^6 eq^{-2})	Liphook N	950	760	1,140		Rectangular
	Liphook SS	950	300	3,000		Rectangular
	Thetford N	950	760	1,140		Rectangular
	Thetford SS	950	300	3,000		Rectangular
	Aber N	100	80	120		Rectangular
	Aber SS	100	10	380		Rectangular
Nitrogen immobilisation (eq $ha^{-1}yr^{-1}$)	Liphook N	214	107	321		Rectangular
	Liphook SS	214	107	321		Rectangular
	Thetford N	71	35.5	107		Rectangular
	Thetford SS	71	35.5	107		Rectangular
	Aber N	214	107	321		
	Aber SS	150	120	180		Rectangular
Nitrogen uptake (eq $ha^{-1}yr^{-1}$)	Liphook N	210			57	Normal
	Liphook SS	500	400	600		Rectangular
	Thetford N	210			57	Normal
	Thetford SS	210	105	420		Rectangular
	Aber N	210			57	
	Aber SS	246	221	271		Rectangular
Denitrification (eq $ha^{-1}yr^{-1}$)	Liphook N	71	36	107		Rectangular
	Liphook SS	71	35.5	107		Rectangular
	Thetford N	71	36	107		Rectangular
	Thetford SS	71	35.5	107		Rectangular
	Aber N	71	35.5	107		Rectangular
	Aber SS	35.5	28	43		Rectangular

Table B2: Default deposition parameters for Liphook, Thetford and Aber

Scenario	$NH4_{dep}$	$NO3_{dep}$	$^{*}S_{dep}$	$^{*}BC_{dep}$	Ca_{dep}
Liphook N	1,350	1,190	780	300	430
Liphook SS	513	392	653	582	175
Thetford N	1,640	1,240	490	190	220
Thetford SS	1,640	1,240	490	190	220
Aber N	1,190	940	830	180	310
Aber SS	1,190	940	1,481	397	485

Values are in eq ha^{-1} yr^{-1}.

Appendix C – Methods Section 5

Appendix C contains input data for TRACK Modelling

Table C1 summarises the input parameters used in the model runs. The baseline model run was based on these parameter values. Table C2 summarises the range of input values used for Monte Carlo simulation. Values of each parameter were taken at random from the range, assuming a uniform distribution. Three hundred model runs were carried out.

Table C1: TRACK model set up parameters

Parameter	**Value**
Receptor grid	Ordnance Survey
Receptor square dimension	20 km
Stability class	4
Mean wind speed	7.5 m s^{-1}
Trajectory duration	96 hours
Trajectory time step	120 seconds
Lookup time step	2880 seconds
Number of incoming trajectories incoming at receptor	24
Number of trajectories per 24 hour period	4
Choice of solver	Fixed time step
Number of vertical levels	Single level
Deposition velocities	Spatially disaggregated values based on surface types (Table 3.6)
Choice of season	All year
Seeder-feeder enhancement	Orographic
Tracer	Off
Source type	Area sources

Table C2: Input ranges of parameters used in Monte Carlo simulation and first order analysis for TRACK

Parameter	Units	Baseline	Range, % of baseline	
			Lower limit	Upper limit
Dry deposition velocity, NO_2		Land use dependent	40	160
Dry deposition velocity, HNO_3		Land use dependent	50	200
Dry deposition velocity, aerosols		Land use dependent	50	200
Dry deposition velocity, ammonia		Land use dependent	50	200
Dry deposition velocity, sulphur dioxide		Land use dependent	50	200
Ozone relaxation rate	$m\ s^{-1}$	$50.\ 10^{-6}$	50	150
Aerosol relaxation rate	$m\ s^{-1}$	0.001	50	150
Wet scavenging coefficient, HNO_3	s^{-1}	$9.\ 10^{-6}$	0	200
Wet scavenging coefficient, aerosols	s^{-1}	$1.3.\ 10^{-5}$	30	1,000
Wet scavenging coefficient, sulphur dioxide	s^{-1}	$1.\ 10^{-6}$	0	200
Wet scavenging coefficient, NH_3	s^{-1}	$9.\ 10^{-6}$	20	100
Reaction rate, $NO + O_3 \rightarrow NO_2 + O_2$		See P4(083)	80	120
Reaction rate, $OH + NO_2\ (+ M) \rightarrow HNO_3\ (+ M)$		See P4(083)	70	130
Reaction rate, $SO_2 + OH \rightarrow SO_4$		See P4(083)	20	100
Reaction rate, $SO_2 \rightarrow SO_4$		See P4(083)	50	200
Reaction rate, $SO_4 + 2NH_3 \rightarrow (NH_4)_2SO_4$		See P4(083)	30	1,000
Reaction rate, $NH_3 + HNO_3 \rightarrow NH_4NO_3$		See P4(083)	30	1,000
Reaction rate, $NO_2 + h\nu \rightarrow NO + O$		See P4(083)	50	125
Reaction rate, $NO_2 + O_3 \rightarrow NO_3 + O_2$		See P4(083)	50	200
Reaction rates gaseous species with aerosols		See P4(083)	50	200
[OH]		STOCHEM field	50	200
[CH_3COO_2]		STOCHEM field	50	200
[NO_3]		STOCHEM field	50	200
[O_3]	ppb	34	80	120
Fraction sulphur released as sulphur trioxide or sulphate		0.05	60	140
Seeder feeder factor on wet deposition rate		2	50	150
Seeder feeder factor on SO_2 oxidation rate			70	130
Ammonia emissions		Inventory	50	150
Hydrogen chloride emission		Inventory	80	120
Oxides of nitrogen emissions		Inventory	80	120
Boundary layer height	m	800	80	120

Monitoring data

Monitoring data used in the assessment were taken from *Management and Operation of the UK Acid Deposition Monitoring Network: Data Summary for 2003*. The data are summarised in Table C3. The data were supplemented by additional data for ammonia and ammonium for 2002 from the UK National Ammonia Monitoring Network for sites included in Table C3.

Table C3: Network sites and measurements made in 2003.

Measurement:	Precipitation		NO_2	SO_2	Part. SO_4	Denuder HNO_3-NO_3
Frequency: SITE:	*daily bulk*	*fortnightly bulk*	*monthly*	*fortnightly*	*daily*	*monthly*
Yarner Wood		✓ - 1	✓	✓ - 2	✓	✓
Lough Navar		✓ - 1	✓	✓ - 2, 4	✓	✓ - 3
High Muffles		✓ - 1	✓	✓ - 2	✓	✓ - 3
Eskdalemuir	✓	✓ - 1	✓	✓ - 2	✓	✓ - 3
Strathvaich Dam		✓ - 1	✓	✓ - 2, 4	✓ - 6	✓ - 3
Barcombe Mills		✓ - 1	✓	✓ - 2	✓	✓ - 3
Stoke Ferry		✓ - 1	✓	✓ - 2	✓ - 6	✓ - 3
Glen Dye		✓ - 1	✓	✓ - 2	✓ - 6	
Goonhilly		✓ - 1	✓			
Compton		✓ - 1	✓			
Flatford Mill		✓ - 1	✓			
Woburn		✓ - 1	✓			
Tycanol Wood		✓ - 1	✓			
Llyn Brianne		✓ - 1	✓			
Pumlumon		✓ - 1	✓			
Preston Montford		✓ - 1	✓			
Bottesford		✓ - 1	✓			
Llyn Llydaw		✓ - 1	✓			
Wardlow Hay Cop		✓ - 1	✓			
Driby		✓ - 1	✓			
Jenny Hurn - 7						
Thornganby		✓ - 1	✓			
Bannisdale		✓ - 1	✓			
Hillsborough Forest		✓ - 1	✓			
Cow Green Reservoir		✓ - 1	✓			
Loch Dee		✓ - 1	✓			
Redesdale		✓ - 1	✓			
Whiteadder		✓ - 1	✓			
Balquhidder		✓ - 1	✓			
Polloch		✓ - 1	✓			
Allt a' Mharcaidh		✓ - 1	✓			
Achanarras		✓ - 1	✓			

(1) The sampling frequency of the bulk deposition monitoring was changed from weekly to fortnightly with effect from November 2001.
(2) The daily bubbler measurement programme was replaced with a fortnightly filter-pack measurement programme during 2001.
(3) A site in the CEH HNO_3 Denuder Monitoring Network (see Section 4).
(4) This site, together with those at Bush, Cwmystwyth and Sutton Bonington, was used as an overlap site for the introduction of the filter-pack sampler.
(5) The daily wet-only measurement was stopped with effect from November 2001.
(6) The daily particulate sulphate measurements were stopped with effect from November 2001.
(7) This site was closed with effect from November 2001.

We are The Environment Agency. It's our job to look after your environment and make it **a better place** – for you, and for future generations.

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Environment Agency
Rio House
Waterside Drive, Aztec West
Almondsbury, Bristol BS32 4UD
Tel: 0870 8506506
Email: enquiries@environment-agency.gov.uk
www.environment-agency.gov.uk