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Assessing the risks of air pollution impacts to the condition of Areas/Sites of Special Scientific Interest in the UK

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Executive Summary

The Conservation Agencies are required to assess the condition of each designated site. To do this the agencies are instituting a CSM (common standards monitoring) scheme to determine the condition of each site at regular (6 yearly) intervals. Sites may be under threat from a wide variety of forces such as inappropriate use and management, climate change, invasive species, pollution and so on. Unfortunately the link between cause and effect can take many years to manifest itself, and in addition it may be difficult to attribute a single cause to an observed effect (as multiple causes can generate very similar looking effects).

Air pollution could have a negative impact on designated sites and hence needs to be investigated by the Conservation Agencies. The concept of a “critical load” and of a “critical level” has been developing since the late 1980’s for use in national and international assessments. The two approaches are very similar and adopt a strict precautionary principle; the critical load is the amount of pollutant that can be deposited without causing harmful effects to sensitive elements (eg, soils, waters, vegetation) of the environment according to present knowledge and the critical level is the concentration of a pollutant in the atmosphere below which there are no known harmful effects. All the major airborne pollutants have been considered but only three groups; oxides of sulphur, nitrogen compounds and ozone are well enough studied to be suitable for routine and widespread analysis.

Any consideration of critical loads and levels must acknowledge that there are considerable uncertainties in the source and magnitude of pollutant emissions, in the location and sensitivity of receptors and in the physiological and biogeochemical processes involved. These issues are discussed throughout the report with some suggestions as to how to minimise their effects on any conclusions drawn. Assessment of designated sites can be restricted by; lack of available data, errors and uncertainties in the available data and lack of scientific knowledge (especially field experiments) on the sensitivity of designated features (requiring expert judgement to be used).

We have considered a number of options for assessing the risks of airborne pollutants to terrestrial designated sites. Some methods would impose very considerable resource commitments from the agencies. Because air pollution is only one of many potential negative impacts on designated sites and because of realistic resource constraints we recommend that a simple hierarchical or staged approach is adopted. Stage 1 is a centralised automated screening of all sites making use of national data and internationally agreed protocols. In many cases the pollution levels will be either so high or so low as to be able to clearly assess the risk to the sites (in the framework of the CSM). Some sites will be receiving pollutants at a rate where it is unclear if the critical load or level is being exceeded; in those cases we recommend that Stage 2 be performed. In Stage 2 the analysis is repeated using whatever site specific data is available. In many cases the most important single piece of site specific information will be on the soil series relevant the designated feature. Finally we recommend that surveyors visiting the sites as part of the CSM be additionally tasked to record the presence of potential sources of pollutant not likely to be included in national data sets (eg new intensive livestock facilities close to the designated sites) and to record the general condition of the vegetation (to provide qualitative confirmation of the predictions in Stage 1 & Stage 2).

1. Introduction

The Joint Nature Conservation Committee (JNCC) is responsible for coordinating the assessment of the state of the designated sites (predominantly Sites of Special Scientific Interest (A/SSSI)) and trying to ensure that they retain their interest. A Public Service Agreement has been set that by 2010, 95% of SSSI (by area) in England will be in a favourable condition. In 2006 JNCC will publish the first comprehensive assessment using the "Common Standards Monitoring" (CSM) protocol of all designated sites. They expect to repeat this exercise every six years.

The CSM protocol for terrestrial sites does not specifically address the issue of air pollution and it is therefore possible that different surveyors will treat the issue in an inconsistent manner. The purpose of this report is:

- To critically evaluate options for assessing the risks to terrestrial SSSIs and their notified interest features from air pollution, based on the concept of critical loads/levels exceedance, and to recommend the option(s) most fit for purpose.
- Evaluate options in the context of whether they add value to current data produced for national mapping or the Environment Agency/SNIFFER site relevant critical loads project.

Critical loads are a widely used tool for assessing whether an area is at risk from the harmful effects of acidifying (sulphur, nitrogen) or eutrophying (nitrogen) airborne pollutants. The critical load is set so that if deposition is below the critical load in the ***long-term steady state*** situation there will be no known detrimental effect; that is it employs a strict "precautionary principle" approach. Exceedance of these "steady-state" critical loads indicates the potential for harmful effects in the long-term. It is therefore not possible to validate critical load exceedances in the field, since unless the system is in steady-state conditions, the system may not actually be exceeded and adverse effects may not be observable. The use of dynamic models (eg, MAGIC) allow timescales for chemical states (eg, soil or water pH) to be calculated, but their consideration is not within the scope of this report. In addition, it may be difficult to disentangle the interactions between the abiotic and biotic stresses, for example, is a change in species composition due to inappropriate grazing, climate change or nitrogen deposition?

In Section 2 we provide a background to the major air pollutants amenable to assessment by a critical loads/levels approach and a brief discussion of pollutants where our understanding of their distribution, behaviour and effects is currently insufficient for a critical loads/levels approach. Section 3 reviews the production of site relevant critical loads and source attribution. Section 4 discusses applying critical loads or levels to a designated feature; this is a particular problem in that critical loads/levels have only been developed in relation to a limited number of vegetation habitats, and these through a few particular species. Section 5 summarises the preceding three sections and offers advice and recommendations on how to assess designated sites. Conclusions are presented in Section 6.

2. Air Pollution and Risk Assessment

This section describes the pollutants to be covered in a risk assessment and why.

2.1 Major pollutants to be considered

Pollutants can affect the natural environment in a number of different ways; in this report we consider the three main ways, being:

- Acidification
- Eutrophication
- direct physiological damage

These are described in the sections below. In addition there are other pollutants and other mechanisms for damaging the environment for which there is currently no consensus (and often no data) on how to perform a risk assessment; these are also listed in a later section.

2.1.1 Acidification

The key acidifying pollutants are various compounds of sulphur and nitrogen. These can occur as gases; sulphur dioxide, nitrogen dioxide and nitric acid; in precipitation (sulphate, nitrate); or as aerosols. Reduced nitrogen species (ammonia, ammonium) can also contribute to acidification (CLAG, 1994; NEGTAP, 2001). Acid deposition in this context therefore comprises the sum of wet, dry and cloud-droplet deposition of non-marine sulphur, oxidised and reduced nitrogen. Whilst soils can naturally become acidified over time, acid deposition can lead to enhanced changes in both soils and waters, which can impact on the dependent habitats. Elvingson & Agren (2004) summarise the key effects of soil acidification as:

- The leaching of plant nutrients, especially the base cations magnesium, potassium, calcium (Mg, K, Ca). This can affect plant growth and combined with lower soil pH levels can lead to the displacement of some sensitive plant species.
- The concentration of toxic metals in soil water can increase; for example, free aluminium (Al) ions can be released which are potentially toxic to plant root systems. In addition, as the mobility of many heavy metals, increases this can have negative effects on soil organisms (especially decomposers).
- Released Al ions bind phosphorous (as Al phosphate) making it less accessible to plants, and this shortage of phosphorous is aggravated by the slower decomposition rates in acidified soils.

Many designated sites aim to protect habitats that are sensitive to acidification, such as dwarf shrub heath and acid grassland. Hence it is important that acidity is included in the risk assessment process.

Work at the national scale has shown many areas of the UK to be at risk from the potential harmful effects from acidification, particularly areas in the uplands and north and west of the country, where the soil types offer less buffering to the incoming acid deposition, and where the rainfall (and hence acid deposition load) tends to be higher.

Calculations of exceedance of critical loads (ie, the excess deposition above the critical load) based on the most recent (2001-2003) deposition data indicate that 59% of sensitive habitats are exceeded for acidity.

2.1.2 Eutrophication (nutrient nitrogen)

Nitrogen is the main soil derived nutrient and plays a major role in plant and ecological processes and inputs of nitrogen from the atmosphere are an important source of this nutrient. Excess nitrogen deposition, known as eutrophication, can favour the growth of a few species of competitive plants (for example stinging nettles, *Urtica* spp.) and lead to changes in ecosystem structure or function and to a reduction in biodiversity. For example, competition by plants that thrive on the higher nitrogen availability can lead to changes in species composition within communities, or transitions from one vegetation type to another, eg, heather to grass. Decreases in communities of bryophytes and lichens are likely in many habitat types. An excess of nitrogen in forests can lead to an increased susceptibility to pathogens and parasites, changes or decreases in the below-ground species composition (eg mycorrhizal fungi), as well as detrimental effects on the ground flora. Further information on the effects of nitrogen on a range of habitat types across Europe can be found in Achermann & Bobbink (2003).

Both oxidised and reduced forms of nitrogen deposition contribute to eutrophication. Many designated sites include areas of habitats sensitive to excess nitrogen deposition, so this pollutant needs to be included in the risk assessment process.

National scale studies show the potential adverse effects of excess nitrogen on natural and semi-natural habitats to be widespread across the UK, with the critical loads of 60% of sensitive habitats being exceeded by present day (2001-2003) total (oxidised plus reduced) nitrogen deposition.

2.1.3 Ozone

Low level ozone (O_3) is formed by complex chemical interactions between oxides of nitrogen, hydrocarbons and sunlight. Ozone concentrations are highly variable both spatially and temporally. Annual averages vary considerably across the country and there are significant differences between rural and urban means. Urban and industrial regions have an annual average ozone concentration of around 10 ppb. In rural areas this is nearer 25 ppb and in upland areas it is as high as 35 ppb (NEGTAP 2001). It is in these upland and rural areas that conservation sites are typically found, leading to a potential impact on A/SSSIs from ozone pollution. There are broad spatial and seasonal gradients as well with summer-time concentrations lowest in the north west, and higher concentrations in the south east of the UK, and into mainland Europe. This is due to higher solar radiation, temperatures, and emissions of precursor gases. This pattern is reversed during the winter months as the ozone is 'destroyed' by the higher NO in polluted areas in the south and south east of the UK.

Impacts of ozone range from visible injury to foliage (although rarely reported in the UK), to the reduction in growth and changed reaction to water stress through changes to the stomatal conductance. Foot *et al.,* (1996, 1997) have shown that heather (*Calluna vulgaris*) exposed to O_3 has an increased susceptibility to damage from frost.

2.1.4 Pollutants for which there is currently insufficient information

Heavy metals are currently excluded from the list of pollutants to be considered for site-specific assessments. Research into the development of heavy metal critical loads (eg, cadmium, copper, lead, zinc) is ongoing in the UK (Ashmore *et al*., 2004) and in Europe (Slootweg *et al*., 2005). However, the national maps based on current methods and data are only intended for assessing the potential impacts of heavy metal deposition at national and international scales. For local or site-specific assessments it is inadvisable to use the national maps as part of a site model for risk assessment (Hall *et al*., 2006). Heavy metal deposition is being monitored in the UK (http://www.heavymetals.ceh.ac.uk/), so it may be appropriate to incorporate heavy metals into risk assessments in the future when more data and supporting research are available.

Critical load methods have been proposed (by the Dutch) for POPs (persistent organic pollutants), however, the UK is not currently planning to develop this approach.

2.2 Deposition, Critical Loads and Exceedance

2.2.1 Deposition

The Acid Deposition Monitoring Network of 32 sites provides data for a number of pollutant species that are used to calculate totals of acid and N deposition. Measurements are made weekly or on a daily basis. Wet deposition is calculated from the concentration of the specified ion in precipitation and the Meteorological Office rainfall field. Pollutant species that are measured include sulphate (SO_4^{2-}), nitrate (NO_3^-), ammonium (NH_4^+), sodium (Na^+), magnesium (Mg^{2+}), calcium (Ca^{2+}), chloride (Cl^-), potassium (K^+), non sea salt sulphate ($nssSO_4^-$), conductivity, hydrogen ion concentration (H^+), and rainfall.

The monitoring stations that are used to calculate wet deposition are used to define the broad spatial patterns in rainfall composition (NEGTAP 2001). The precipitation weighted concentrations of each ion is calculated as the product of the annual precipitation and provided for the UK at a 5km grid resolution. However, the monitoring stations do not sample the full range of precipitation amount or composition particularly at high altitudes. Therefore, an orographic enhancement is added to the mapped UK concentration field to account for wash-out of hill cloud through the seeder-feeder effect (Fowler *et al*., 1998).

Dry deposition of NO_x comes from rural nitrogen dioxide diffusion tube data, and NH_3 comes from the UK National Ammonia Monitoring Network. Recently further monitoring has been carried out for nitric acid (HNO_3^-) and aerosol concentrations of NO_3^-, NH_4^+, SO_4^{2-}.

Acid deposition is defined as the sum of non-marine sulphur, plus oxidised and reduced (ie, ammonia) nitrogen deposition. The deposition values are the sums of wet, dry and cloud-droplet deposition. This calculation assumes that all non-seasalt sulphur and nitrogen deposition is acidifying. In practice, a fraction of the nitrogen deposition may be accumulated by the ecosystem resulting in actual acidification being less than the figure estimated here. The ameliorating effects offered by non-marine base cation (calcium plus magnesium) deposition are taken into account in the calculation of the acidity critical loads (ie, in determining the maximum critical load of sulphur – see Section 2.2.7).

Estimated nitrogen deposition includes all reduced (NH_3, NH_4^+) and oxidised (NO_2, NO_3^-) forms including nitric acid (HNO_3^-). It is usually assumed that both oxidised and reduced nitrogen lead to the same ecological effects (Sutton & Fowler, 1993; Hornung *et al.* 1995a), but there is some indication that this may not always be strictly be true. The effects of ammonium (from ammonia) and of nitrates (from nitrogen oxides) potentially differ. Emissions of nitrogen oxides are more widely dispersed from their sources than the emissions of ammonia, which tend to be deposited locally.

Deposition data are mapped for the UK on a $5km^2$ grid at CEH Edinburgh and produced as annual averages over a three year period, so as to smooth out some of the inter-annual variability related primarily to meteorology. The most recent available are the mean data for 2001-2003. Deposition rates are affected by the aerodynamic roughness of the land cover (approximately proportional to the average height above the ground), therefore three separate data sets are produced:

(i) assuming an average deposition to all land cover categories
(ii) assuming low vegetation everywhere (the "moorland" option)
(iii) assuming tall vegetation everywhere (the "woodland" options)

The values in (ii) are used in the calculation of exceedances for all non-woodland terrestrial habitats and those in (iii) for all woodland habitats. The average data set (i) are used for freshwater habitats (lakes, rivers) and are not considered within this report.

It is planned to update the three-year mean data sets on an annual basis, in the spring of each year. In addition to the data for 2001-2003, data sets are also available for the years 1995-97, 1998-2000 and 1999-2001. The data are available free of charge from CEH or by accessing the Air Pollution Information System (APIS) web site (www.apis.ac.uk), where single 5km deposition values can be extracted by location (using GB OS grid reference). In the near future the data will be available from a dedicated CEH web site (http://www.uk-pollutantdeposition.ceh.ac.uk).

In addition to the estimates of present day deposition, the FRAME (Fine Resolution Atmospheric Multi-pollutant Exchange) model provides modelled estimates of future or hindcast deposition scenarios. FRAME uses emission data (mainly from the National Atmospheric Emission Inventory) to estimate the contribution to deposition of sulphur and nitrogen across the United Kingdom based on a 5km grid (http://www.frame.ceh.ac.uk).

FRAME is a Lagrangian model which considers an air column moving along straight line trajectories. The atmosphere is divided in to 33 separate layers with variable thickness, varying from 1m at the surface to 200m at the top of domain. Separate trajectories are run at a

15^o resolution for all grid edge points. The model includes a diurnal cycle which influences boundary layer height. A climatological wind rose is used to give the appropriate weighting to directional deposition and concentrations for calculation of total deposition and average concentration. Wet deposition calculated using a diurnally varying scavenging coefficient depending on mixing layer depth. A precipitation model is used to calculate wind-direction-dependent orographic enhancement of wet deposition. Dry deposition for NH_3 is ecosystem specific, including a version with bi-directional NH_3 exchange. Dry deposition of NO_2 and SO_2 is derived from the CEH deposition model and is ecosystem dependent, but based on a grid average only. The model chemistry includes gas phase and aqueous phase reactions of oxidised sulphur and oxidised nitrogen and conversion of NH_3 to ammonium sulphate and ammonium nitrate aerosol The chemical species treated include NH_3, NH_4^+ aerosol, NO, NO_2, HNO_3, PAN, NO_3^- aerosol, SO_2, H_2SO_4 and SO_4^{2-} aerosol.

There is generally good agreement between the measured interpolated UK map and FRAME. However, there are a number of inconsistencies with some under-estimation of NO_y dry deposition and over-estimation of NO_y wet deposition by FRAME. Good correlation is found with measurements of wet deposition and ammonium aerosol concentrations. There is considerable scatter in the correlation with ammonia gas concentration measurements due to the high sub-grid spatial variability for this species. While for SO_2 concentrations, FRAME was successful in estimating concentrations at remote sites but less accurate for concentrations near strong sources, which again may be linked to sub-grid variability. A general tendency of the FRAME model was to generate lower estimates of deposition than the measured map for the remote areas of Scotland with higher values near source areas, particularly in northern England.

The current year for FRAME is 2002, with previous years covering 1996 and 1999. FRAME is also used in modelling future years (2010). Datasets are available from CEH Edinburgh.

2.2.2 Uncertainties in atmospheric deposition

While concentrations in the air are easily measured and monitored it is much harder to estimate the amount of pollutant that is deposited to vegetation. Atmospheric N deposition can be estimated using a range of techniques including measurements, models and a combination of both. Techniques and uncertainties can also vary in estimating wet and dry deposition. However, obtaining accurate concentrations for both primary (NO_x and NH_3) and secondary components (NO_3^-, NH_4^+, HNO_3^-) are equally as important as understanding deposition processes (Sutton 2002).

Wet Deposition: Annual wet deposition can vary considerably at a site and any assessment of site deposition should be carried out over a number of years. The variation can be up to nearly 40% in some areas of the UK. Measuring wet deposition can be estimated at a site from simply measuring the amount and composition collected in an open funnel rain collector. However, these collectors can also capture dry deposition and in areas of high concentrations this may provide a positive result. This capturing process may occur at lower concentrations for highly reactive gases like ammonia. As measurements of wet deposition are often underestimated corrections are made using records of precipitation from meteorological data.

Dry Deposition: Continuous dry deposition modelling is done at only a few sites and is a much more complex than measuring wet deposition. For this reason intensive measurement regimes (every half hour) are used to develop models that can estimate dry deposition processes. By using atmospheric concentrations a model of dry deposition of nitrogen species (based on Ohm's law), has been developed whereby the total deposition is estimated from the potential difference in concentrations and the resistance to turbulent transfer. Resistances include turbulent aerodynamic, boundary layer and canopy resistances. The deposition velocity is generated from the sums of the aerodynamic resistance, the laminar boundary layer resistance and the surface resistance. However, this can be complicated more by the inclusion of the "ammonia compensation point" where emissions from vegetation have to be taken into account.

Other factors that control these type of models include local wind speeds and surface roughness, atmospheric stability, wetness, humidity and canopy structure. Differences in factors like canopy roughness and stomatal resistance leads to different deposition velocities between types of vegetation. For example, forest canopy being much rougher than shorter vegetation, like grassland, has a higher deposition velocity. For this reason UK deposition maps are produced based on vegetation type and are calculated for forest and semi-natural vegetation. Using grid average values of the total deposition would give an over estimation for semi-natural vegetation and an under estimation for forests. Any variation within a grid square is therefore dependant on the mix of land-uses with lower variation and uncertainties found where the grid cell is more or less entirely made up of one land-use type.

Similar hypotheses can be applied to a A/SSSI, where, at the edges of a site, surrounded by an agriculture land-use, deposition will be much higher than in the middle of the site. Fifty metre grid modelling carried out by Dragosits *et al*. (2002) found that dry deposition could be as much as 5 times larger at the edges than at the centre of a semi-natural site.

2.2.3 Local quality assurance of the deposition estimate

Assessing the potential impacts of N or acid deposition to a A/SSSI using only two categories of vegetation structures (forest or semi-natural) is rather limiting. Further uncertainties arise where A/SSSIs are made up of multiple habitat types of differing structure. For some pollutants like ammonia, where the spatial variation is so high, it is impractical to incorporate dense networks for measuring concentrations or deposition. In these cases it is recommended that concentrations are obtained from a less dense network and that simple models are used to assist with the interpolation process. Calculations can then be made for deposition by using local estimates of rainfall and land-use specific deposition velocity rates.

Some simple models like the SCAIL model (Simple Calculation of Ammonia Impact Limits) (Theobald *et al.*, 2001) can provide quick estimates of deposition of ammonia to a receptor site from a local source. The model is used to define the concentration profile downwind from a source, and takes into account the wind direction distribution, emission source size, receptor type and the type of land-cover between the source and the receptor.

There will be variation across larger sites which makes within site monitoring at single locations problematic. However the following may help to build confidence in use of the UK estimates of deposition at a site level.

Wet deposition:

(a) It is possible to measure wet deposition directly. This can be achieved by measuring rainfall and rain ion concentration in two different funnels. Although it is important to note that measuring over short time scales may be insufficient to capture the large between-year variability.
(b) Rain ion concentration can be measured alone, and the national rainfall map is used to generate wet deposition; or by comparing the rain ion concentrations with concentrations from the national maps it is possible to build up a level of confidence in the wet deposition estimate
(c) If appropriate, stream chemistry can be used to calculate catchment runoff for the site, particularly for sulphur. However, there are problems with other forms of diffuse pollutant sources within catchments, e.g. nitrates, etc.

Dry deposition:

(a) If of sufficient importance, it is possible to monitor dry deposition at a site using micro-meteorological techniques. For longer term sampling at low cost, the Time Average Gradient (TAG) sampling system is one possibility (Famulari pers. comm. 2005). It is most suitable for semi-natural vegetation. TAG sampling can provide direct long term average flux-gradient measurements for a range of trace gas species between atmosphere and terrestrial surfaces.
(b) Monitoring of gas and particle concentrations at the site will improve the estimate of dry deposition, calculated using either a deposition velocity or, particularly for ammonia, a canopy compensation point model (Sutton 1994; Smith *et al.* 2001).

Deposition estimates from the national maps are provided by land use. Unless the 5km square has large topographic variation, e.g. on the side of a mountain, these land use specific estimates are probably as good as using a local scale model. However, this is not the case for NO_x close to roads or NH_3 close to intensive farming activity. Clearly major pollutant sources should be modelled separately where appropriate, e.g. close to large power station. There will be decreasing deposition of air pollution within large blocks of woodland, but there will also be increases and decreases in pollutant deposition generated by complex landscapes, e.g. hedges beside fields or shelter belts. Generally, at present, these effects are not well quantified. There is scope for expert judgement in improving the deposition estimates, but there would have to be agreement of procedures beforehand to allow for adequate audit and support for the conclusions. An alternative is to invest in local scale validated models.

2.2.4 Critical loads

Critical loads data and maps are available nationally on a $1km^2$ grid for Biodiversity Action Plan (BAP) Broad Habitats that are sensitive to acidification and eutrophication and for which sufficient data exist to map their distribution nationally (Table 2.1). The methods used to map the Broad Habitats and to calculate critical loads are described in Hall *et al.* (2003, 2004) and will not be repeated in full here.

Habitats are mapped using a combination of CEH Land Cover data (Fuller *et al*., 2002) and ancillary data sets on species distributions, soils and altitude. It should be noted that the habitat maps used do not include every small area of sensitive habitat at the regional or local scale, but they do give a *national* picture of the main habitat types, adequate for *national* critical loads mapping purposes. There are uncertainties associated with the maps, due to uncertainties in the underlying data sets, the resolution of the data and the methods used to combine the data sets. Whilst these uncertainties can be identified it is not possible to quantify them. This subject has been examined in detail for the Environment Agency and is reported in Skeffington *et al*. (2006).

Table 2.1. Habitats for which acidity and nutrient nitrogen critical loads are mapped nationally

BAP Broad Habitat	**Critical loads Broad Habitat maps**	**Critical loads mapped for:**	
		Acidity	Nutrient N
Broadleaved, mixed and yew woodland	Managed broadleaved woodland Unmanaged broadleaved & coniferous woodland	✓ ✓	✓ ✓
Coniferous woodland	Managed coniferous woodland	✓	✓
Calcareous grassland	Calcareous grassland	✓	✓
Acid grassland	Acid grassland	✓	✓
Dwarf shrub heath	Dwarf shrub heath	✓	✓
Bog	Bog	✓	✓
Montane	Montane (Racomitrium heath)	✓	✓
Supralittoral sediment	Dune grassland	✗	✓
Lakes/Rivers	Selected sites only*	✓	✗

* Acidity critical loads calculated for 1722 selected sites (upland streams/lakes) only across the UK.

2.2.5 Acidity critical loads

Two methods are used for calculating acidity critical loads for terrestrial habitats in the UK: the empirical approach applied to non-woodland habitats and the simple mass balance (SMB) equation applied to both managed and unmanaged woodland habitats. Both methods provide critical loads for systems at long-term steady-state. Exceedance of these critical loads therefore demonstrates the *potential* for harmful effects to a system at steady-state.

The empirical approach is based upon the mineralogy and weathering rate of the dominant soil in each 1km^2, specifically the dominant soil series within the dominant soil association found in each 1km^2. Mineral weathering in soils provides the main long-term sink for deposited acidity. Using this principle, critical loads of acidity can be based on the amount of acid deposition which could be buffered by the annual production of base cations from mineral weathering. However, this approach is not appropriate for peat soils which contain little mineral material. For these the critical load is based on the amount of acid deposition that would prevent the soil solution pH falling below pH 4.4. Thus critical loads for peat soils are calculated using a formulation of the mass balance equation with the soil solution pH as the chemical criterion, and setting the leaching of aluminium and base cation weathering to zero. This method is suitable for acid peat soils, but not for some lowland/arable fen peats, some of which (eg in parts of Lincolnshire and Cambridgeshire) develop in calcareous conditions and so are less sensitive to acidification. Peat soils in lowland arable landscapes are assumed to be of this type and were given a high critical load on the national map (4.0 keq ha^{-1} year^{-1}), (Hornung *et al*., 1995b).

For managed and unmanaged woodland (broadleaved and coniferous) a simple mass balance model is applied. The objective is to balance 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 *et al*., 1990; Sverdrup & de Vries, 1994). A critical molar ratio of calcium to aluminium of one in soil solution is used as the critical chemical criterion, set to protect the fine roots of trees (UBA, 2004; Cronan & Grigal, 1995). For the managed woodlands, it is assumed that both phosphate and potassium fertilizers are applied and these contribute to the base cation budget. Calcium is removed from the system by the harvesting and removal of trees.

Data, methods and appropriate chemical criteria required for the calculation of critical loads have only been agreed for those habitat types listed in Table 2.1. The limitations of, and uncertainties in, the national critical loads data have been the subject of extensive research in recent years and are reported in Hall *et al*. (2004b) and Skeffington *et al*. (2006). Uncertainties are due mainly to the uncertainties in the individual underlying data sets and the paucity of UK data on which to base appropriate chemical criteria and some model input parameters. However, the national maps are based on the best available data and the methods used consistent with those developed under the Convention on Long-Range Transboundary Air Pollution (UBA, 2004). The national critical loads data were last updated in February 2004; no further updates are currently planned. The data are available free of charge on request from CEH Monks Wood and will shortly be available directly from the UK National Focal Centre web site (http://critloads.ceh.ac.uk).

2.2.6 Nutrient nitrogen critical loads

Critical loads are calculated nationally for a number of Biodiversity Action Plan Broad Habitats sensitive to eutrophication (Table 2.1). Two approaches are used in the setting of nutrient nitrogen critical loads, empirical and mass balance. For the empirical approach critical loads are estimated for different habitat types based on experimental or field evidence of thresholds for change in species composition, plant vitality or soil processes. The mass balance approach is based on balancing the long-term inputs and outputs of nitrogen from a system, to derive a critical load to prevent exceedance of a pre-defined acceptable level of nitrogen leaching. The mass balance approach is suitable for systems where the necessary inputs can be quantified and the key concern is nitrate leaching. In the UK this method is applied to managed woodland habitats and the empirical method is applied to natural and semi-natural habitats where the long-term protection of biodiversity and/or ecosystem function are the key concerns.

The empirical critical loads for nutrient nitrogen were reviewed and updated at a UNECE CLRTAP Workshop in 2002 (Achermann & Bobbink, 2003). Ranges of critical load values were assigned to habitat classes of the European Nature Information System (EUNIS) to enable consistency of habitat terminology and understanding across Europe. For the UK the EUNIS habitats were translated into Biodiversity Action Plan Broad Habitats and a single critical load "mapping" value assigned within each range for use in mapping critical loads and calculating exceedances (Hall *et al*., 2003). UK experts were actively involved in the Workshop and in deriving the UK mapping values. Table 2.2 summarises the ranges and mapping values for the EUNIS classes applicable to the UK, together with their corresponding UK habitat types. The potential impact of using the upper or lower value of these ranges, or the agreed mapping value, in the calculation of exceedances is explored in Section 2.2.8. The reliability of the empirical nitrogen critical load ranges is expressed as follows by Achermann & Bobbink (2003):

Reliable when a number of published papers of various studies showed comparable results;
Quite reliable when results of some studies were comparable;
(#) *Expert judgement* when no empirical data were available for this type of ecosystem. For this, the nitrogen critical load was based upon expert judgement and knowledge of ecosystems which were likely to be comparable with this ecosystem.

The national critical loads data were last updated in February 2004; no further updates are currently planned. The data are available free of charge on request from CEH Monks Wood and will shortly be available directly from the UK National Focal Centre web site (http://critloads.ceh.ac.uk).

Table 2.2 Empirical nutrient nitrogen critical loads by habitat for the UK

Broad Habitat[1]	EUNIS class	Critical load range[2] (kg N ha^{-1} $year^{-1}$)	UK mapping value (kg N ha^{-1} $year^{-1}$)
Acid grassland	Dry acid & neutral closed grassland (E1.7) Moist or wet oligotrophic grassland (E3.5)	10-20 # 10-20 #	15 15
Calcareous grassland	Semi-dry calcareous grassland (E1.26)	15-25 ##	20
Dwarf shrub heathland	(Lowland) dry heaths (F4.2) (Lowland) *Erica* wet heaths (F4.11) (Upland) *Calluna* wet heaths (F4.11)	10-20 ## 10-25 # 10-20 (#)	12 15 15
Bogs	Raised and blanket bogs (D1)	5-10 ##	10
Montane	Moss & 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 & 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) Stable dune grassland (B1.4)	10-20 # 10-20 #	15 15

Notes:

[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 & 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

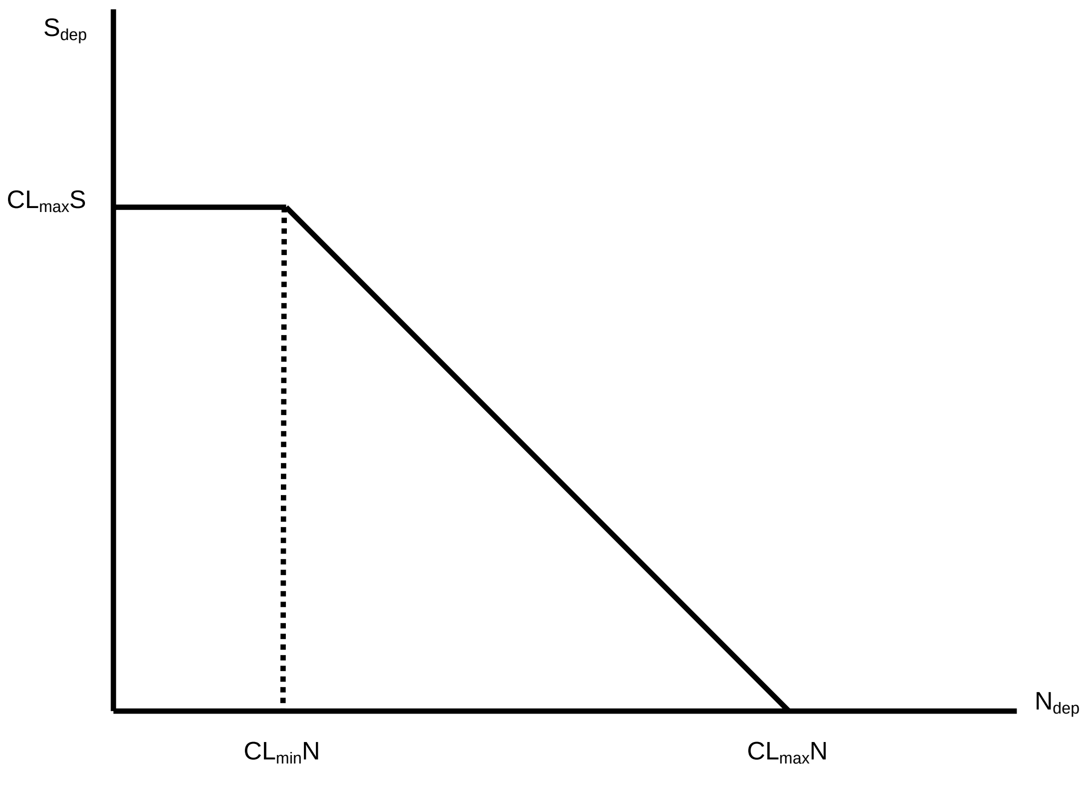
2.2.7 Exceedance of critical loads

The amount of excess deposition above the critical load is called the exceedance. Critical load exceedances are calculated separately for each 1km grid square in which each habitat is mapped nationally. The deposition values (described in Section 2.2.1) are assumed to be constant across the whole of the 5km square. For nutrient nitrogen, the exceedance is the amount of excess total nitrogen deposition above the critical load:

$$(CL_{nut}N)_{exc} = N_{dep} - CL_{nut}N$$

Deposition of both sulphur and nitrogen compounds can contribute to exceedance of acidity critical loads. The Critical Load Function (CLF), developed under the UNECE CLRTAP (Posch *et al.*, 1999; Posch & Hettelingh, 1997; Posch *et al.*, 1995; Hettelingh *et a.l*, 1995) defines combinations of sulphur and nitrogen deposition that will not cause harmful effects. The CLF is a three-node line graph representing the acidity critical load (Figure 2.1).

Figure 2.1 The acidity Critical Load Function (CLF)



The intercepts of the CLF on the sulphur and nitrogen axes define the following critical load values:

(i) The maximum critical load of sulphur ($CL_{max}S$) is the acidity critical load in terms of sulphur only, ie, when nitrogen deposition is zero. $CL_{max}S$ is based on the acidity critical load for the habitat (see Hall *et al.*, 2004a) but also takes into account the base cation deposition to the soil system and base cation removal from the system:

$CL_{max}S = CL_A + BC_{dep*} - BC_u$
Where:
CL_A = the acidity critical load for the habitat
BC_{dep*} = non-marine base cation deposition minus non-marine chloride deposition
BC_u = base cation removal and uptake by vegetation

(ii) The minimum critical load of nitrogen ($CL_{min}N$) is the sum of the long-term nitrogen processes in the soil:

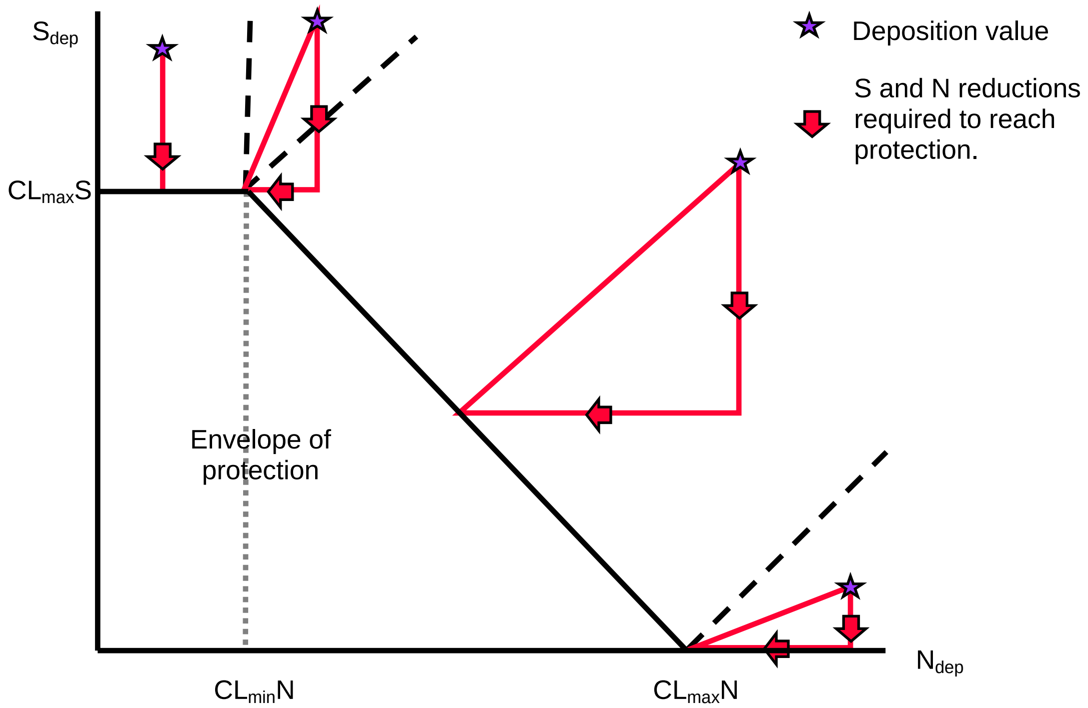
$CL_{min}N = N_u + N_i + N_{de}$
Where:
N_u = nitrogen removal and uptake by vegetation
N_i = nitrogen immobilisation
N_{de} = denitrification

(iii) The maximum critical load of nitrogen ($CL_{max}N$) is the acidity critical load in terms of nitrogen only, ie, when sulphur deposition is zero. It is calculated as:

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

Combinations of sulphur and nitrogen deposition above the CLF exceed the critical load, while all areas on or below the CLF line represent an “envelope of protection” where critical loads are not exceeded (Figure 2.2). Using the CLF exceedances are calculated for the habitat critical load values in each 1km square in which they occur across the country. Figure 2.2 shows the principles of the calculations; the “shortest distance” exceedance is comprised of the amount of sulphur plus nitrogen deposition reductions required to reach the CLF line.

Figure 2.2 Calculating exceedances of acidity critical loads using the CLF



The exceedance values can be presented for each habitat as 1km resolution maps. Alternatively maps that combine all habitat types can be generated by using percentile critical loads in the exceedance calculations, ie, critical loads data that combine the values for all habitats to give a critical load for each 1km square that will protect a percentage of the total habitat area in a grid square. For example, the 5th percentile critical load will protect 95% of the total habitat area within a square. Whilst maps are useful for providing the spatial patterns of exceedance, they are less useful when comparing the results of different deposition scenarios, since the maps may look very similar to one another. Alternatively the data can be used to provide summary statistics such as:

- The area of sensitive habitats exceeded (by country and UK)
- The percentage area of sensitive habitats exceeded (by country and UK)
- The Accumulated Exceedance: AE (by country and UK)

AE is calculated as:

AE (keq year^{-1}) = exceedance (keq ha^{-1} year^{-1}) * exceeded area (ha)

Thus AE is a measure of both the magnitude of exceedance and the area exceeded. This can be a more useful term to consider when comparing scenarios, since different scenarios may lead to the same area of habitat being exceeded, but the magnitude of exceedance may be different.

The national exceedance calculations therefore provide information on the area of sensitive habitats in the 1km squares where the critical load values are exceeded.

These values are summed to provide the statistics by habitat and country. To provide an estimate of the area of designated sites exceeded the following procedure is used:

(i) The digital files of the boundaries of A/SSSIs, SACs and SPAs are converted from their polygon boundaries to 1km gridded data sets; each 1km square then has the area of A/SSSI, SAC and SPA associated with it. Note that this means the area may represent more than one site in a 1km square, or the area of a single site may be divided between more than one square. This process enables the calculations below to be carried out quickly for the routine analyses required by Defra.

(ii) The national exceedance programs identify the 1km squares in which ***any*** of the terrestrial habitat critical loads are exceeded.

(iii) The area of A/SSSIs, SACs, SPAs associated with the exceeded squares are summed to give values by country and for the UK. It should be noted that the total area of designated sites quoted in Tables 2.3 and 2.4 below only include areas of sites within 1km grid squares for which we map critical loads for terrestrial habitats.

The exceedance statistics for the UK based on exceedance of critical loads by current measured deposition for 2001-2003 are given in Tables 2.3 and 2.4 below.

Table 2.3 Exceedance statistics for acidity for designated sites in the UK

Site type	Area (km^2)#	Exceeded area (km^2)	Percentage exceeded area
A/SSSIs	16740	11536	68.9%
SACs	10860	7600	70.0%
SPAs	8531	5774	67.7%

This is the area of designated sites that occur in 1km grid squares of the UK for which critical loads for terrestrial habitats are mapped.

Table 2.4 Exceedance statistics for nutrient nitrogen for designated sites in the UK

Site type	Area (km^2)#	Exceeded area (km^2)	Percentage exceeded area
A/SSSIs	21061	14191	67.4%
SACs	14625	9144	62.5%
SPAs	12119	7081	58.4%

This is the area of designated sites that occur in 1km grid squares of the UK for which critical loads for terrestrial habitats are mapped.

To date the number of designated sites exceeded has not been calculated due to the structure of the available digital site data. For example, there was no easy way to deal with sites consisting of multiple land parcels. More recent boundary data has both polygon (ie, land parcel) and site identifiers, which could enable information to be summarised at the site level, however there are still differences in the structure of the data between countries (ie, conservation agencies). CEH will investigate developing a method to provide exceedance statistics for the number of sites exceeded (wholly or partially) in addition to the area.

2.2.8 The sensitivity of nutrient nitrogen exceedances to critical load value

Section 2.2.6 describes the empirical critical loads of nutrient nitrogen and presents the critical load ranges and the associated mapping values used in the national calculations of critical loads exceedances. This section presents the effect on the habitat area exceeded of using the upper, lower or mapping critical load value in these calculations. The habitat used in this example is acid grassland, which has a critical load range of 10-20 kg N ha^{-1} year^{-1} and a mapping value that is in the middle of this range, ie, 15 kg kg N ha^{-1} year^{-1}. Exceedances were calculated using the 2001-2003 mean total nitrogen deposition for moorland and the results are shown in Table 2.5 below.

Table 2.5 Exceedance statistics for the UK for acid grassland

Exceedance parameter	Exceedance statistics based on a critical load (kg N ha^{-1} year^{-1}) of:		
	10	15	20
Mean exceedance$^{\#}$ (kg N ha^{-1} year^{-1})	0.05	0.03	0.02
Mean accumulated exceedance (kg N year^{-1})$^{\#}$	52824	26063	298
Mean ratio (N_{dep} / $CL_{nut}N$)$^{\#}$	1.9	1.4	1.2
Exceeded area (km^2)	11594	8847	4237
Percentage exceeded area	76.1%	58.0%	27.8%

$^{\#}$ based on UK exceeded areas only

The national exceedance statistics used by Defra are based on the mid-range mapping value of 15 kg N ha^{-1} year^{-1}. If the critical load at the lower end of the range were used the area of acid grassland exceeded across the UK would increase by 18.1% and if the upper range value were used the area exceeded would decrease by 30.2%. This illustrates that where within the range the critical load value is set can have a significant impact on the exceedance results. At present there is little additional information to help in selecting the appropriate value within the range and the mapping values were set for national assessments. Achermann & Bobbink (2003) suggest a simple methodology for setting the critical load at the upper, lower or middle part of the range based on temperature, frost period, soil wetness, base cation availability, phosphorous limitation and management intensity, but it is unclear whether this guidance is aimed at the whole of Europe (which would suggest using mid-range values for much of the UK), or for individual country applications. Ashmore and Hicks (2005) have discussed the principles for developing a decision support tool that incorporates the above factors and will be preparing a draft report of their proposed methodology to Defra in March 2006.

2.3 Atmospheric concentrations

Estimates of atmospheric concentrations rather than deposition have benefited from a large number of monitoring stations contributing to the UK National Air Quality Information Archive.

Concentration maps are derived from a number of models and the application of statistical methods such as kriging. Models are often used to generate concentration (and deposition) maps from source emissions and atmospheric transport and chemistry. These models can be local air dispersion models (e.g. ADMS) or long range transport models (e.g. FRAME www.frame.ceh.ac.uk), and their outputs can be compared to measured data from the monitoring networks.

The UK National Air Quality Information Archive (www.airquality.co.uk) was developed by NETCEN, part of AEA Technology Environment, on behalf Defra and the Devolved Administrations. Some data go back as far as 1960.

There are over 1500 sites across the UK where air quality is monitored. They are organised into networks that gather information on a wide number of pollutants.

Non-Automatic Networks

- NO_2 Diffusion Tube Network
- Lead and multi-element monitoring network
- Smoke and Sulphur Dioxide Monitoring Network
- Toxic Organic Micro Pollutants
- Acid Deposition Monitoring Network
- Rural Sulphur Dioxide Monitoring Network

Automatic Networks

- the Automatic Urban Network (AUN)
- the Automatic Rural Network (ARN)
- the Hydrocarbon Network

The networks provide measured hourly, daily and annual pollutant concentrations, and statistics, including air pollutant concentrations for individual sites, maps of pollutant emissions and acid deposition data. The majority of the 1500 sites are in urban areas and are used for monitoring air quality, predominantly to protect human health. However, they provide valuable measurements used in calculating pollutant maps for the UK and in calibrating modelled outputs. Data from the networks, primarily from the approximately 20 to 40 rural network sites depending on pollutant, are used in calculating UK concentrations maps for NO_2, NOx, SO_2 and O_3 as well as deposition maps for acidity and nitrogen.

2.3.1 Uncertainties in atmospheric concentrations

Monitoring data are useful when assessing pollution levels at a specific site, or at sites quite close to a monitoring station. For most pollutants, there are less than 50 “rural” sites in a network across the UK, so there are always large areas where there is no local monitoring. In consequence, local site factors are not taken into account in estimating concentration or deposition and therefore, combined with uncertainty in interpolation, any predicted exceedance of the critical level or load can be quite uncertain.

As an illustration, the decline in concentration of NH_3 with distance from a major source such as a pig or poultry unit is quite rapid and is on a scale of the order of metres to a few kilometres; likewise the effect of road traffic on the NO_2 concentration has a local scale commonly of less than 1km. These geographic scales are not realistically represented on maps of the mean concentration over a 5km x 5km grid square. Another issue occurs where large areas of the East Midlands show low AOT40 exposures and low mean concentration. This is due to the fact that the monitoring station in the Trent valley is situated near major roads and urban areas. Subsequently it is measuring lower concentrations of ozone then expected due to the scavenging effect of ozone by NO emitted by vehicular and other combustion sources, so the monitoring site, although representative of its local environment, is less representative of the regional scale ozone pollution climate.

Distance from a monitoring station can be very important when trying to compare concentrations at a A/SSSI site, as a result of the inherent uncertainty in interpolating between monitoring locations. As an example, Smith (2001) compared sulphate concentrations in rain at various distances from monitoring stations, particularly for estimating inputs to the UK Acid Waters Monitoring sites. Although it is expected that sulphate concentrations in rain will vary slowly across the landscape, the coefficient of variation approximately doubled from 15% close to a monitoring site to 30% at 30-40km from a monitoring site. These translate into approximate 95% confidence intervals for interpolated concentrations within ±30% of the mean close to sites to ±60% of the mean for sites up to 40 km. The uncertainty in the wet deposition estimate will be greater than that for the concentration estimate, with the increased uncertainty depending on the rainfall properties. To aim for an uncertainty in deposition within ±50%, it was suggested that sites should be located where possible within 40km of another monitoring location.

Most air quality monitoring is targeted at measuring exposure to humans and assessing levels against national air quality objectives. Hence, the majority of monitoring stations are located to measure concentrations around areas where people are exposed to high concentrations, such as close to emissions sources in urban areas and adjacent to busy roads. There are also urban background monitoring sites that measure away from potential emission sources but still within urban areas. They do not generally give a good estimate of air quality at nature conservation sites or other semi-natural areas which are often (but not always) situated in rural locations, but may be useful to provide better estimates of air pollution at the urban fringe.

Generally in rural areas, monitoring stations are set up in areas that are not directly affected by local emitters and are used to provide a relatively slowly varying background rural concentration field. Therefore it will often be necessary to model the effect of local process emissions to assess the expected level of pollution at any target location, or to use local expert knowledge. It is also important when assessing process contributions and their impact on conservation sites to make appropriate adjustments for local source effects on the background concentrations both (a) to avoid any double counting of the effect of local industrial processes and (b) to include effects of increased local emissions from other sources, such as, for example, those associated with close proximity to roads or intensive agriculture.

2.3.2 Ammonia

Ammonia (NH_3) in the atmosphere results primarily from the decomposition and volatilisation of animal wastes. As agricultural livestock numbers have dramatically increased, together with increases in nitrogen fertilization, NH_3 emissions have increased accordingly (Sutton *et al.,* 1993). As a result, some of the most acute problems of NH_3 deposition are for small relict sites located in intensive agricultural landscapes (Sutton *et al*., 1998). Other sources of ammonia emission include direct volatilisation from mineral fertilizers (particularly urea), agricultural crops and a wide range of non-agricultural sources including sewage, catalytic converters, seabirds and industrial processes (Sutton *et al.* 1995, 2000). Importantly, ammonia may also be emitted as well as deposited onto vegetation and the surface exchange of NH_3 can be treated as a bi-directional process (Sutton 2000). Ammonia has a very short life in the atmosphere (<10-100 km), and as a result concentrations show great local variability. Typically most of the pollution gradient is in the first few km of the source. Atmospheric ammonia has impacts on both local and international (transboundary) scales. In the atmosphere ammonia reacts with acid pollutants such as the products of SO_2 and NO_X emissions to produce fine ammonium (NH_4^+) containing aerosol.

2.3.3 Ammonia concentrations

The UK National Ammonia Monitoring Network is managed by CEH and funded by Defra. The ammonia network adopts a two tiered approach to monitoring both ammonia (NH_3) and ammonium (NH_4^+). A baseline network of around 50 sites samples using active denuders (where power is available), while a secondary network of passive diffusion tubes explores air concentration variability in high concentration areas, with the method calibrated at 10 sites against the denuder approach. Sampling is on a monthly basis which allows seasonal patterns to be investigated, while allowing sufficient resources for long term trends and spatial patterns to be assessed. The network website (www.cara.ceh.ac.uk) also provides interpolated concentration maps for the UK. As local variation in ammonia concentrations close to sources is high, instead of using just a relatively smooth interpolation an alternative uses FRAME, a multi-layer atmospheric transport model using emissions, to generate a UK NH_3 concentration map with noticeable variability at the 5km x 5km scale. The output from FRAME is calibrated to the UK measurement network to calculate NH_3 deposition.

APIS currently holds UK concentration data for 1999 (resolution of 5km). 2002 data is now available and will be uploaded to the APIS site soon (www.apis.ac.uk).

A new web site will soon be launched by the Centre for Ecology and Hydrology (www.uk-pollutantdeposition.ceh.ac.uk) that will provide a full inventory of the latest data sets on a series of air pollutants. These will cover a number of pollutant species including NH_3, NH_4^+, NO_2, NO_3^-, HNO_3, SO_2, and SO_4^{2-}.

2.3.4 Ammonia critical levels

The aim of the critical level for ammonia is to protect the functioning of plants and plant communities and the long term effects are thought to be more significant than the short-term effects. Direct exposure to high concentrations of ammonia can lead to:

- Direct damage to sensitive species, for example, premature senescence and leaf loss.
- Suppression of root uptake of cations such as Ca, Mg and K leading to nutrient imbalances (DoE 1993).
- Reduced ability of stomata to close under drought conditions, leading to plant water stress (Van Hove *et al.*, 1991, Erisman and Draaijers,1995).
- Changes in the composition of groundflora, bryophyte and lichen communities. (Power *et al.* 1995).
- There may also be subtle changes in plant morphology, physiology and biochemistry which can not only increase growth, but also increases sensitivity to environmental factors such as wind, frost, drought and pests (e.g. increased tissue N concentrations can predispose plants to insect attack).

At present it is not possible to set critical levels for NH_3 for individual vegetation types, instead a single critical level for NH_3 has been assigned to protect all vegetation types, with different values estimated according to the time period of exposure. Critical levels have been set for hourly (3300 µg m^{-3}), daily (270 µg m^{-3}), monthly (23 µg m^{-3}) and annual (8 µg m^{-3}) periods. However, recent lichen work carried out around a poultry farm (Pitcairn *et al.*, 2003) has shown that the annual critical level of 8 µg m^{-3} is possibly too high. While nitrophyte lichen species increased with NH_3 concentrations, acidophytic species declined with NH_3 concentrations, with losses occurring at lower concentrations than the increase in nitrophytes. Moreover, trunk acidophyte communities survived up to around 8 µg m^{-3}, while twig acidophytes were completely lost above 2 µg m^{-3}. This is substantially less than the current UNECE critical level of an annual mean of 8 µg m^{-3}. A workshop on "Atmospheric ammonia: detecting emission changes and environmental impacts" is to be held on 4-6 December 2006; this meeting will review the current critical level for ammonia.

2.3.5 Oxides of Nitrogen

Nitrogen oxides are produced in combustion processes, partly from nitrogen compounds in the fuel, but mostly by direct combination of atmospheric oxygen and nitrogen in flames (e.g. power plants and traffic). The primary pollutant, directly emitted, is nitric oxide (NO), together with a small proportion of nitrogen dioxide (NO_2). NO and NO_2 are collectively known as NO_x because they are rapidly inter-converted during the day. NO_2 is split up by UV light to give NO and an O atom, which combines with molecular oxygen (O_2) to give ozone (O_3). During the day NO, NO_2 and O_3 all exist in a quasi-equilibrium depending on the amount of sunlight. At rural locations, away from sources of NO, most of the nitrogen oxides in the atmosphere are in the form of NO_2. Eventually, NO_2 is oxidised to nitric acid (HNO_3, vapour) which is absorbed directly at the ground, is converted into nitrate-containing particles, or dissolves in cloud droplets. Although nitric acid is rapidly absorbed on contact with surfaces (cloud droplets, soil or vegetation), the other nitrogen oxides are removed only rather slowly, and may travel many hundreds of km before their eventual conversion to nitric acid or nitrates. Consequently, emissions in one country will be deposited in others. The UK exports about three-quarters of its emissions of NO_x.

2.3.6 Nitrogen oxide concentrations

Maps of estimated background annual mean concentrations for NO_x and NO_2 are available for the year 2002 onwards and are at a 1 km x 1 km grid. Maps of projected concentrations are also available for NO_x and NO_2 for the years 2005 and 2010.

Maps for NO_x and NO_2 are made up of a combination of interpolated rural measured concentrations (from the Acid Deposition Monitoring Network), area source modelled values calibrated to the national measured concentrations, and contributions from major points sources using ADMS modelling (Stedman, 2001). An additional roadside contribution is added to account for locations close to road traffic sources.

APIS provides data on NO_2 for comparisons with critical levels of NO_x. On APIS NO_2 data is presented at a 5 km resolution, although the underlying methodology is the same as for the 1 km data set. In the near future NO_x 1km x 1km concentrations will be added to APIS to provide the correct comparison to assess exceedances of the critical level. Current critical level exceedances are being underestimated by using the mapped NO_2 concentrations from APIS, as the NO concentration has been ignored.

2.3.7 Oxides of nitrogen critical levels

The gases NO and NO_2 are considered together partly because their concentrations in air are inextricably linked through their atmospheric chemistry, and partly because little is known of the direct effects of NO alone. Emission controls are driven by its role as an ozone precursor rather than because of its direct effects. However, direct effects may occur in the immediate vicinity of major roads and in the centre of cities, caused by high NO_X emissions from vehicles. The main impacts on vegetation are:

- Visible injury symptoms for example, leaf discoloration.
- The vulnerability to direct damage of mosses, liverworts and lichens which receive their nutrients largely from the atmosphere.
- Changes in species composition

Critical levels for NO_x are based on the sum of both NO and NO_2 concentrations because there is insufficient knowledge to establish individual critical levels for each pollutant. There is some evidence that at low concentrations (ie, ambient) NO is more phytotoxic than NO_2. Increased growth of certain species, through a fertiliser effect, are of greatest concern for natural and semi-natural vegetation due to the likelihood of changes in competition between species. Due to a lack of information critical levels were not set for different types of vegetation; the 24-hour mean of 75 $\mu g\ m^{-3}$ and annual mean of 30 $\mu g\ m^{-3}$ (expressed as NO_2) are applied to all vegetation types.

However, for mapping critical levels and their exceedances it is strongly recommended that the annual mean is used as mapped and modelled values of this parameter have much greater reliability, and the long-term effects of NO_x are thought to be greater than the short-term effects (UBA, 2004).

2.3.8 Sulphur dioxide

The main sources of SO_2 emissions are coal-fired power stations, industry, shipping and domestic fuel combustion. Over the past 20 years there has been a reduction in the emissions from low level sources including domestic combustion, so that emissions are now dominated by a few large high level sources. However, reductions in emissions are not currently matched by linear reductions in deposition and impacts in the most sensitive areas of the UK. These impacts include both soil and freshwater acidification.

The spatial variability of SO_2 concentrations tends to be high, but not as high as that for ammonia. For rural areas away from local sources, this variability is largely caused by spatial and temporal changes in the degree to which individual sites are ‘vertically’ connected to the main reservoir in the boundary layer.

2.3.9 Sulphur dioxide concentrations

The latest SO_2 concentration data set for the UK is based on a 3 year average, 1999-2001, and is available from the APIS website.

Rural SO_2 measurements from the 32 stations are interpolated to provide a concentration field across the UK. This is then combined with an additional urban SO_2 component derived by a regression procedure to produce a final 5km by 5km concentration map for the UK.

An alternative method has been used to calculate the SO_2 concentration field for the UK by NETCEN, using a similar procedure as that for NO_x with local dispersion models providing a 1km x 1km national concentration field. When the two mapping methods were compared in 2003, it appeared that the method used in APIS gave a better representation in rural areas and hence for the purposes of assessment of protected nature conservation sites. Further improvements in the dispersion modelling, and in particular local modelling near point sources, may provide a better estimate of SO_2 concentration in the future.

2.3.10 Sulphur dioxide critical levels

The critical levels for SO_2 were established at Egham in 1992 (Ashmore and Wilson, 1993) and are still valid to this day (UBA, 2004). There are critical levels for four categories of receptor – sensitive lichens (10 $\mu g\ m^{-3}$), forests (20 $\mu g\ m^{-3}$), semi-natural vegetation (20 $\mu g\ m^{-3}$) and agricultural crops (30 $\mu g\ m^{-3}$). The SO_2 is expressed as average concentration that should not be exceeded for either the winter months

(October to March) or the year as a whole (annual mean). Characteristic impacts of an exceedance of the SO_2 critical level included impaired photosynthetic performance and growth, leaf discolouration and eventual decline of species.

2.3.11 Ozone

O_3 in the troposphere is regionally important as a toxic air pollutant. Concentrations have increased during this century. Mixing with stratospheric air provides a natural global average background of around 10-20 parts per billion (ppb), although there is some debate about the concentration. Additional quantities of tropospheric O_3 are produced by photochemical reactions of nitrogen oxides (NO_x) and volatile organic compounds (VOCs), which include various hydrocarbons. NO_x and VOCs originate from fossil fuel combustion and natural sources. As these emissions have increased since the mid 19th century, so tropospheric ozone has increased. The chemical cycles producing and destroying O_3 depend on pollutant and light levels so different reactions are established during the day and night. For a description of these reactions see PORG (1997) or NEGTAP (2001). In warm, summer conditions, photochemical events often occur in which O_3 concentrations increase successively over several days. These "ozone episodes" provide concentrations of O_3 (>40 ppb) which are toxic both to human health, buildings and wildlife.

2.3.12 Ozone exposure

The UK has a rural network of 20 stations and an urban network of over 50 stations for monitoring ozone. Interpolating between measurements from the rural networks is highly reliant on incorporating the effects of topography and windspeed. Annual mean concentrations for the UK are further modified to take into account diurnal cycles in ozone especially for the summer months, and further enhancements are then made to incorporate measurements from urban monitoring stations (see PORG, 1997). For effects of ozone on vegetation, the critical level is based on an accumulated exposure over a threshold of 40ppb (AOT40), which is mapped using a similar approach as for the mean ozone concentrations. Since the AOT40 method is mainly of interest to semi-natural and forest areas (i.e. rural areas), an urban correction is not applied.

The method used to map AOT40 is described in Coyle *et al.* (2002). It can be shown that during the afternoon, when the planetary boundary layer is well developed, rural monitoring sites observe concentrations that are representative of a wide geographical area (within approx. 100 km radius). Hence a map, based on a 1 km resolution, can be produced by interpolating the afternoon value of a variable from rural sites then modifying the resulting grid cell values to account for the diurnal cycle in ozone concentration. For the AOT40 this is done using the ratio of AOT40 during daylight hours to that during the afternoon period. This ratio is then related to the locations altitude (h) using the linear relationship:

(AOT40 daylight hours)/(AOT40 afternoon) = m.h +c,
where m is the slope of linear regression and c the intercept

The magnitude of the diurnal cycle in ozone concentration at a given location generally depends on the wind speed and atmospheric stability. However wind speed measurements are not available at all monitoring sites and so altitude is used as a surrogate. AOT40 values for each monitoring site are calculated from the ratified hourly measurements provided by the National Air Quality Archive (AEAT, Netcen).

Data are based on a five year average. New five year datasets are produced every 2-3 years. The 2001-2004 dataset is now available.

2.3.13 Ozone critical levels

The critical levels for ozone for forests and semi-natural vegetation are based on the AOT 40 exposure threshold. AOT40 is the Accumulated Concentration Over a Threshold of 40 ppb. If an hourly average ozone concentration exceeds 40 ppb the difference between the concentration and 40 ppb is added to a running total. For the critical levels the AOTs are summed during daylight hours over fixed time periods. To reduce the effect of inter-annual variability in ozone concentrations the critical level is assessed on the 5-year average AOT40 rather than an AOT40 for individual years. For forests the AOT40 is summed for April to September daylight hours inclusive, while for semi-natural vegetation the AOT40 is summed for May to July daylight hours inclusive. Daylight hours are defined as when solar radiation exceeds 50 W m^{-2}. Critical levels have been based on these cumulative exposures. The critical level for forests is 5000 ppb hours. The critical level for semi-natural vegetation has recently been updated by the International Cooperative Programme (ICP) on Vegetation and agreed by the ICP on Mapping and Modelling in April 2006; it has been decided that the current value of 3000 ppb hours (over 3 months) applied to semi-natural vegetation is most appropriate for communities dominated by annual species. For semi-natural vegetation communities dominated by perennial species a new critical level of 5000 ppb hours (over 6 months) is recommended. The background information relating to this decision will be published as an annex to Chapter 3 of the Mapping Manual (UBA, 2004).

Concentration-based critical levels have also been set for agricultural and horticultural crops. In addition, stomatal flux based critical levels are recommended for quantitative assessments of risk to wheat and potatoes. For visible injury to horticultural crops a vapour-pressure-deficit (VPD) modified concentration-based critical level is recommended (UBA, 2004).

Impacts of exceedance of the critical level include a reduction in annual increment for trees, reduced yields in crops, and species composition changes in semi-natural vegetation.

2.3.14 Exceedance of critical levels

The calculation of exceedance of critical levels is a straightforward comparison of the critical level and the pollutant concentration. As described in the sections above, depending on the pollutant the critical level can be an annual/hourly/daily mean for all vegetation types, or an annual mean for a particular vegetation type. The critical levels for the different pollutants are summarised in Table 2.6 below.

Table 2.6 Summary of critical levels[#]

Pollutant	Receptor	Time period	Critical level
NH_3	All vegetation types	Hourly	3300 µg m^{-3}
		Daily	270 µg m^{-3}
		Monthly	23 µg m^{-3}
		Annual mean	8 µg m^{-3}
NO_x (as NO_2)	All vegetation types	24 hour	75 µg m^{-3}
		Annual mean	30 µg m^{-3}
SO_2	Cyanobacterial lichens	Annual or half year (Oct-Mar)	10 µg m^{-3}
	Forest ecosystems[###]	Annual or half year (Oct-Mar)	20 µg m^{-3}
	Semi-natural vegetation	Annual or half year (Oct-Mar)	20 µg m^{-3}
	Agricultural crops	Annual or half year (Oct-Mar)	30 µg m^{-3}
Ozone	Forests	AOT40	5000 ppb hours
	Semi-natural vegetation[##] (a) dominated by annuals (b) dominated by perennials	 AOT40 (over 3 months) AOT40 (over 6 months)	 3000 ppb hours 5000 ppb hours

[#] Source: UBA, 2004
[##] Updated following decisions by ICP Vegetation and ICP Mapping & Modelling, April 2006.
[###] Includes forest understorey vegetation

Exceedance statistics for designated sites have been calculated for Defra, based on critical levels for NO_x and SO_2. As for exceedances of critical loads, the statistics currently give the area of designated sites exceeded rather than the number of sites. Some example results are given in Table 2.7 below. The critical levels used in this example are assumed to be applicable to all site and habitat types and the site areas refer to the total areas of designated sites in the UK. In these examples the areas of sites exceeded across the UK are relatively small.

Exceedance statistics for ozone have not been requested or calculated, but a similar approach could be applied to assess the potential impact of ozone on designated areas.

Table 2.7 Statistics of exceedance of critical levels based on 2003 concentration data (1km concentration data from Netcen)

Critical level	Exceedance parameter	A/SSSIs	SACs	SPAs
Exceedance of NO_x critical level (30 µg m^{-3})	Area of sites in UK (km^2)	24154	24111	14362
	Exceeded area UK (km^2)	416	125	722
	Percentage area exceeded	1.72%	0.52%	0.50%
Exceedance of SO_2 critical level (20 µg m^{-3})	Area of sites in UK (km^2)	24154	24111	14362
	Exceeded area UK (km^2)	0.76	0	0.72
	Percentage area exceeded	0.003%	0%	0.005%
Exceedance of SO_2 critical level (10 µg m^{-3})	Area of sites in UK (km^2)	24154	24111	14362
	Exceeded area UK (km^2)	29	2.4	19
	Percentage area exceeded	0.12%	0.01%	0.13%

It should be noted that these exceedance statistics are for the whole of the UK, including both urban and rural areas. The critical levels of NO_x and SO_2 (at 20 µg m ^{3}) have been adopted as the European limit values for the EC Air Quality Daughter Directive (DD) and as the air quality objectives for the UK Air Quality Strategy. Assessments undertaken for the DD by Stedman *et al.* (2005) show no exceedance of designated sites in rural areas of the UK for either pollutant using concentration data for 2003.

3. Summary of Site Relevant Critical Loads & Source Attribution

As part of the work to meet their obligations under the Habitats Regulations (Regs 48-51), the Environment Agency and SNIFFER (on behalf of SEPA and EHS) funded work to conduct a country-wide assessment of the extent to which Special Areas of Conservation (SACs) and Special Protection Areas (SPAs) may be under threat by current and future emissions of air pollution from major point and area sources. For any existing consent which was likely to have a significant effect on a European site (e.g. SAC or SPA), either individually or in combination with others, an appropriate assessment of the implications for the site was carried out.

The definition of site-relevant critical loads, or more strictly feature-relevant critical loads, represents the allocation of the most relevant critical load for every designated feature at a particular European site (SPA or SAC). Critical loads for both acidity and nutrient nitrogen are assigned for each designated feature that is sensitive to acidification or eutrophication. The data used were the February 2003 version of the national (UK) critical loads (Hall *et al.*, 2003). Levels of justification are given for allocating a particular critical load class and information is provided about the critical load values and impacts of an exceedance of this critical load. Other information is given about a suitable Biodiversity Action Plan (BAP) Broad Habitat type for each feature and notes describe feature ecology or habitat preferences (e.g. bird species).

3.1 Assigning Relevant Critical Loads to designated features

3.1.1 Assigning critical loads for acidity

There are eight habitat classifications for acidity critical loads (acid grassland, calcareous grassland, dwarf shrub heath, bog, montane, unmanaged coniferous and broadleaved woodland, managed deciduous woodland, and managed coniferous woodland), all based on BAP Broad Habitats (see Table 2.1). Only six of these were used in this assessment, which excluded managed coniferous and managed deciduous woodlands, since these are not protected within the network European designated sites.

The next step was to assess whether the designated features were sensitive to acidification. Once the sensitivity of each feature had been established, the Annex 1 habitat and Annex II plant species were matched to their corresponding critical load Broad Habitat. Justification for these linkages was noted as well as a description of an impacts of exceedance.

3.1.2 Freshwater acidity critical loads

Freshwater critical loads of acidity were also considered in the site relevant critical loads assessment. Critical loads for freshwaters, are based on steady state models and water chemistry, are set against a target organism, Salmo trutta (Brown trout). The acid neutralising capacity (ANC $\mu eq\ l^{-1}$) of the water is used as an indicator of the viability of the target organism, with this set originally at a critical value of 0 $\mu eq\ l^{-1}$, which matches to a 50% probability of reduced trout populations occurring (Curtis *et al.* 2003). However, it should be noted that in 2004 the critical ANC limit was re-considered and a value of 20 $\mu eq\ l^{-1}$ was applied to all UK freshwater sites included in the national data set, except those where site-specific information suggested they were naturally acidic, in which case the ANC limit of zero was retained (Hall *et al.,* 2004a). The mean acidity critical load values (ie, CLmaxS, CLminN, CLmaxN) for the freshwater sites in the 2004 data set were between 6.5% and 12.4% lower than the 2003 data set; however, (a) this does not necessarily mean that the critical loads for all freshwater sites decreased, and (b) this difference is not due to the alteration in the ANC limit alone, since the 2004 data set contains 1722 sites compared to only 1163 in 2003.

The results of Curtis *et al.* (2003) have been used in the assessment of freshwater critical loads here. In their report, Curtis *et al.* identified designated features associated with freshwaters and have also produced a list of potentially acid sensitive freshwater SAC and SPA sites where these features are found. This was carried out in three stages: 1) identifying designated features associated with freshwater habitats, 2) identifying the potentially sensitive features in terms of habitat sensitivity, and 3) a refinement of the risk assessment. A final shortlist of 32 SACs and 1 SPA were identified as being potentially sensitive together with sensitive features. The report only covers sites in England and Wales.

3.1.3 Assigning critical loads for nutrient nitrogen

A similar process to acidity critical loads was adopted to assign critical loads for nutrient nitrogen. Annex I habitat and Annex II plant species were linked with the most suitable EUNIS class used in classifying empirical critical load habitat classes. Similar justifications for this match were noted for transparency and consistency. Using the critical loads defined at the Bern Workshop (Achermann & Bobbink, 2003), the relevant critical load values, reliability of these values and the likely impacts of exceedance were all noted.

3.1.4 Freshwater nutrient nitrogen critical loads

Empirical critical loads for freshwater features are limited to oligotrophic inland surface waters, where a range of field and experimental evidence is available (Bobbink *et al.*, 1996). Since most surface waters, especially non-oligotrophic waters, in the UK are limited by phosphorous, nitrogen enrichment is considered unimportant.

3.1.5 Assigning critical loads for SPA features and SAC Annex II non-plant species

There are few, if any, instances of direct effects on bird species from nitrogen and acid deposition. Therefore, assigning critical loads directly to bird features is an unsuitable method for assessing likely impacts. However, examining the relation between a bird's integrity and that of its habitat provides for a better causal link between potential bird decline and atmospheric pollutant deposition. The same methodology was applied for Annex II non-plant species. Where the habitat in which a bird or non-plant species occurs was assessed to be insensitive to either acidity or eutrophication, no critical load was assigned.

For each species the following series of questions were applied:

1. What is the relevant BAP Broad Habitat for this species? For example, in the case of bird species - what is the habitat that the SPA bird species is dependent upon in terms of breeding, feeding or roosting? This often led to a bird species being dependant on more than one BAP broad habitat, and each instance was recognised as a new record.
2. Is this habitat sensitive to eutrophication (from atmospheric deposition) or acidification?
3. If yes, what are the impacts of eutrophication or acidification on this habitat and will it affect the viability of the breeding, feeding or sometime roosting of that species?
4. If there are potential negative effects on the species, the most relevant critical loads for nutrient nitrogen or critical load for acidity is assigned based on the broad habitat in which the species is present.
   Similar justifications were made as that for SAC features and the relevant critical load values and impacts were noted.

The decision flow diagram in Figure 3.1 shows an example of how the links between habitat impacts and the consequent bird feature impacts were assessed. The example describes the assignment of a critical load for nutrient nitrogen to the feature *Charadrius morinellus* (Dotterel).

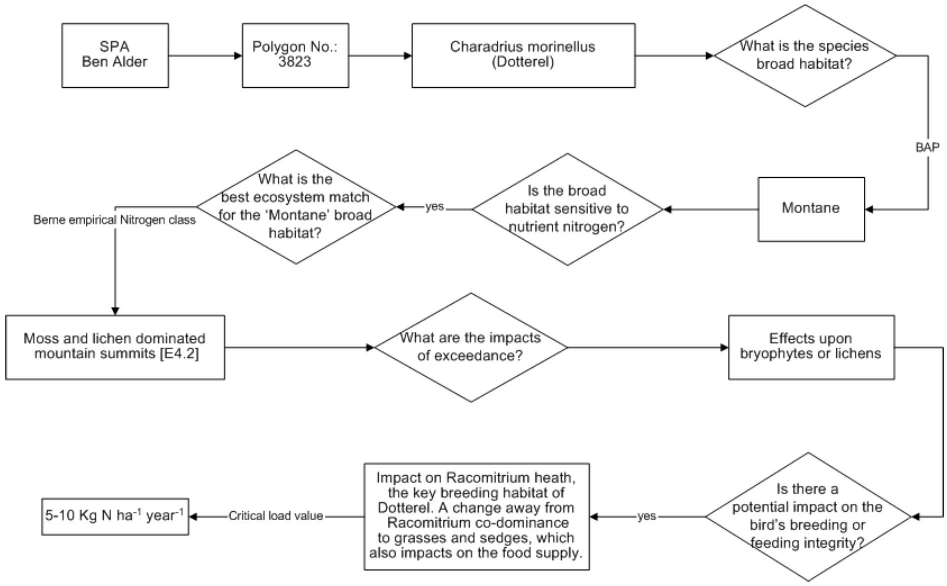


Figure 3.1 Decision flow diagram for assessing potential impact on SPA features.

3.1.6 Assigning acidity critical loads to site polygons

Each SAC and SPA site is made up of a number of GIS polygons or for smaller sites often just one polygon. Using ArcView GIS, the 1 km acidity critical loads for each of the six habitat classes, were overlaid on the site polygons to produce a grid of critical load values. By comparing the critical load values across each of the six 1 km grid squares, the maximum and minimum values for each polygon, and for each of the six broad habitat types, were determined. These maximum and minimum values, make up the CLmaxS, CLminN and CLmaxN components that comprise the critical load function (Section 2.2.7).

3.1.7 The database

By combining site information, including feature lists and polygon IDs, and the critical loads work, a database of site based critical loads by feature and polygon was built for each SAC and SPA in the UK network. The finished database was presented as an interactive Excel spreadsheet database. Some notes have been added to the Excel spreadsheet that provides guidance and instruction in the use of critical loads and the critical load function. The database will soon be incorporated into APIS. Users will be able to search by SAC/SPA and retrieve information on designated features (e.g. their sensitivities to acidification and eutrophication) including a breakdown of the sources that are contributing to nitrogen and acid deposition at any chosen site.

3.1.8 Limitations in using critical loads for site-based assessments

The application of critical loads to site based assessments should, in general, be based on the soil type and location on which the feature lies. Using national data sets in this exercise has led to a number of inconsistencies and limitations in the use of critical loads to assess site based habitat and species features.

1. Adopting the use of national critical loads for acidity gives rise to a number of problems including:
    - National maps for critical loads of acidity are based on the dominant soil type for each 1 km grid square. This leads to problems for all features that make up small areas or mosaics of habitats, but are not represented by the dominant soil type for that 1 km grid square.
    - Some features can be found on a number of different soil types, (e.g. ranging from calcareous to acidic) and are therefore represented by a number of different acidity classes. For example *Juniper communis* occurs on calcareous, acidic and montane soils. The critical load values will in most cases reflect the correct underlying soil type, but prior knowledge of the soil type on which the feature occurs on a particular site should also be taken into account.
    - Some features, which are sensitive to acidity, may have missing critical load values. This occurs particularly with coastal sites where the 1 km gridded national critical loads maps from which the site relevant values have been derived do not exactly match the real coastline.
2. For all sites there is the assumption that all features are in every polygon. Future improvements in the mapping of designated features to A/SSSI or polygon level (as has been initiated for SACs in England) would greatly improve the efficiency of the database and avoid duplication of feature accounts.
3. The use of critical loads in assessing the importance of exceedance and how this is relative to other drivers of ecological change impacting on the features like land management or climate change, is not tackled in this project. Integrating such drivers was never intended for the purposes of this exercise, which focuses on conducting a screening of likely sensitive feature and sites. Such drivers would certainly apply to decisions made when a more detailed site based assessment is conducted out. For example, many of the eutrophication effects need to consider the active land management context of grazing or mowing.
4. In the case of bird species this assessment does not take into account the ability for birds to select optimal locations, or the dependence additional foraging habitats during critical parts of the year. This is particularly important for agricultural habitats which need to be in the vicinity for many of the listed species. An example like the above is intrinsically linked with the entire process of SPA selection as all eventualities of a birds requirements should be taken into account.

4. Assigning Critical Loads/Levels to Notified Features

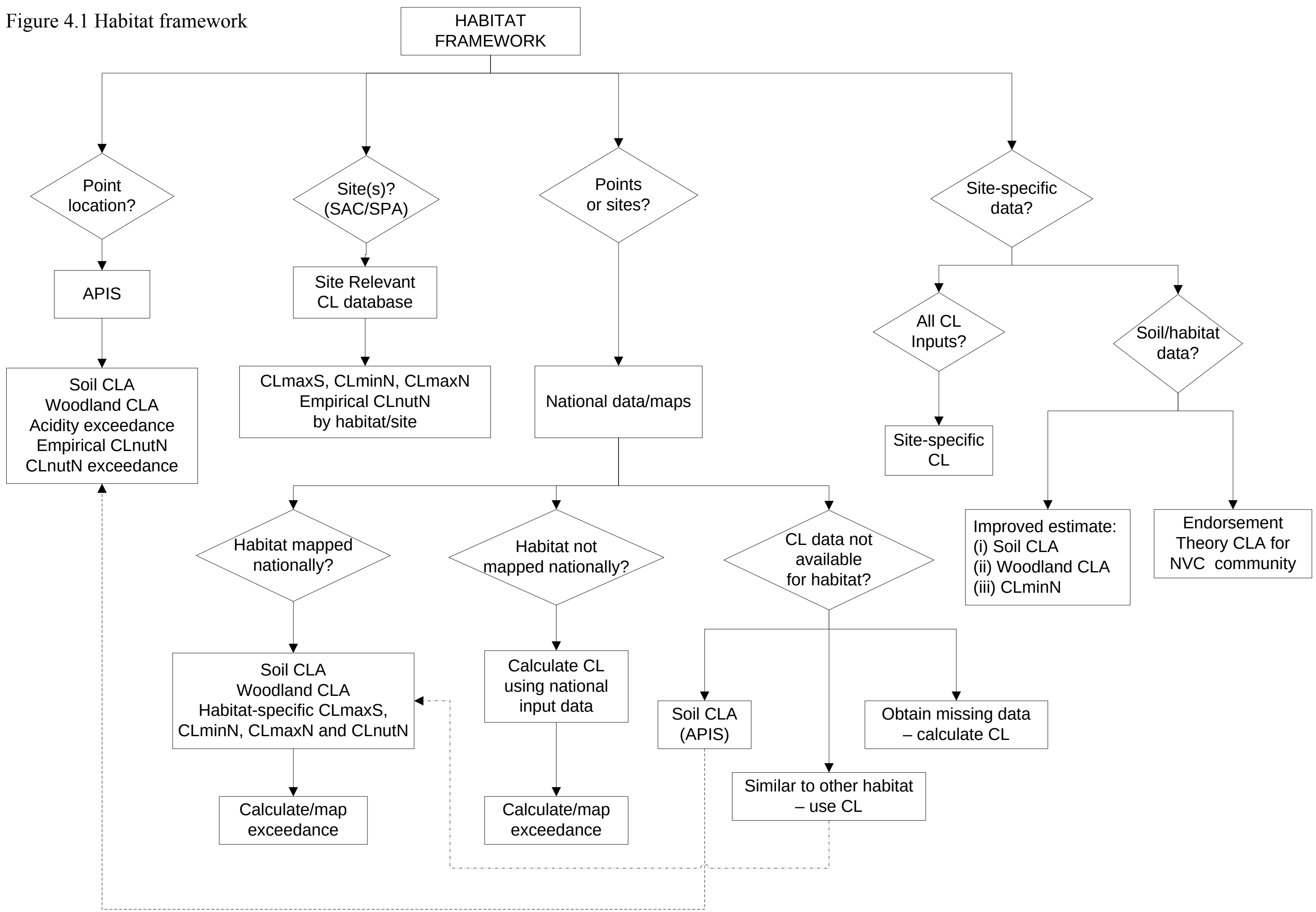
Section 3 described how critical loads were assigned to the designated features (habitats or species) of SACs and SPAs under earlier work jointly funded by SNIFFER and the Environment Agency. In this section we consider the methods used in Section 3, together with other options for assigning critical loads (and levels) to the interest features of A/SSSIs.

The reporting of feature condition by JNCC in 2006 will be based on a common list of notified interest features for A/SSSIs. This list is currently based on BAP broad habitats for habitat features and taxonomic groups for species features. In the UK there are a total of 6821 A/SSSIs which have a mix of earth science (geological), habitat and species features. Of the 6821 sites, 5043 have habitat features and approximately half of these (2509) have habitat features only, and as such should be easier to assign critical loads to. However, it will depend on whether or not (a) the habitats in question are sensitive to acidification or eutrophication; and (b) critical loads data/methods are available for the habitats. The remaining 2534 sites with habitat features additionally have species (and/or geological) features, so the assessments for these may be more complex. Under this study we will not be considering the geological features of the A/SSSIs. There are 2627 A/SSSIs with species features and of these 260 have species features only. Both habitats and species will need to be considered separately in any critical load assessment, since even where there are both habitats and species it could be possible that the notified species are not related to the notified habitat(s). For example, the notified habitat could be a heathland, and the notified species a flowering plant found only in woodland. In such cases setting a critical load to protect the habitat may not necessarily protect the species.

The methods that can be applied to set critical loads to habitats or species are outlined in Sections 4.1 and 4.2, and assigning critical levels to designated features is addressed in Section 4.3.

The first step in any assessment is to determine if the feature (habitat or species) is sensitive to the pollutant in question; APIS provides information on the potential impacts of pollutants on selected habitats and species, and the UK Biodiversity Action Plan (BAP) web site (http://www.ukbap.org.uk) may provide additional information on specific BAP broad or priority habitats.

Figure 4.1 Habitat framework



4.1 Sites with habitat interest features

This section outlines a number of options for assigning critical loads to habitat features; they are summarised in Figure 4.1.

4.1.1 UK Air Pollution Information System: http://www.apis.ac.uk

This system provides information on:

- Impacts of pollutants on habitats or species selected from a list
- Acidity critical loads, acid deposition and exceedance of critical loads
- Nutrient nitrogen critical loads, nitrogen deposition and exceedance of critical loads.

The user can select critical load and exceedance information for any habitat (available in the drop-down lists) for any single point location.

The critical loads data available via APIS are:

(a) Empirical soil acidity critical loads based on the dominant soil in a 1km grid square (see Section 2.2.5); these are the values presented when the user specifies a non-woodland terrestrial habitat. It should be noted that these are not habitat-specific critical loads; the critical load is set to protect the dominant soil and does not include any additional information related to habitat type.
(b) Mass balance acidity critical loads calculated for unmanaged woodland habitats and based on the dominant soil in a 1km grid square; these are the values presented when the user specifies a woodland habitat, they are not specific to woodland types or species. Values are available for all 1km grid squares in the UK for which there are sufficient data to calculate the critical loads, ie, they are not restricted to areas of mapped woodland, instead it is assumed the woodland habitats can occur in all 1km grid squares of the UK.
(c) Empirical nutrient nitrogen critical loads as defined for EUNIS habitats at the Berne workshop (Achermann & Bobbink, 2003).

The following points need to be considered when using APIS:

(i) The acidity critical load values are based on the dominant soil in a 1km grid square; other soil types may be more or less sensitive to acidification and have lower or higher critical loads. The feature (habitat/species) the user is interested in setting a critical load for may not occur on the dominant soil and hence the critical load value may be inappropriate.
(ii) As noted above the acidity critical loads for the non-woodland terrestrial habitats are set to protect the dominant soil and are not habitat-specific. The way APIS works this means that any non-woodland habitat selected within a 1km grid square will be assigned the same acidity critical load value. That value may not always be appropriate for the habitat in question, for example, if

both acid and calcareous grassland occur within the same grid square, APIS will return the same critical load value for both habitats.

(iii) APIS enables the user to select critical loads for any habitat type (of those listed) for any location, irrespective of whether the habitat occurs in that location, ie, it does not mean that habitat is present at that location. Whilst this point (and others relating to the derivation of critical load values) is made clear on the information associated with the critical load values provided, it is up to the user to apply the system appropriately.

(iv) APIS does not provide the habitat-specific critical load values CLmaxS, CLminN and CLmaxN required by the Critical Loads Function (CLF) for the calculation of exceedances. The acidity exceedances calculated by APIS are based on the soil acidity critical loads described above and total acid deposition (ie, the sum of non-marine sulphur, oxidised and reduced nitrogen). By comparison the CLF effectively uses “net acid deposition” via CLmaxS and CLminN. “Net acid deposition” can be calculated as:

(NMS + ((NOx + NHx) – (Nu + Ni + Nde))) – ((BCdep* - Cldep*) – Bcu)

where:

NMS = non-marine sulphur (based on moorland or woodland values)
NOx = oxidised nitrogen (based on moorland or woodland values)
NHx = reduced nitrogen (based on moorland or woodland values)
Nu = nitrogen uptake (dependant on habitat type)
Ni = nitrogen immobilisation (dependant on soil type)
Nde = denitrification (dependant on soil type)
BCdep* = non-marine base cations (based on moorland or woodland values)
Cldep* = non-marine chloride (based on moorland or woodland values)
BCu = base cation uptake (dependant on habitat type)

Net base cation deposition (ie, BCdep* - Cldep* - BCu) can range from zero to 0.4 keq ha^{-1} $year^{-1}$ and the nitrogen removal processes (ie, Nu + Ni + Nde) can range from 0.14 to 1.214 keq ha^{-1} $year^{-1}$. These values will vary for individual grid squares depending on soil and habitat types. In terms of the uptake values it should be noted that values have only been determined for the habitat types listed in Table 2.1 of this report.

Hence the acidity exceedances for the woodland habitats provided by APIS may be over-estimated. For the non-woodland habitats acidity exceedances are based only on the soil acidity critical loads and sulphur and nitrogen deposition and do not take into account any net base cation deposition or nitrogen removal processes.

(v) Data extraction from APIS is done manually for individual point locations; this approach is fine when only a small number of sites/points require assessment. However, it would be time consuming (and tedious) to use this system for assessing large numbers of sites/points/habitats. APIS could be developed further to provide multiple searches, but limits on the amount of searches in any one session would have to be offset against server usage required by other users. However, multiple searches could be set up to run at times of the day that did not affect normal use by others (e.g. during the night).

4.1.2 Site relevant critical loads

This database was generated under the “Site relevant critical loads and source attribution” project described in Section 3 of this report. The database provides:

- Acidity critical loads (in terms of CLmaxS, CLminN and CLmaxN) for each designated feature (habitat/species) for each SAC and SPA across the UK.
- Empirical nutrient nitrogen critical loads as defined for EUNIS habitats at the Berne workshop (Achermann & Bobbink, 2003).

The minimum and maximum values of CLmaxS, CLminN and CLmaxN for each acidity habitat (Table 2.1, this report) within each 1km grid square of each site polygon were determined using GIS (Section 3.1.6). The appropriate habitat-specific values were then selected for each site using relationships derived between the designated feature habitats and the critical load habitats.

This method has therefore provided estimates of habitat-specific critical loads for each SAC and SPA designated feature in the UK. In applying this approach the user needs to be aware of:

(i) The acidity critical loads are still based on the national scale maps, and hence the dominant soil type in each 1km grid square; the feature of interest may occur on a sub-dominant soil type which may be more or less sensitive to acidification.

(ii) The site relevant critical loads database is based on the UK critical loads data as of February 2003; further (relatively minor) updates were made to the UK critical loads data in February 2004 (Hall *et al*, 2004a).

(iii) The relationships between designated features and critical load habitats have been defined for the UK SACs and SPAs; A/SSSIs may be notified for features (habitats/species) not considered within SACs and SPAs and therefore further work may be required to assess all A/SSSIs.

(iv) Whilst this methodology can be (at least partially) automated using GIS, it can be a lengthy and time-consuming task.

4.1.3 National maps and data

APIS and the site-relevant critical loads work both make use of the national critical loads databases and maps. This section describes more generally how the national data could be used in different situations, using GIS technology to extract the relevant data on a site-by-site basis.

(i) Where the habitat type of interest has been mapped nationally.

Table 2.1 of this report identifies the Broad Habitats for which critical loads have been mapped nationally. Where the habitat(s) of interest are on this list, the habitat-specific critical loads (CLmaxS, CLminN, CLmaxN) can be extracted for a site.
For small sites (ie, falling within a single 1km grid square) the critical load values for the appropriate habitat(s) can be selected and used, bearing in mind the uncertainties in the national data described above and in Sections 2.2.4-2.2.8. For larger sites, covering a larger area and composed of one or more land parcels there could be many critical load values. For example:

Site X: total area 15 km^2 made up of two land parcels
Designated habitats: acid grassland and dwarf shrub heath
Land parcel 1: area 5 km^2, two habitats occurring in each 1km square = 5 values of CLmaxS, CLminN and CLmaxN for each habitat
Land parcel 2: area 10 km^2, two habitats, one occurring in 7 of the 1km squares and the other occurring in 5 of the 1km squares = a total of 12 sets of CLmaxS, CLminN and CLmaxN values.

For these types of sites there are different ways the data could be used:

- Select the minimum values of CLmaxS, CLminN, CLmaxN by habitat within each site (or site polygon) to give a set of critical load values to calculate a habitat-specific critical load exceedance.
- Select the minimum values of CLmaxS, CLminN, CLmaxN of all habitats within each site (or site polygon) to give a set of critical load values to calculate an exceedance aimed at protecting all the habitats considered within the site.
- Alternatively select the data as above, but use a different statistic (eg, mean) of the CLmaxS, CLminN and CLmaxN values for calculating the critical load exceedance.

Whichever approach is used the uncertainties in the national data still need to be taken into consideration.

For nutrient nitrogen the empirical values from Hall *et al*. (2003, for UK specific mapping values) or Achermann & Bobbink (2003) can be selected for the appropriate habitat(s) of interest.

(ii) Where the habitat is not mapped nationally.

The national critical load maps, as used for national-scale impact assessments for Defra only include critical loads for the 1km squares of the country for which the habitat has been mapped nationally (Table 2.1). For APIS woodland acidity critical loads were calculated for all 1km grid squares across the UK for which sufficient data were available irrespective of the presence of woodland. If data are available either from national databases or site-specific data, other habitat-specific critical loads (ie, CLmaxS, CLminN, CLmaxN) can be calculated for any location in the UK. As in all other instances, where the national data are used the limitations apply.

(iii) Where there is a lack of data or methods for the habitat of interest.

For acidity the following approaches can be considered:

- Use the acidity critical load for the dominant soil in the 1km square (ie, set the critical load to protect the soil upon which the habitat depends). These data can be extracted from APIS (selecting a non-woodland habitat type) or from the national data available from the UK NFC.
- If it is a woodland habitat, then APIS could be used to extract the acidity critical load for woodlands, bearing in mind that these critical loads data are appropriate for unmanaged woodlands only, are still based on the dominant soil in a 1km grid square and are not woodland type or species-specific.
- If the soil acidity critical load is available, what other data are required to calculate CLmaxS and CLminN and can they be obtained?

    - BCdep* available from national 5km databases
    - Cldep* available from national 5km databases
    - BCu information may be available in the literature, or the habitat may be considered to be similar to another habitat for which critical loads are currently calculated and the same value applied.
    - Nu as for BCu
    - Ni and Nde can be set if the soil type is known.

  Values of BCu and Nu for the habitats considered in Table 2.1 are published in the UK NFC 2003 and 2004 Status Reports (Hall *et al.*, 2003, 2004a). Values of Ni and Nde by soil type are published in Hall *et al.* (1998).

For nutrient nitrogen for non-woodland habitats and unmanaged woodlands the full list of empirical critical loads published by Achermann & Bobbink (2003) should be consulted. If the habitat is not included, then the user needs to consider if there is any information available to suggest the habitat of interest is likely to respond to nitrogen in a similar way to another habitat for which a critical load is available. However, it must also be recognised that for some habitats there is insufficient data on which to base a nitrogen critical load. In addition, because a critical load exists for a particular habitat this does not necessarily mean that the value will protect all possible species that may occur within the habitat, since the values are only based on the habitat or species data that exist and were agreed upon at the Berne workshop (Achermann & Bobbink, 2003).

4.1.4 The value of collecting site-specific data

The collection of site-specific data for the calculation of critical loads for each site feature would be a time-consuming and expensive task, and may not necessarily result in a critical load significantly different to those derived from the national data (Sections 4.1.1-4.1.3). However, the values from the national data/maps may be inappropriate at the site-specific level for the reasons discussed above and in Sections 2.2.4-2.2.8. Users of the national critical loads data are provided with a document outlining the limitations in their use and interpretation. However, many users query the appropriateness of critical load values extracted from the national data that they are using at the site-level. In the majority of cases the reason the critical load could be inappropriate for their habitat of concern, is that the habitat may not be occurring on the dominant soil on which the critical load is based. Therefore the most important site-specific information that would improve the estimates of acidity critical loads for site-specific assessments are:

- The locations of the features (habitat/species) of interest within the site.
- The soil types at those locations described in the standard nomenclature of the National Soils Resources Institute (NSRI), preferably to soil association or soil series level.
- The relationship between the soil type (again NSRI nomenclature) and the habitat/species of interest (ie, which soil types are the habitat/species associated with; this information would be useful even if the location within the site was not known).

Access to this information would have the following benefits:

- Soil acidity critical loads could be defined using the look-up table by soil associations (Loveland, 1991) for England and Wales.
- For woodland habitats knowing the soil type(s) would improve the estimates of base cation and calcium weathering rates required for the simple mass balance equation. The equation can be very sensitive to these parameters at some sites.
- Ni and Nde could be defined based on the site soil type(s), enabling an improved estimate of CLminN.
- The Endorsement Theory approach to setting acidity critical loads by vegetation community could be applied.

The Endorsement Theory approach, developed by Wadsworth & Hall (2005) and Wadsworth & Hall, (in press) relates information on the soil and geology characteristics for each terrestrial NVC class to the soil and geology information used in setting the empirical acidity critical loads for soils (Loveland, 1991; Hornung *et al.*, 1995b). By determining the "weight of evidence" (in terms of soils and geology information) for a particular critical load class, an endorsement for one or more classes (ie, ranges) of critical load values can be defined. For further information on the methodology refer to Wadsworth & Hall, in press.

The differences in acidity critical load values depending on whether national input data or site-specific input data are used in the calculations were also studied as part of the Environment Agency R&D project "Uncertainty in critical load assessment models" (Skeffington *et al.*, 2006). There are very few sites in the UK for which sufficient site-specific data exist to enable this type of comparison. Table 4.1 shows the acidity critical load and exceedance results for two coniferous woodland sites studied.

Table 4.1 Comparing the results of calculating acidity critical load and exceedances using site-specific and national data for two coniferous woodland sites.

Parameter	Data used	Critical load & exceedance values (keq ha^{-1} $year^{-1}$) for:	
		Liphook	Aber
Critical load	National	0.70	1.53
	Site-specific	0.52	1.76
Exceedance	National	2.10	0.94
	Site-specific	-0.02 (not exceeded)	1.34

Table 4.1 shows a difference of about 0.2 keq ha^{-1} $year^{-1}$ between the critical load values based on site-specific and national data. However, this cannot be taken as a general difference likely to be observed at other sites, differences may be smaller or greater. The uncertainty studies showed that the critical load calculations could be sensitive to different input parameters at different sites. The exceedance results for Liphook show no exceedance when site-specific measured data are used, compared to a high exceedance using national data; in this case the measured deposition at the site was much lower than the national estimate. For Aber both national and site-specific data result in exceedance of the critical loads; here the site-measured deposition is greater than the national data suggest. However, as for critical loads no general rule can be taken from these results.

4.2 Sites with species interest features

The taxonomic groups reported for species interest features are listed in Table 4.2 together with the frequency of their occurrence in A/SSSIs in the UK.

Table 4.2 Reporting categories of taxonomic groups of species

Reporting category	Number of features	Percentage of features
Amphibians	19	3.5
Birds	2	0.4
Birds: aggregations of non-breeding birds	89	16.4
Birds: aggregations of breeding birds	133	24.5
Birds: assemblages of breeding birds	11	2.0
Birds: unspecified	33	6.1
Butterflies	10	1.8
Dragonflies	6	1.1
Fish	16	2.9
Invertebrates	8	1.5
Invertebrates: other	32	5.9
Mammals	63	11.6
Plants: non-vascular	49	9.0
Plants: vascular	72	13.3

The number of features in each category varies; however, it is unlikely that all features within a taxonomic group will be (a) sensitive to acidification or eutrophication; or (b) found in the same habitat type(s) or location(s). For example, Fuller (1982) has examined the bird communities that depend on different habitat types, such as lowland heath, uplands, woodland etc and shows the variability in numbers of bird species and habitat. A species may be found in a number of different habitats and may occur in different habitats in different parts of the country.

For all taxonomic groups there is unlikely to be a uniform habitat or a uniform response to a particular pollutant. Therefore, we consider this level of reporting too coarse for assigning critical loads of acidity and nutrient nitrogen. This means that information on the species present within the taxonomic group at individual sites would be required to perform an assessment. The species for which a site has been notified must have been recorded at some stage and then assigned to a taxonomic group. For example, Holme Fen in Cambridgeshire is both a SSSI and an NNR and has been notified for its birch woodland and named plant species that are relicts of the raised mire habitat (Source: English Nature web site). For the purposes of reporting site condition, we assume these would be recorded as vascular and non-vascular plants, if reporting is only done to taxonomic group.

For critical load assessments ideally the species list by site is required, and if possible a list of the broad habitat(s) associated with each species at each site. As it may be simpler to assign broad habitats to plant species than non-plant species, the two are discussed separately below.

4.2.1 Sites with plant species interest features

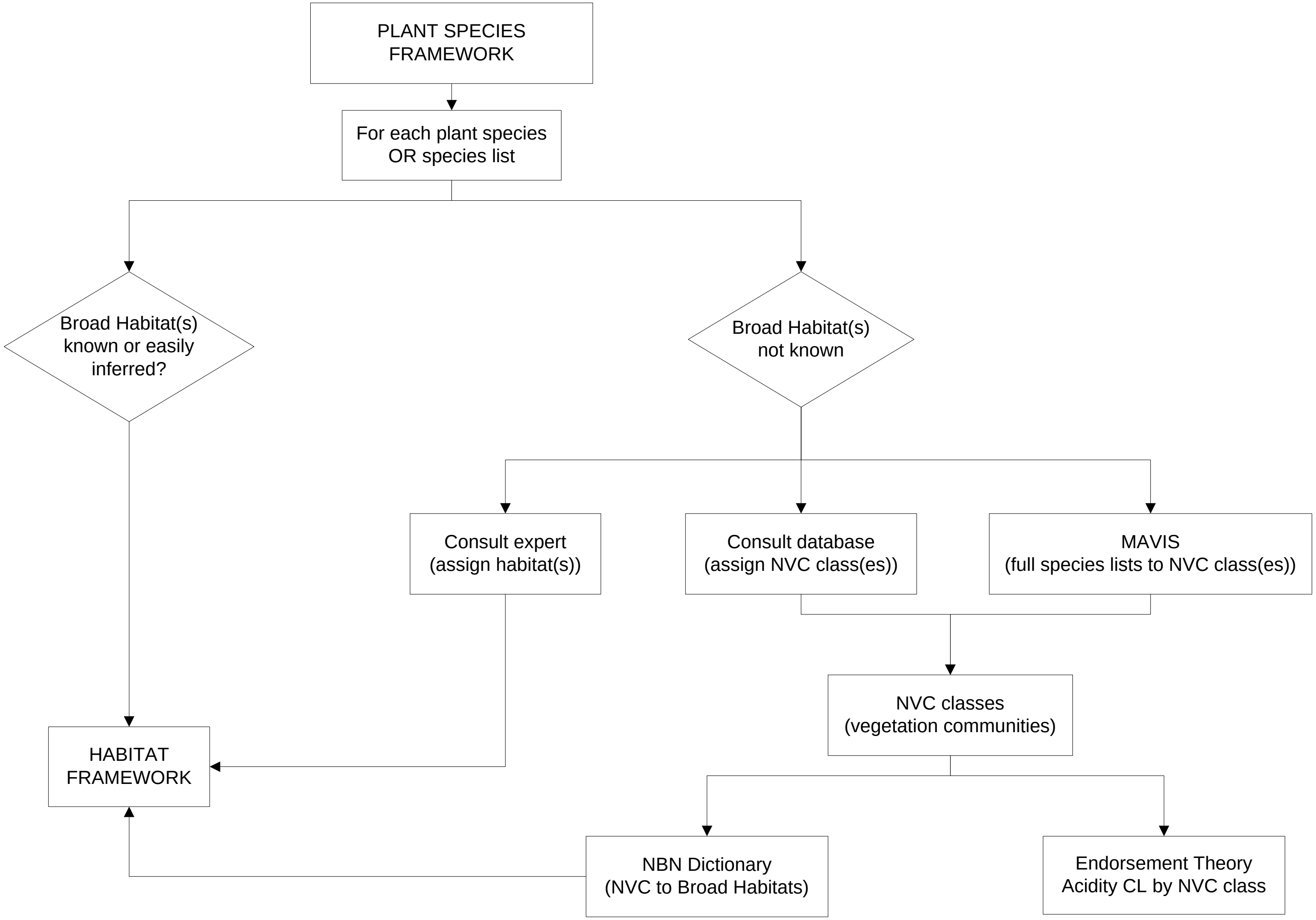
Critical load methods have generally been developed and applied at the habitat or vegetation community level. There is often limited information to enable models to be parameterised for individual plant species. One of the criteria that is used in the Simple Mass Balance equation for calculating acidity critical loads is the critical molar ratio of calcium (or base cations) to aluminium in soil solution. A commonly used value for this ratio is one, which is applicable for coniferous woodland ecosystems and also considered appropriate for broadleaved woodland in the UK. Critical ratios of base cations to aluminium have been published for a range of tree, grass and herb species (Sverdrup & Warfvinge, 1993), however, these data tend to have been derived from studies carried out across Europe and East Asia and may not necessarily be applicable in the UK; ideally values based on UK studies are needed.

Therefore, we propose the following approach that is based on linking the species to habitats or communities. Assuming information on the notified species could be made available, it may be possible to perform a critical load assessment for these interest features (Figure 4.2). A species may be found in one or more habitat types; if the associated habitat(s) is known or can easily be inferred, then the habitat framework (Figure 4.1) can be followed. If the associated habitat(s) are not known, then the following options can be considered:

(i) Consult a relevant habitat expert (or botanist) or database to determine the appropriate habitat(s) upon which the species depends.
(ii) CEH have developed a database under an EA R&D project (Wadsworth & Hall, 2005) which includes look-up tables of species by terrestrial NVC class.
(iii) CEH software MAVIS can assign NVC class(es) from full species lists, ie, a full list of species and their abundance within sample quadrats.

Options (ii) and (iii) provide NVC vegetation community classes rather than broad habitats. The NVC class(es) can be translated into broad habitats using the National Biodiversity Network (NBN) Habitats Dictionary (http://www.nbn.org.uk/habitats). It should be noted that there is unlikely to be a simple one-to-one relationship between NVC class and broad habitat, an NVC class will be defined as being "contained" in, "overlaps with" or "may occur in" a broad habitat. However, identifying the relevant broad habitats will enable the habitat framework in Figure 4.1 to be applied. Alternatively, for acidity, the Endorsement Theory approach (Section 4.1.4 and Wadsworth & Hall, in press to setting critical loads by NVC class could be applied.

Figure 4.2 Plant species framework



4.2.2 Sites with non-plant species interest features

Critical load methods exist for terrestrial (grassland, heathland, montane, bog, woodland) and freshwater (lake, stream) habitats. Critical loads have not been developed for the non-plant taxonomic groups listed in Table 4.2, with the exception of fish, where the criterion used in the calculation of acidity critical loads is based on protecting a percentage of the fish population, specifically brown trout (Hall *et al.*, 2004a). If the habitats or vegetation communities upon which a non-plant species depends can be determined, and critical loads data exist, it may be possible to perform a critical load assessment using the habitat framework in Figure 4.1. Such a process was developed for site relevant critical loads as described in Section 3. SPA bird features were assessed by judging whether the impact on the bird's habitat (nesting, feeding or roosting habitats) would have an intrinsic impact on the integrity of the bird itself. Therefore, for both plant and non-plant species any such assessment is aimed at protecting the habitat (or community), rather than individual species.

4.3 Assigning critical levels to notified features

Critical levels have been set for fewer receptors than critical loads (Table 2.4). For both ammonia and nitrogen oxides single values (hourly, daily, month, or annual mean concentrations) have been set to protect all vegetation types. For sulphur dioxide critical levels have been set for three ecosystems (forest ecosystems, semi-natural vegetation, agricultural crops) and cyanobacterial lichens. Threshold values for ozone have been defined for forests and semi-natural vegetation.

Hence critical level values can be assigned to designated features by determining which of these general ecosystem types is most relevant. APIS can also be used to obtain the appropriate critical level(s) where the habitats/species listed in the system have been assigned to one of the ecosystem types. However, since critical levels have been set at this broad scale, it does not necessarily mean that all species or habitats within these categories will be protected; the effects-based data and research on which these values are based will be limited to the particular species or habitats studied.

5. Producing a Risk Assessment for Critical Loads and Levels Exceedance

The national critical loads and critical levels data have been used as a screening tool in attempts to identify which designated sites are at risk from the potential harmful effects of some airborne pollutants (e.g, Section 3). The emphasis has largely been on acidification and eutrophication with a lesser amount on sulphur dioxide and nitrogen oxide concentrations (Section 2.3.14). Methods are available for the calculation of critical loads for heavy metals, but these are still undergoing further development and testing (UBA, 2004). Methods have also been proposed for persistent organic pollutants (POPs), particularly by the Dutch, but the UK is not currently planning to adopt this approach.

Since the national data were never intended for site-specific assessments, and are based on national-scale databases, they *can* give misleading results for individual sites. Despite this fact, adopting any of the alternative approaches is generally hampered by a number of pragmatic and scientific issues. The alternative methods require either:

- a substantial increase in resources (particularly staff time),
- more data (some of which is difficult or expensive to measure), or,
- utilising less well-determined science.

In this section we consider each of seven alternative approaches for carrying out risk assessments at the site level and highlight the advantages and disadvantages of each method. Assessment of these alternatives leads us to make the specific recommendation that, at the moment, a combined approach is the pragmatic choice, however, as science and technology continually develop this decision should be revisited, perhaps in five years time.

5.1 Possible Approaches

Table 5.1 presents seven possible options for performing a site specific assessment. For each method Table 5.1 summarises their associated risks and benefits; these are discussed in more detail below.

Of the seven possible methods we suggest that four options;

- direct visual inspection,
- biomonitoring,
- sampling the environment,
- measuring critical load parameters,
-

are unlikely to be practical in the near future (at least 5 years) while the remaining three options:

- Specific desk studies,
- Internet tool APIS,

- National maps and data,

are more practical but have resource implications, and the latter two may still be constrained by limitations in the use and applicability of the national-scale data.

5.1.1 Direct Visual Inspection

In an ideal world the surveyors carrying out the monitoring under the common standards scheme would be able to observe the effect of air pollutants directly whilst making their other observations. Direct visual inspection is therefore a theoretically beguiling option; unfortunately it is in fact impractical under our current knowledge of the environment. The crucial problem is that by the time that the damage is unambiguously visible a great deal of invisible damage may have already occurred (eg, changes in soil chemistry or in biology or in ecosystem function). Moderate levels of damage, such as a poor growth rate, could be due to many factors not just air pollution so that it could be difficult to justify an attribution to one cause or another. For example, there are some published photographs of damage attributed to ozone (Figure 5b http://www.ceh.ac.uk/sci_programmes/GroundLevelOzone.htm) that look very similar to potassium deficiency (Figure 5.1a copied from http://www.hbci.com/~wenonah/min-def/clover.htm Wallace 1943).

Figure 5.1 The problem of attributing damage to a particular cause; (a) Potassium deficiency (left) and (b) ozone (right) damage on clover.

The situation in minimally managed situations (such as A/SSSI) is even more complex as plants may be subject to multiple stressors in terms of both shortages and over abundance of nutrients (for example an excess of nitrogen and a deficit of phosphate), together with potential effects from diseases, viruses, temperature and water stress or water logging, these multiple stressor will all modify the appearance of the plants.

Direct visual inspection might be useful as a qualitative corroboration of estimated impacts on a site. For example, we would not expect to see obvious damage on a non-exceeded site but, because the critical load is calculated for long-term equilibrium conditions damage may not be visible on exceeded sites where steady-state has not been reached.

While the direct visual observation of damage to vegetation is not possible, it is possible to identify potential sources of pollutants that could be affecting the site. In a lowland rural setting the most likely important local sources are intensive livestock production where emissions of ammonia could have serious local impact.

We recommend that surveyors carrying out the common standards monitoring observe and record the general condition of the vegetation but that this information is used in comparisons with adjacent (or nearby) sites as a possible "flag" that further investigation is required.

We recommend that surveyors carrying out the CSM observe the land use and activities around the designated site for signs of unusual and potentially important sources of pollution.

5.1.2 Biomonitoring

A number of biomonitoring techniques are currently under development. JNCC has funded a detailed study into the use of bioindicators and biomonitoring to specifically assess the impact of nitrogen depostion on designated sites (Sutton *et al.*, 2004). A number of techniques were investigated that can be broadly grouped into;

- chemical and biochemical methods; includes nitrogen accumulation methods of direct N concentrations and biochemical response methods which includes the analysis of enzymes,
- species compostion; can include higher plants using changes in the "Ellenberg" scores as vegetation communities change or lichens which are known to be particularly sensitive, but which require specialists to identify,
- transplant methods; includes methods that use "standardized" grass plants or which transplant native species.

Generally the methods require repeated actions over time scales of a few years (although shorter time scales are possible). Sutton *et al.* (2004) conclude that each of the different approaches have strengths and weaknesses but when used in conjunction and when sufficient resources are available they can provide a robust estimate of the effects.

Many other biomonitoring techniques have been designed to measure the impact of multiple stressors from industrial pollutants where the synergistic effects are unknown (and not amenable to laboratory investigation due to the large number of possible combinations) or where the cost of chemical analysis is prohibitive. A biomonitoring technique that might be capable of being adapted to the assessment of airborne pollutants (eg, acidity, nitrogen) on soils is "bait lamina" strips. Bait lamina strips are thin strips of plastic which have 16 holes ~ 2mm in diameter drilled in them; these holes are filled with "bait" (bran and other material attractive to soil dwelling organisms). The strips are inserted vertically in the ground and left for a few days to a couple of weeks (dependent on weather and soil conditions). The amount of bait consumed is an indication of the amount of biological activity in the soil; the assumption is made that the more biologically active the soil the healthier it is. Bait lamina strips are currently used to compare polluted and non-polluted sites or to compare between different management options (Wadsworth 2005). It might be possible to "calibrate" the method so that it could be used to assess the effect of excessive acidity or nutrient input, BUT, these studies have not been carried out. Although bait lamina strips would require a minimum of two visits to each site they would be relatively cheap to do. The drawback is that at the moment there is no information on the likely activity in soils under natural or semi-natural vegetation communities. It is hoped that in the long-term biomonitoring methods will prove to be efficient and effective; however, different methods may be required for different pollutants and further development is still needed in this area.

We recommend that the conservation agencies continue to support the development of appropriate biomonitoring techniques for acidity as well as nitrogen. We also suggest that they start to develop a database of what the expected values are from polluted and pristine sites.

5.1.3 Sampling

Like direct visual inspection, sampling an environmental attribute provides evidence from the site; however, it is not always clear what should be measured and how it could be measured. The simplest case is probably for some of the habitats dependent on peat soils; the acidity critical load for peat soils is based on maintaining the soil water pH (ie, the critical chemical criteria) above a particular threshold (Section 2.2.5). In such a case it might be possible to measure the soil water pH at the site and deduce whether that critical chemical threshold is being exceeded or not. In the hypothetical case suggested here there is an additional unfortunate complication in that water pH can vary by time of day as well as between days or seasons so it is not clear how many samples would be required at what time of the day, season or year. For other habitats it is more difficult to decide what to sample.

We recommend that direct sampling of biophysical parameters is not attempted.

5.1.4 Measuring critical load parameters

There are relatively few parameters in the equations used to calculate critical loads, so it should in theory be possible to measure one or more of the parameters to refine or validate any estimate. The soil weathering rate is one of the most important parameters in determining acidity critical loads, that is, it is important in the sense that the calculations are sensitive to changes in this parameter. There are a number of methods available for determining soil weathering rates (UBA, 2004), ranging from the empirical assignments by Nilsson & Grennfelt (1988) and utilised in the UK in the setting of acidity critical loads for non-peat soils, to the use of complex models (eg, PROFILE, Warfvinge & Sverdrup, 1992 and 1995) or catchment budget exercises. However, the literature suggests that direct measurements are not easy to perform.

The recurring problem with this approach and the preceding approaches is that the critical loads are based on steady-state conditions and so even if this were possible; it may still be open to interpretation and hence be of little use. The difficulty and expense of directly measuring critical load parameters is revealed by the observation of how few sites there are across the UK which have a comprehensive monitoring scheme (even after two decades of research).

At some sites there may be local environmental factors that mean that deposition or other factors are atypical. For example, Table 4.1 (Section 4.1.4) shows the difference site-measured versus national deposition estimates can have on critical load exceedances. In addition, some locations are particularly prone to fog and hence deposition can be enhanced over and above what might be expected from national estimates. Although these local factors could be important it is not certain what the crucial factors are or how these could be assessed during a site visit.

One piece of field data that could be collected that would be very valuable, is the specific soil type under the designated feature. To be useful the soil would have to be identified to "association" or "series" level using the nomenclature of the NSRI (see Section 4.1.4). It is uncertain how easy it would be to train a non-soil scientist to do this reliably and consistently. Some simple soil measurements would also assist in the development of dynamic models (see Section 5.3).

We recommend that direct measurement of critical load parameters is only attempted to identify the relevant soil information.

5.1.5 Desk Studies

Desk studies encompass a wide range of possible activities; the most significant of which are shown in figures 4.1 and 4.2 (Section 4). A desk study entails gathering all the readily available information for a site (as opposed to collecting new field data) on the site of interest. For example, there is almost certainly local and more accurate knowledge of the soils and vegetation on a site than is contained in any national database although it might not be directly usable; for example soils may be described in different methods to the NSRI series/association description. Local information on pollution loads might also be more relevant (accurate, precise) than can be obtained from national data sources. How useful this is will depend on the amount and availability of data, particularly site-specific data, and on the further development of methods to enable the information from different sources to be combined. If soil series/association data are available it will increase the utility of performing a desk study considerably and can improve the estimate of several critical load input parameters (Section 4.1.4). However, even a simple desk study could require several hours of work, while this is acceptable for a single site when multiplied by the many thousands of designated sites the resource implications are considerable.

We recommend that desk studies are only carried out where the national data produce indeterminate or ambiguous results (see Section 5.1.7 below).

5.1.6 APIS

APIS and the national estimates will provide similar results. It has been previously noted (Section 4.1.1) that the underlying data in APIS are based on the same national-scale critical loads data sets generated for UK assessment work for Defra (Hall *et al*., 2004a), but the web interface could initially give the impression that it was more thematically detailed. Despite the ability to specify any of a large range of habitats (34 in total) the acidity critical load value produced is (with the exception of woodland habitats) the soil acidity critical load based on the dominant soil in each one kilometre grid square. As with all internet services it is also important to confirm that it contains the most recent version of the data and approved methodologies. APIS uses the empirical soil critical loads for acidity for **all** non-woodland habitats and the Simple Mass Balance equation for woodland habitats. For nutrient nitrogen the ranges of empirical critical loads as provided in Table 2.2; for habitats not included in either this table or Achermann & Bobbink (2003), values are provided for the habitat considered to be most similar to the habitat of interest.

An APIS user can also select any of a large range of species; however, in this case only the nutrient nitrogen critical loads values are provided based on the allocation (within APIS) of each species to one of the nutrient nitrogen habitats in Table 2.2 or Achermann & Bobbink (2003). The critical levels of NO_x and NH_3 are not differentiated for different habitat types or species and so APIS assigns the single critical level values shown in Table 2.6 (Section 2.3.14); for SO_2 and ozone the species are assigned (where possible) to the vegetation categories also in Table 2.4.

In addition, for individual point locations for habitats, APIS provides critical load and critical level exceedance values. The exceedances are based on the critical loads values and critical levels values described above and the national estimates of deposition (acid, nitrogen) and concentrations (SO_2, NO_x, NH_3) from 5km resolution data sets for the UK and 1km resolution AOT40 data for ozone. For acidity the critical load exceedance values in APIS do not include any ameliorating effect of base cation deposition (Section 4.1) and hence in some circumstances may over-estimate exceedance. Exceedances are not calculated in APIS at the species level.

The advantages of APIS are;

- that it can be accessed across the Internet and
- the assessor can specify any habitat for the site under consideration (although this doesn't affect the result for acidity critical loads, except woodland vs non-woodland).

The disadvantages of APIS are;

- that with so many sites doing them all one at a time will take a long time and,
- the user needs to ensure that the most up-to-date version of the data and methods have been implemented,
- it is difficult to ensure "error free" selection of the appropriate habitat (that is, there is no restriction on selecting "salt marsh" as a habitat for a site in the Cairngorms).

APIS includes warnings on the limitations of the national critical loads data, which the user has to accept prior to obtaining the values for a particular location; however, given the type of queries referred back to CEH, it would suggest users are not reading this information. We therefore recommend that APIS is used with care.

5.1.7 National Screening

By national screening we mean the process by which national estimates (maps) of habitat specific critical loads are used together with national estimates of deposition loads to identify designated sites that are potentially exceeded. These methods have previously been applied to SACs and SPAs (eg, Section 3). The advantage of a screening exercise using the national data sets is that the process can be largely automated and hence the "true" cost of assessment per designated site is low, however as the cost would be to an external agent it would be "visible" whereas encouraging the use of APIS the costs are "hidden". However, unless some "expert rules" are included in an automated screening exercise potential anomalous values may not be readily identified. In addition to extracting "site-relevant" critical loads, estimates of critical load exceedance are also required. In a national screening it should be sufficient to categorise sites into three groups those; definitely exceeded, definitely not exceeded, uncertain. These three groups could be delineated using the standard statistical convention of the 95^{th} percentile.

Therefore a national screening exercise consists of following two tasks:

a) extracting (and possibly calculating) the most appropriate critical load(s) for each site.

b) calculating the *probability* of exceedance.

A similar approach could be adopted for critical levels using the national concentration maps overlaid with the designated areas and calculating the probability of critical levels exceedance for each site; however, at present these calculations have not been performed and some method development may be required.

Note that because this approach relies on national data sets the results for some sites may be inconsistent with the specific conditions of particular sites.

We recommend screening designated sites using national data every few years coincident with the performance of the CSM scheme.

5.1.8 Hybrid and hierarchical approaches

For convenience each of the seven options has been presented above as independent and competing approaches. Obviously, as each approach has strengths and weaknesses a hybrid, linked or hierarchical approach that combined in sequence more than one approach will lead to the most robust estimate. Section 5.4 (Recommendations) describes a multi-stage approach where a national screening is followed by specific desk studies (on sites where the results are ambiguous) followed by a qualitative assessment for those sites where the desk study is still unable to resolve the issues.

Table 5.1 The options, costs and benefits of each assessment approach

What are the options, and what are the costs, benefits and risks?				
Option	State of Science	Risk(s)	Costs (time/money)	Benefits
1. Visual observation / inspection of damage to the environment	CL is the "no known effect" therefore difficult to know what to look for. CL and exceedance based on steady-state conditions so difficult to validate in field.	The level of visual damage that is unambiguously caused by air pollution may be well above the level where it is affecting the soil/vegetation chemistry or biology. Requires interpretation of observation, therefore, could be open to challenge.	If it was combined with the Common Standards Monitoring visits only marginal difference.	If (and only if) science issues can be resolved (ie what to look for) then relatively, simple, quick, cheap and defensible.
2. Sampling biota	Biomarkers eg lichens as bio-indictors of N effects; bait lamina strips well developed to measure biological activity, however, little research on pristine or near pristine sites.	Degree to which biomarkers capture essence of environment. There can be multiple stressors contributing to an observed value, so difficult to attribute damage to cause. Can be sensitive to local conditions, weather. Season, soils etc.	Experts needed to correctly identify lichen species. Bait lamina method requires several visits to each site in each monitoring campaign within a restricted time frame. Individual sample (eg bait lamina strip) is cheap, but on a complex site more needed and the costs would escalate.	Objective, quantitative.
3. Sampling environment (eg, soil, water)	Direct measure of damage, eg water quality of runoff.	Difficult to know what to measure	Chemical analysis may be difficult and expensive	Objective, quantitative.
4. Collecting CL model parameters for site-specific calculations	Models are well known and have been developed over many years	Some parameters, eg non-marine calcium deposition difficult to measure, tendency to bias data by what is easy to collect	Could be very expensive, only a handful of sites (in UK) have a significant proportion of parameters measured. Even monitoring a single site would be a long-term and expensive commitment.	Unambiguous result.
5. Desk study of site	Incorporating local qualitative knowledge with quantitative national data possible but still an active area of research.	Inappropriate data fusion?	Time to complete desk study could be comparable to a site visit. Knowledge of site(s) and understanding of CL needed.	Specific results can include whatever local knowledge is available.
6. APIS	Based on consensus opinion (published material, agreed methods/data)	By virtue of being protected conservation sites are unusual and therefore potentially atypical. Inevitable inconsistency and error in national data sets is often obvious when a specific site is identified.	Staff time (training?) assessment of a single site relatively quick but because there are 6,000 sites total cost would be considerable.	Consistent approach across sites and regions.
7. National assessment / sieve	Based on consensus opinion (nationally/internationally agreed methods/data)	Similar to above. May be more difficult to identify anomalies when automated.	As process can be semi-automated costs slightly lower than above but still significant.	Consistency across sites and regions.

5.2 Costs and resource implications

The costs and resource implications of each option are approximated and give our best estimate of the time we think would be reasonably required to perform those tasks. For options that require a field visit there is an additional significant component of travel to those sites, we presume that that information is accessible to JNCC. Resource implications (as far as they can be estimated) are summarised in Table 5.2.

Table 5.2 Resource implications of different approaches to estimating Critical Loads

Approach	Field work per site	Proportion of sites visited in the field	Desk work per site	Total time (days)[1]	Other costs
1. direct visual inspection	15-20 mins	100%	5-10 mins (documenting observations)	270 to 405	none
2. biomonitoring	2 visits; 5 replicates per site; 1-2 hours total	100%	10-15 mins assessment and documentation	946 to 1800	£50+ per site
3. Sampling	1-2 visits; 1-5 measurements; 1-6 hours total	100%	10-15 mins assessment and documentation	946 to 5068	£? Might include chemical analysis.
4. measuring CL parameters	1-2 visits; 1-5 measurements; 1-6 hours total	100%	10-15 mins assessment and documentation	946 to 5068	£intensive (collection and analysis)
5. Desk study	0	0	10 mins to 4 hours	133 to 3200	0
6. APIS	0	0	3 to 5 mins	40 to 68	0
7. national screening	0	0	-------	10 to 15	0

Notes:
1. assumes 6,000 designated sites.

5.3 Developments on the Horizon

Digital data for designated sites are gradually changing from a cartographic to a GIS product; this change makes an assessment of the parcels within complex sites possible, while developments in computer power make the process less time-consuming than previously.

Biomonitoring is a rapidly developing field; it is possible that there will be appropriate techniques developed for the next cycle of the Common Standards Monitoring assessments. Biomonitoring techniques are well developed for complex pollutant mixtures (such as PAHs etc). Methods are being developed and tested for acidity, nitrogen and ozone, for example, recent work by CEH and the Natural History Museum has focused on developing and field trialling methods that use lichens as indicators of nitrogen impacts on A/SSSIs (Sutton *et al.*, 2004).

The capacity to forecast the adverse impacts of nitrogen deposition on biodiversity is beginning to be developed, through combining biogeochemical models of nitrogen dynamics in soils and vegetation with static models that relate species occurrence to environmental factors. The static models identify niches defined by environmental factors such as soil pH, carbon:nitrogen ratio or available nitrogen, that provide favourable conditions for the species (or communities) to be protected. Effects of grazing, current flora and dispersal are also considered. The biogeochemical models determine the nitrogen load required to maintain or return to (in the case of critical loads and target loads, respectively) these favourable environmental conditions. For the UK, the dynamic model MAGIC (Cosby *et al.* 2001) and the species occurrence model MOVE (Smart *et al*. 2005) have been combined to form MAGIC-GBMOVE (Rowe *et al.* 2005), which has been tested against data from the few UK sites which have long-term soil and floristic datasets. These models are suitable for assessing nitrogen exposure at designated sites and forecasting change in relation to pollution changes and management interventions, but require some simple soil measurements in addition to species data.

5.4 Recommendations

Our overall recommendation is that a multi-stage process be carried out. Stage 1 consists of a comprehensive national screening (Section 5.1.7) of all designated sites, to be performed every few years in synchrony with the CSM scheme. The end result of the national screening will be to categorise sites into those where the critical loads or critical levels are definitely not exceeded, those that are definitely exceeded and those that are uncertain using the standard statistical convention of the 95th percentile.

Under Stage 2, those sites for which exceedance is uncertain should be investigated in more detail using specific desk studies. For critical loads the utility of the desk study will be considerably enhanced if site (or feature) specific relevant soil association or series data are available. It is predicted (following assessment of the uncertainties in critical loads, Skeffington *et al.*, 2005) that the major sources of uncertainty in critical load values derived from the national data sets will be:

- the inappropriateness of the dominant soil (on which the acidity critical load is based) for the site feature(s)
- inappropriate land cover, for example, bog areas classified as arable or fen on national maps, thereby resulting in inappropriate acidity critical load values
- errors in national-scale soil or land cover databases
- an inappropriate or erroneous allocation of a designated feature (especially non-plant species) to a habitat type.

The result of these two stages (national screening and selected desk studies) should then be qualitatively validated with any reports on the general condition of the vegetation and on any reports of local significant pollutant sources.

We also recommend that a watching brief is kept on the development of, and data requirements for nitrogen impact models (eg, MAGIC-GBMOVE), as well as developments in biomonitoring in the next 5 to 10 years time to see if they have developed sufficiently to be useful in routine monitoring to determine the threat of airborne pollutants on designated sites

6. Conclusions

- In assessing the threat to designated sites from airborne pollution the key pollutants that should be considered are sulphur (acidity), nitrogen (acidity and eutrophication) and ozone. There are many other pollutants that may be threatening or damaging features on designated sites; but the science for assessing risk is not well enough developed to be implemented in a standard methodology. It is possible that in the next few years the methods for calculating heavy metal (lead, cadmium, copper, zinc) critical loads and their associated data sets will be suitable for site-based assessments, if heavy metal deposition is considered to be posing a threat to designated sites. Proposals for the assessment of POPs (persistent organic pollutants) using critical loads have also been made but there are currently no plans for this approach to be adopted in the UK.
- The desire for site-relevant values and that fact that “site-relevant” critical loads have been assigned to SACs and SPAs can obscure the fact that there is detailed data for a very limited set of habitats and no data at all for individual species. When a value is given to a specific species or priority habitat the actual process is to allocate that species to one of only eight terrestrial broad habitats. This allocation is almost always done on the basis of expert opinion. In addition to being based on expert opinion the allocation of a species, habitat or other designated feature to a single habitat, is typically performed as a one-to-one relation whereas in reality a species may be dependent on a number of different habitats.
- Critical levels for sulphur dioxide are only available for three general ecosystem types (forests, semi-natural vegetation, agricultural crops) and critical levels for ozone are for forests and semi-natural vegetation only; a flux-based approach for ozone is available for agricultural crops. For nitrogen oxides and ammonia it has not as yet been possible to set critical levels for different ecosystem types, so single values are applied to all vegetation types. Whilst it is possible to assign critical levels to appropriate features of designated sites, the user needs to be aware of the paucity of data and critical levels values currently available.
- Designated sites are by definition special, and hence often contain unusual features and combinations of plants, animals and the environment. National critical loads data are often only available for the dominant variable within a region. These factors contribute to the fact that any estimate made using national data may not be completely appropriate for a particular site.
- Critical loads can be assigned to A/SSSIs where the designated features are broad habitats, using a similar methodology to the site-relevant critical loads study. For species, the taxonomic group is too coarse a classification to assign critical loads, since it is highly unlikely all taxa will have the same response to a pollutant, hence information on individual species is required. We suggest this route (using the presence of particular species) is best pursued for plant species. For non-plant species more care is required in ensuring the appropriate species to habitat(s) relationships are defined. In all cases it is important to remember that the critical load has usually been developed to protect a habitat or ecosystem, and not individual species.
- We believe that at present site specific assessments based on visual inspection, on measuring particular characteristics of a site, on measuring critical load/levels parameters or on biomonitoring are all impractical for both scientific, logistical and resource reasons (Section 5 and Table 5.2) and should not be carried out as part of a national survey. There is however, one piece of information that would make a significant improvement to

estimates of risk; that is the association of the designated feature with the soil type where the soil type is described to the series or association level using the NSRI nomenclature.

- We recommend JNCC keep a watching brief on the development of, and data requirements for, nitrogen impact models such as MAGIC-GBMOVE.
- We recommend JNCC keep a watching brief on biomonitoring techniques as these may become viable and valuable for assessing acidity or eutrophication within the next 5 to 10 years (if the research is funded).
- We recommend a national screening assessment, taking into account the caveats associated with the national data, to coincide with the 6-yearly status report of the condition of features on A/SSSIs across the UK and follows the process outlined in Section 5.4. This should be followed by more detailed “desk studies” for those sites where the results are ambiguous (that is, where it is not certain whether the site is under threat or not).

7. References

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