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Spatialities and temporalities of metrics calculated by Integrated Assessment Models: Exceedance of ecosystem-specific Critical Loads

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Abstract: Integrated Assessment Models are designed to calculate a variety of metrics which describe the impacts upon ecosystems and human health resulting from specified spatial patterns of emissions of air pollutants. These metrics can provide policy makers with useful information which can assist in sustainable policy development and negotiation of international protocols. However, they can also present a very different picture of impacts depending upon the spatial resolution of the models used to describe emissions, atmospheric dispersion, deposition and air quality, and the metrics themselves may change as scientific understanding evolves over time. Using exceedance of ecosystem Critical Loads as an example of such metrics, we show how the area of critical load exceedances increases both as the spatial resolution of modelled deposition increases and as scientific advances in the representation of ecosystem specific deposition patterns are included in the models. Furthermore, advances in scientific understanding of Critical Loads, atmospheric dispersion and deposition rates can lead to apparently paradoxical quantifications of impacts when the effectiveness of policies are subsequently reviewed. Policy makers demand use of the most up-to-date science and may thus, when comparing against earlier assessments, discover an apparent increase in the ecosystem area exceeded due to scientific improvements in the models rather than reduced emissions; such results reflect the different and potentially conflicting temporalities of scientific developments and the policy making processes, as opposed to impacts in the real world. Thus, we also highlight the importance of coordinating these temporalities and using equivalent tools or models for both assessment of potential impacts of policies and any subsequent review of the effectiveness of those policies.

Keywords: *Spatial and temporal resolution; critical loads; integrated assessment; Gothenburg Protocol*

1. INTRODUCTION

Integrated Assessment Models are designed to calculate metrics which describe the impacts upon ecosystems and human health resulting from specified spatial patterns of emissions of air pollutants. Such metrics can provide policy makers with useful information which can assist in sustainable policy development and negotiation of international protocols. The successes of integrated assessment modelling in relation to development of the Gothenburg Protocol [UNECE, 1999], the EU National Emissions Ceilings [EC, 2001] and urban air quality has been documented by Hordijk & Amann (2007), continuing with the ongoing revision of the protocol [UNECE, 2010].

However, these metrics can also present a very different picture of impacts depending upon the spatial resolution of the models used to describe emissions, atmospheric dispersion, deposition and air quality, and calculation of the metrics themselves will change as scientific understanding evolves over time. Figure 1 presents an overview of the scientific and policy developments from the formation of EMEP [UNECE, 1984] to the ongoing revision of the Gothenburg Protocol, highlighting the 'new' science which was not available during negotiation of the original protocol. In summary, atmospheric dispersion modelling has evolved at the European scale from a 150km^2 resolution Lagrangian model calculating grid-average depositions to a 50km^2 Eulerian model which calculates ecosystem-specific deposition [Simpson *et al.*, 2003]. Models have also been developed at 5km^2/1km^2 resolution (eg. FRAME [Fournier *et al.*, 2005; Dore *et al.*, 2009] and EMEP4UK [Vieno *et al.*, 2009]) which are used by national integrated assessment models such as the UKIAM [Oxley *et al.*, 2003; Oxley & ApSimon, 2007].

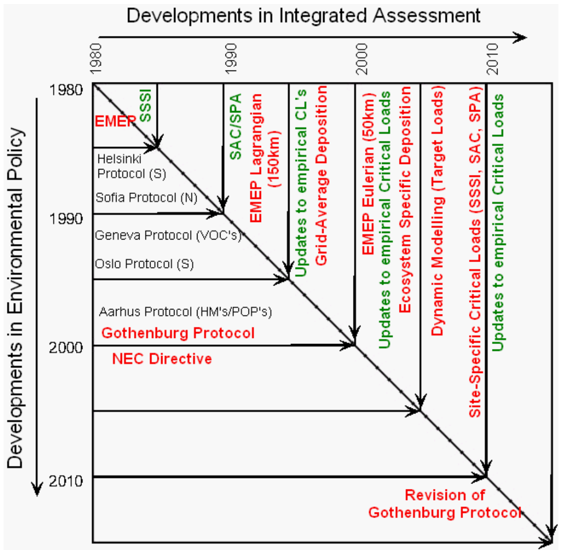

Figure 1 - Timeline which highlights the rapid rate of scientific developments in relation to policy in particular since negotiation of the Gothenburg Protocol [UNECE 1999]

Furthermore, developments in Critical Loads (CL) over the last two decades have progressed from the original acidity CL's for soils to include water and then ecosystems. These developments include changes to the chemical criteria and the development of and updates to empirical CL's for nutrient Nitrogen (CLnutN). These developments, based upon the best scientific evidence/knowledge, have been regularly reported in CCE Status Reports[1]; it is important to note that such updates can result in a *decrease* in the CL values (eg for empirical CLnutN for selected habitats), which implies an increase in exceedance for the affected ecosystems for no change in deposition rates. In addition to these developments to CL's, since negotiation of the Gothenburg Protocol dynamic modelling has been used to develop Target Loads [Hettelingh *et al.*, 2007; Posch *et al.*, 2003] which address *recovery* of ecosystems, whereas CL's only capture damage to ecosystems. Finally, whereas protocol negotiation focussed upon impacts on ecosystems and habitats nationally, subsequent application of Critical Loads to designated sites provides the ability to directly address impacts on SSSIs, SACs, SPAs and other Natura 2000 sites. [Hall *et al.*, 2003; 2008; http://jncc.defra.gov.uk/].

The effect of these developments is that revision of the Gothenburg Protocol can benefit from greater scientific understanding of ecosystem impacts than was available during the original negotiation, which is undoubtedly a positive development. However, great care is needed when reviewing the present state of policy metrics against the impacts predicted upon the basis of earlier science, to ensure that assessments are comparing like with like. In the following sections we highlight the impact spatial resolution can have upon quantifications of metrics and the difficulties that contrasting temporalities in science and policy can create in specifying a consistent basis for comparison.

2. SPATIAL RESOLUTION

Using exceedance[2] of ecosystem Critical Loads as an example metric, we show how the area of critical load exceedance increases both as spatial resolution of deposition data increases and as scientific advances in the

[1] Status reports from the *Coordination Centre for Effects* available from http://www.rivm.nl/en/themasites/cce/

[2] 'exceedance' is the rate of deposition of acidity (S/N) or nutrient N over and above the Critical Load for an ecosystem

representation of ecosystem specific deposition patterns are included in the models. As the EMEP Eulerian model was still under development at the time of the Gothenburg Protocol, the 150km^2 resolution Lagrangian model provided the basis for negotiations [EMEP, 1998]. The significance of this in relation to exceedance of Critical Loads is that subsequent calculations were based upon the 50km^2 resolution model [Simpson *et al.*, 2003]; immediately the area of exceedance increases, *ceteris paribus*, as shown in Figure 2.

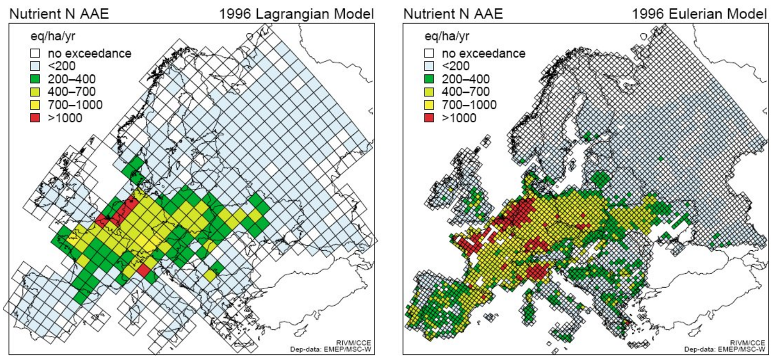

Figure 2 - A change in model resolution (from 150km^2 to 50km^2) significantly increases quantifications of exceedance and affects the spatial distribution of exceedance (from Posch *et al.*, 2001)

Subsequent scientific developments in relation to the GAINS model and all constituent sub-models have been documented by Amann (2010), with an external review of the modelling methodology highlighting further implications, arising from the development of higher resolution national models, in relation to exceedance of critical loads [EC4MACS, 2009].

Estimates of risk for ecosystem damage have already been reported to be a factor of 3 higher for eutrophication (30-50% for acidification) than estimates used during negotiation of the Gothenburg Protocol; this increase results from the shift to a higher resolution deposition model combined with updates to emissions and critical load data, the use of a chemical transport model for deposition, and land-cover specific deposition [Fagerli & Tarrason, 2006]. This effect is further exacerbated when exceedances are calculated by the national scale UK Integrated Assessment Model at 5km^2 resolution [Oxley *et al.*, 2003; Oxley & ApSimon, 2007].

Figure 3 presents a comparison between the effects of using grid-average or ecosystem-specific deposition to calculate exceedance of eutrophication critical loads; when EMEP grid-average deposition is used, *ceteris paribus*, no exceedance is apparent except in a few hot-spots. Table 1 quantifies these variations in relation to area exceeded for the different deposition rates, with almost half the ecosystem area displaying exceedance when high resolution ecosystem-specific deposition is used as compared with less that 1% area exceeded when low resolution grid-average deposition is used. This difference has also been exacerbated by the inability of EMEP to adequately capture localised orographic enhancement and the spatial pattern of deposition with the

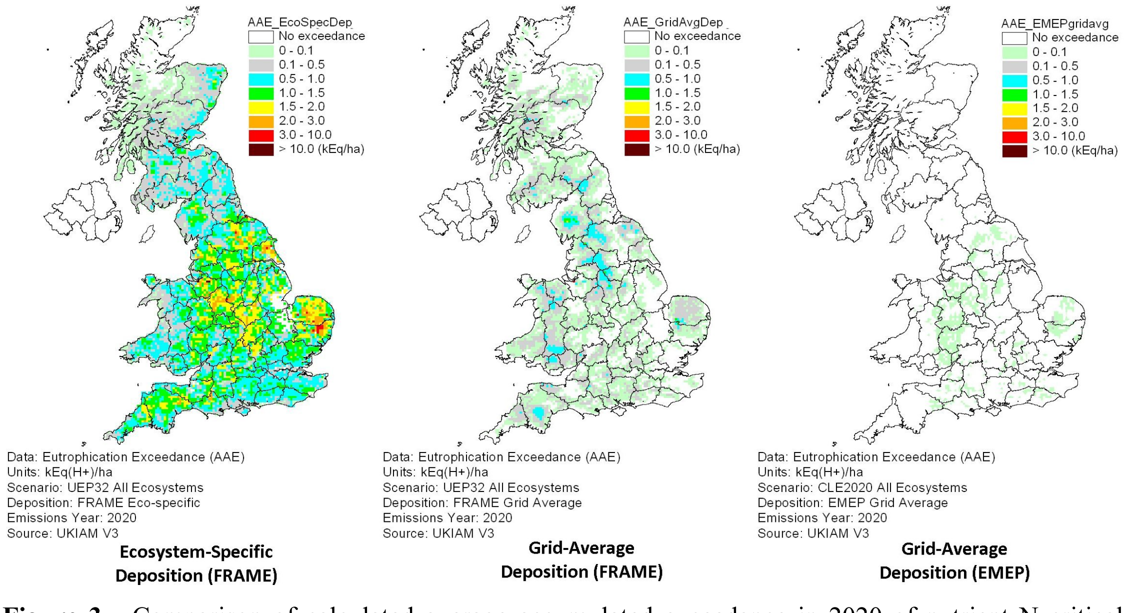

Figure 3 - Comparison of calculated average accumulated exceedance in 2020 of nutrient N critical loads in the UK (excluding N Ireland), for all ecosystems, based upon (a) FRAME ecosystem-specific deposition, (b) FRAME grid-average deposition, and (c) EMEP grid-average deposition

highest deposition in the Southeast whereas FRAME shows greatest deposition in the Northwest.

Table 1 - Nutrient nitrogen exceedances (all habitats) for Great Britain based upon projected emissions in 2020, comparing effects of alternative deposition models. Site-specific exceedances for SSSI's (see Figure 4) are also shown for comparison

Source of deposition data	Exceeded Area (km^2)	Percentage Area Exceeded	Accumulated Exceedance (kEq/year)
FRAME - Ecosystem specific	35,193	49.62	2,390,045
FRAME - Grid-average	25,965	36.61	796,013
EMEP - Grid-average	684	0.96	4,119
FRAME - Eco-specific (SSSI)	*13,375*	*64.61*	*1,175,693*

It is clear from these examples that spatial resolution (of atmospheric dispersion modelling, specification of critical loads, etc.) is a significant factor in quantifying exceedances. Whereas the Gothenburg Protocol was negotiated based upon information such as shown in Figure 2 (with maximum exceedance of nutrient N critical loads in the UK less than 0.2kEq/ha in 1996), current projections to 2020 suggest there may be exceedances greater than 10kEq/ha in some areas even having taken into account the considerable emissions reductions between 1996 and 2020. Note that Figure 2 is based upon 1996 Critical Loads whereas Figure 3 is based upon the most recent updates, which will also affect the calculation of area exceeded,

This numerical effect is further exacerbated when exceedances are calculated using $1km^2$ resolution deposition data or when analyses focus upon specific sites. Although quantification of exceedances (Figure 3) are useful for describing impacts nationally, this attaches equal importance to each ecosystem area, irrespective of whether it may be a Site of Special Scientific Interest (SSSI) or a European Natura 2000 site (which includes Special Areas of Conservation (SAC) designated under the EC Habitats Directive, and Special Protection Areas (SPA) protected by the EC Birds Directive). Thus, exceedances based upon site-specific critical loads have been developed [Hall *et al.*, 2006; 2007], providing policy makers with information quantifying exceedances for different habitats and features for individual sites (see Figure 4); it should be remembered, however, that these data relate to only a subset of ecosystems nationally and therefore results cannot be directly compared with those for all habitats as described in Figure 3.

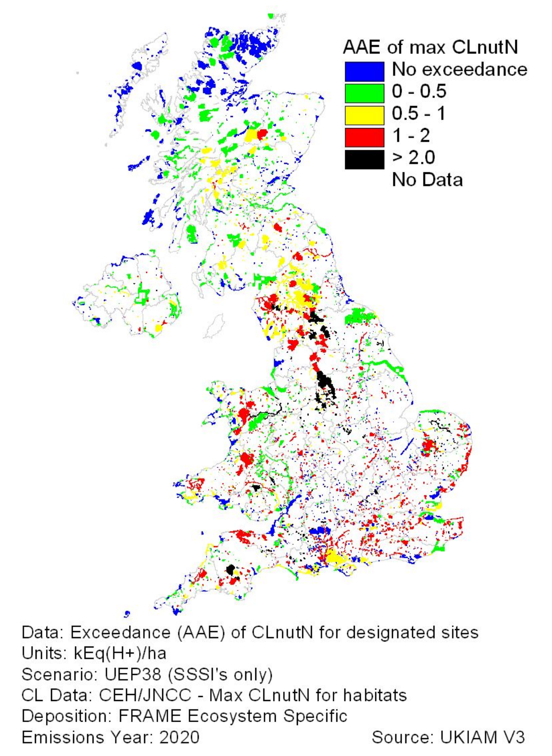

Figure 4 - Site-specific critical loads are defined for a subset of ecosystems nationally, with exceedances calculated for individual features and habitats

3. TEMPORALITIES

In addition to the spatial issues discussed above, advances in scientific understanding of critical loads, atmospheric dispersion and deposition rates can lead to apparently paradoxical quantifications of impacts when the effectiveness of policies are subsequently reviewed. The difference in exceedances displayed between Figure 2a and Figure 3a are not solely the effect of changing spatial resolution, but also influenced by more detailed critical load data [Hall *et al.*, 2008], updates to dispersion models [Simpson *et al.*, 2003; Dore *et al.*, 2009, Fagerli & Tarrason, 2006], and development of national scale integrated assessment models [Oxley *et al.*, 2003; Oxley & ApSimon, 2007].

In the context of reviewing the Gotheburg Protocol and quantifying the progress made towards protection of the environment relative to expectations, a choice is necessary between the extremes of using the latest emissions scenarios to drive the original 150km Lagrangian model or using the Gothenburg emissions to drive the latest models. Both options have advantages and disadvantages, with a pragmatic compromise involving the most up-to-date models but using grid-average deposition rates [TFIAM, 2007]; this overcomes the largest impact upon exceedance calculations (see Figures 3a/3b), but will under-predict the impacts especially for woodland ecosystems where deposition rates are higher than average.

An additional conceptual issue for policy makers to grasp is what is the meaning of critical loads and what does non-exceedance imply? Critical Loads are, by definition, *"a quantitative estimate of exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge"* [Nilsson & Grennfelt, 1988]. Any non-exceedance of critical loads therefore only implies that *no additional damage* is being done to these ecosystems. It does not imply that ecosystems have recovered or will recover from existing damage. To this end the concept of Target Loads was developed, based upon dynamic modelling, to describe *"a deposition pathway which ensures recovery in a given year and maintained thereafter"* [Hettelingh *et al.*, 2007]. Unlike critical loads, unique target loads do not exist for a given ecosystem since target loads also capture the target year, the protocol year (when emissions reductions start) and the implementation year (emissions reductions complete), each of which are definable by policy makers. Target Loads aim to achieve a certain rate of deposition in a target year; a rate related to achieving a particular chemical status such as a specific soil pH. However, achieving chemical recovery will not necessarily be accompanied by biological recovery, which could take much longer or not occur at all. This relationship between deposition, chemical response and biological response is shown in Figure 5, highlighting the potential delay times before recovery occurs. Clearly, more dramatic emissions reductions are required to meet target loads than to achieve non-exceedance of critical loads since *recovery* of damaged ecosystems is central to the concept of target loads [Jenkins *et al.*, 2003].

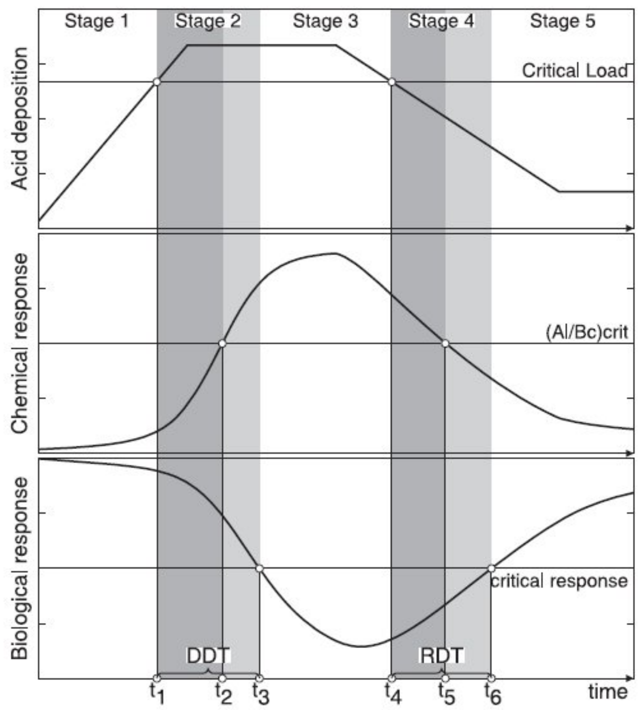

Figure 5 - 'Typical' past and future development of deposition effects on a soil chemical variable and the corresponding biological response in comparison to the critical values of those variables and the critical load derived from them. The delay between the (non)exceedance of the critical load, the (non)violation of the critical chemical criterion and the crossing of the critical biological response is indicated in grey shades, highlighting the Damage Delay Time (DDT) and the Recovery Delay Time (RDT) of the system. [from Posch, 2004]

The impact of these scientific developments upon policy making is to introduce a series of sometimes conflicting temporalities related to anthropogenic actions and environmental responses. The effect of interaction of such temporalities has been described elsewhere in the context of desertification in the Mediterranean, where farmers' daily irrigation needs can lead to aquifer depletion years or decades later, ultimately caused by short-term financial decisions to replace olive production with oranges [Oxley *et al.*, 2002]. Similar temporalities can be observed in relation to the protection of ecosystems under CLRTAP: emissions reductions now may promote ecosystem recovery in decades or even centuries, but this is further complicated by delays in negotiating and implementing emissions reductions. Subject to political feasibility of emissions reductions, it is clearly beneficial to aim for reductions to meet target loads, but since these are inherently dependent upon the timing of reductions, as delays in commencing action occur the extent of action required to achieve protection by a specified time will rapidly increase.

4. DISCUSSION

Policy makers demand use of the most up-to-date science and may thus discover an apparent *increase* in the ecosystem area exceeded when comparing against earlier assessments; such results may reflect changes to spatial resolution and/or the different and potentially conflicting temporalities of scientific developments and the policy making processes, as opposed to impacts in the real world. This highlights two key issues to be taken into account:

Firstly, any review of impacts of previous/existing policies should ensure that there is an appropriate degree of comparability between the tools used for negotiation and those used for review. For practical and scientific reasons reverting to the original tools in their entirety often makes little sense, hence the use of the most recent data and models for review of the Gothenburg Protocol but utilising grid-average deposition rates since ecosystem-specific deposition rates were unavailable at the time of negotiation [TFIAM, 2007]. It is important to ensure that policy makers are provided with appropriate information (data, maps etc.) which enable them to compare like with like, with sufficient clarification to make the differences between results explicit.

Secondly, given the extent of recent scientific developments, different policy makers and stakeholders will be observing and interpreting impacts from different spatial perspectives (from the local to the international) and focussing upon alternative metrics describing the state of the environment. Whereas national or European policy perspectives may be satisfied with nationally aggregated representations of average accumulated exceedances for all habitats, scientists' perspectives may have moved forward to observe metrics which look beyond exceedance of critical loads to metrics based upon target loads and issues of biodiversity, ecosystem health, species richness etc. [Slootweg, 2010]. Resolving these different perspectives of effects should therefore be made explicit during review/negotiation in order to arrive at a coherent collective understanding of both the rate and extent of emissions reductions required to achieve new targets of environmental protection.

Understanding these inherent implications and temporalities is important to ensure that any misrepresentation of the expected effects of emissions abatement policies can be avoided. If a policy scenario suggests that exceedance of critical loads can be reduced and this is interpreted as protecting ecosystems (as opposed to doing no further damage) there is a likelihood that further emissions reductions will be required in the future to ensure *recovery* from damage already caused. It is probable that significant reductions of the current exceedance of critical loads will result from emissions reductions already planned [TFIAM, 2007]. However, in order to promote ecosystem recovery, which may take decades or more, significantly greater emissions reductions will be required which may be economically or politically prohibitive; as suggested by Figure 5, target loads are dynamic and delays in either the 'protocol year' or the 'implementation year' will make it increasingly difficult to achieve ecosystem recovery.

The dynamic nature of target loads and potentially long recovery timescales for ecosystems demand that action should be sooner rather than later. However, political temporalities and delays in policy development, negotiation and implementation of protocols will also affect the extent of action required. Furthermore, in the current economic climate, for example, can policy makers be expected to prioritise the possibly expensive recovery of ecosystems by next century when *"it's the economy, stupid"*? [3] Such questions are beyond the scope of this paper but serve to highlight political perspectives influencing policy development which must also be considered when interpreting critical load exceedances and target loads.

5. CONCLUSIONS

In this paper we have described the spatial and temporal issues influencing modelled predictions of the effects of past and future emissions scenarios upon exceedance of Critical Loads, which are then used as the basis for policy developments towards protection of the natural environment. The publication of regular status reports[4] ensures openness and transparency of the data and methods used to calculate Critical Loads so that, when updates are made, policy makers are provided with detailed information on the impact these changes may have on the calculation of critical load exceedances so they understand the cause of the change and that results are not misinterpreted.

Finally, we have shown that the differences in results presented to policy makers may be due to one or more of the following:

- <u>Scientific developments in deposition modelling</u>: Ranging from EMEP Lagrangian to Eulerian dispersion models, from grid-average to ecosystem-specific deposition, and from 150km^2 to 50km^2 resolution, and at the UK scale from 20km^2 resolution in the late 1980's to 10km^2, 5km^2 and 1km^2 resolution deposition [Dore *et al.*, 2009; Smith *et al.*, 2000]; Differences in spatial patterns of modelled deposition with EMEP showing highest deposition in the Southeast of the UK and FRAME showing highest deposition in the Northwest with orographic enhancement in mountainous areas;
- <u>Improvements in the science and understanding of effects and the development of Critical Loads</u>: In the UK this started with an acidity CL map for soils, one for UK freshwaters (selected sites only), followed by ecosystem-specific acidity CL's and the use of different chemical criteria; For nitrogen the empirical

[3] Quote used by John Carville, Bill Clinton's presidential campaign strategist (1992)
[4] See http://www.rivm.nl/en/themasites/cce/publications/ and http://cldm.defra.gov.uk/Status_Reports.htm

CLnutN have been periodically reviewed/updated internationally with increased knowledge of impacts resulting in decreases in *CLnutN* for some habitats; or

- Changing emissions scenarios: Changes in the quantification and recording of emissions, the sources captured (including natural), and updates to future emissions projections.

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