4 Use of dynamic modelling forecasts of streamwater chemistry to derive future target loads for N and S in atmospheric deposition

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4.1 Introduction

Previous work undertaken using ICP IM data towards the objectives of the UNECE Convention on Long-range Transboundary Air Pollution has identifi ed the need for consistency between critical load assessments and dynamic modelling activities so as to pinpont Damage Delay Times and Recovery Delay Times in response to changing deposition patterns (Jenkins *et al*. 2003).

Here the work is continued for one of the sites studied previously, Afon Hafren (GB02), identifying the nature of any exceedances in critical loads and the response of chemical impact indicators. Subsequently Target Load Functions were to be quantifi ed for a range of pre-specifi ed chemical conditions deemed desirable. With these in mind projected recovery could be compared alongside the current expected nature of recovery as predicted by dynamic models. The process of quantifying acidifi cation critical load, exceedance and recovery was to be extended to assessments of eutrophication risk.

The objective of the work was to defi ne a generalised methodology for ICP IM sites for undertaking the assessments described above. The methodology will be suitable for sites for which suffi cient data are available for quantifi cation of critical loads and their exceedances and for parameterisation and successful calibration of the MAGIC dynamic model of freshwater acidifi cation (Cosby *et al*. 2001).

4.2 Materials and methods

4.2.1 Site description

The Afon Hafren (ICP IM site GB02: 52 29• Œ N 3 41• Œ W) lies in the Cambrian Mountains of mid-Wales. From its confl uence with the Afon Hore the 358 ha catchment forms the headwaters of the River Severn. The catchment rises from 355 m at the sampling station to 690 m at Blaenhafren. Podzols cover approximately 60% of the catchment with organic peaty soils comprising the remaining area. The underlying geology consists of Ordovician grits and Silurian mudstones and shales. Plantation forestry, primarily Sitka and Norway spruce, covers 41% of the catchment. The forestry forms part of the larger Hafren Forest which was planted primarily between 1948-1959 and 1963-1964. There has been some thinning and windblow, with removal of

windblown trees. Up to 1992 about 5% of the catchment consisted of recently felled forest. Moorland grasses and Calluna occupy the remainder of the catchment and are utilised for rough grazing.

4.2.2 Data availability

Stream water chemistry has been collected monthly at the Hafren site as part of the UK Acid Waters Monitoring Network and for modelling purposes these data were summarised as annual mean concentrations from 1998-2001. EMEP grids of S and N deposition were used as a basis for defi ning historical sequences of inputs. These were modifi ed using more locally relevant measurements and estimates. The values used under current legislation to 2010 are similar to those applied by Jenkins *et al*. (2003) but include lower estimates of reduced N deposition as a result of subsequent improved estimates from atmospheric modelling.

4.2.3

Modelling methods

For dynamic modelling purposes a version of the MAGIC model incorporating N processes was used (Cosby *et al*. 2001). The model was applied at an annual time step and calibrated to1988 using observed soil and stream chemistry data. The process of calibration followed the account given by Jenkins *et al*. (2003). Parameterisation requirements are in part provided for by observed data, notably lake and catchment morphological characteristics, soil chemistry and physical properties, base cation uptake by vegetation allied to information on forest history. These have been covered in detail elsewhere (e.g. Jenkins *et al*., 2003).

Two methods were used to determine critical loads of acidifi cation, as defi ned by Equation 1:

(1)

where:

Lcrit = acidity critical load (keq ha^{-1} yr^{-1})

BCw = base cation weathering rate (keq ha^{-1} yr^{-1})

BC*dep = non-marine base cation deposition (keq ha^{-1} yr^{-1})

ANCcrit = chosen value for critical ANC in streamwater receptor (keq ha^{-1} yr^{-1})

Q = annual fl ow (l ha^{-1} yr^{-1})

Firstly, the FAB model was used (Henriksen and Posch 2001). Secondly, MAGIC calibrated weathering rates of base cations were used to estimate critical loads. The base cation weathering rate is the sum of the component rates for Mg, Ca, Na and K. The critical loads were calculated for 3 different threshold receptor values of ANC in the streamwater (0, 20 and 40 ìeq l^{-1} respectively; converted to keq ha^{-1} yr^{-1}).

Critical loads for nutrient nitrogen are, by defi nition, land-use specifi c. The Hafren catchment comprises 41% forest, 30% grassland and 29% heathland. The constituent critical load calculations follow standard UK methodologies embodied in the International Programme on Mapping under the Working Group on Effects. Grassland and heathland nutrient N critical loads are empirically derived, the fi xing of values being based on observations of changes in species composition, plant vitality and soil processes. The critical load for forest is calculated using a mass balance method (Hall *et al*. 2003) whereby components for uptake, immobilisation, leaching and denitrifi cation are estimated and summed to give a total load. A catchment average

fi gure is calculated based on area weighting by land use type of the constituent critical loads.

For both acidifi cation and eutrophication impacts, critical load exceedances were calculated both for recent observed conditions (1998-2000) and under UK NECD projections based on the Gothenburg Protocol (2010). Acidifi cation exceedances (keq ha^{-1} yr^{-1}) were calculated using Equation 2:

(2)

where Nexp and Sexp are given in Equations 3 and 4 as follows:

(3)

(4)

Ndep = total N deposition (sum of wet and dry oxidised and reduced N deposition)

Sdep = total wet and dry non-marine S deposition

RhoN and RhoS are in-lake retention terms (set to 0 for Afon Hafren)

CLminN value was obtained from FAB model application.

In the circumstances of total N deposition not exceeding the value of CLminN, Equation 5 was applied:

(5)

where LCratio is the area ratio of lake to total catchment (set to 0 fro Afon Hafren). Target loads for deposition reduction completed by 2020 for a streamwater target ANCcrit of 20 ìeq l^{-1} were calculated. Two exercises were undertaken for illustrative purposes. Firstly calculation was done for SO_4, NO_3 and NH_4 individually in turn for a range of target years between 2030-2060, holding the loads of the other two species constant at the baseline predicted levels under the Gothenburg Protocol (TL Exercise 1). Some assumptions and constraints were made about the variation in time of atmospheric deposition in the context of the target loads. Linear interpolation was made between 2010 projected deposition and the target load for 2020 and loads were to remain constant after 2020. The effect through to 2100 was evaluated. Secondly, multiple critical load functions were calculated for the same range of target years, varying SO_4 and NH_4 deposition whilst holding NO_3 at 1988 levels. The functions were completed by evaluating the target SO_4 load at zero N deposition (TL Exercise 2).

4.3 Results

Figure 4.1 shows the output from the MAGIC model in terms of stream ANC. Large short term variations in simulated output during the 1990s are due to the availability of year-specifi c wet deposition data to drive the model. The observed annual means, taken from a weekly sampling program, are shown; these data are not the same as those used in model calibration. The calibration process involved optimisation against an assemblage of soil and stream chemical parameters, using, in the case of stream data, annual means of monthly observations from 1988 (for which the mean ANC was 12 ìeq l^{-1}). Large variations in seasalt inputs are propagated through to stream ANC variability and this is primarily the cause for any mismatch between observed and predicted values for specifi c years.

Critical load calculations are summarised in Table 4.1 which shows good agreement between the FAB and MAGIC-derived values. For these calculations a value for mean annual discharge at Afon Hafren of 1924 mm (1924000 L ha^{-1}) was used. Applying an ANCcrit value of 20 ìeq l^{-1}, the minimum critical load of nitrogen (CLminN) as calculated by FAB is 0.37 keq ha^{-1} yr^{-1}, an N deposition below which all deposited N is assumed immobilised by soil and plant uptake. Therefore the maximum critical load of N above which harmful effects are seen is equivalent to the sum of the Lcrit and CLminN values. In reality, unique critical loads cannot be generally defi ned due to the contribution of S to acidifi cation. Hence a critical load function for the Afon Hafren (Figure 4.2) can be plotted in relation to present day and Gothenburg protocolprojected deposition. At Afon Hafren it is clear that legislation will prove effective in lowering deposition below critical levels for acidity.

Table 4.1 also includes the landcover-specifi c critical load values for nutrient N and an overall estimate for the Afon Hafren (CLnutN). It is less certain that legislation will reduce deposition below critical levels in terms of eutrophication risk. Table 4.2 shows exceedance values. The uncertainty is noteworthy for coniferous woodland ecosystems where the value of CLnut(N) is subject to uncertainty of +/-0.13 keq ha^{-1} yr^{-1} (Posch *et al*. 2003).

Table 4.1 Critical loads calculations (values are in keq ha^{-1} yr^{-1}).

	acidifi cation: ANCcrit values (ìeq l^{-1})			**eutrophication**
	0	**20**	**40**	**nitrogen**
FAB Lcrit	1.45	1.06	0.68	
MAGIC Lcrit	1.39	1.01	0.62	
grass/heath				1.07
woodland				0.79
catchment average (CLnutN)				0.95

Table 4.2 Critical load exceedance calculations under present day and UK NECD (Gothenburg protocol) deposition loads (values are in keq ha^{-1} yr^{-1}). Acidifi cation exceedance is based on ANCcrit = 20 ìeq l^{-1}.

Present day deposition Gothenburg Protocol deposition
acidifi cation eutrophication acidifi cation eutrophication

FAB model 0.16 -0.58
MAGIC-derived model 0.21 -0.52
grass/heath 0.23 -0.29
woodland 0.52 -0.01
catchment
average (CLnutN)
0.35 -0.18

Figure 4.1 MAGIC model performance in terms of stream ANC at the Afon Hafren.

When critical loads are evaluated in the context of dynamic modelling (Figure 4.3a) it can be seen that for the Afon Hafren deposition has been above the critical load and the stream chemical criterion violated for many years. It is predicted that measures of recovery, currently underway, will result in the stream criterion no longer being violated in 2010. However, MAGIC model output through to 2100 suggests concern over whether the recovery is sustainable. The impacts of climate change may have an important bearing on whether or not stream water will remain above ANCcrit.

For stream ANC, damage delay and recovery delay times (Figure 4.3b) are predicted to be short (3 years and 2 years respectively). The biological responses (not covered here) are typically longer. Figure 4.3c shows that soils are of low base saturation and have always been in violation of the UK-defi ned critical levels (equivalent Ca: Al ratio = 1). It is arguable that the ratio is not very stable, and it is instructive to calculate the soil base cation pool (Posch *et al*., 2003). The MAGIC model estimates levels of 6.1 eq m^{-2} in 1848 decreasing to 3.6 eq m^{-2} at present day and recovering gradually to 4.2 eq m^{-2} by 2100.

Analysis for the Afon Hafren has shown that streamwater chemical targets can be attained prior to 2010 under current legislation. Therefore, for this site, target loads do not need to be defi ned to drive further deposition reductions. TL Exercise 1 revealed that for all streamwater chemistry target years the target load for total acidifying deposition is at or above critical levels. Guidelines specify that whenever target loads are higher than the critical load the target should be set to the critical level so as to ensure future ecosystem protection. Unfortunately, dynamic modelling reveals that achieving deposition targets below the critical level (for example those shown on Figure 4.4, the plot of multiple critical load functions) may not be suffi cient to offer future protection. Protection is guaranteed only up to the target year because stream ANC is predicted to deteriorate slightly if deposition is held constant up to 2100 at the levels achieved in 2020. This occurs regardless of the 2020 deposition loads specifi ed. Hence, as illustrated by TL Exercise 2, the target loads for non-violation of ANCcrit in 2060 are more stringent than for 2030 as values above 20 ìeq l^{-1} must be maintained for longer. A function for the year 2100 has been added to Figure 4.4, showing that deposition projected under Gothenburg Protocol is more than adequate to ensure streamwater protection until 2100.

Figure 4.2 The critical load function for Afon Hafren defi ned using FAB and MAGIC model methodologies. Current deposition and projected 2010 deposition under the Gothenburg Protocol are also shown.

Figure 4.3 Past and future development of acid deposition effects on chemical indicators illustrated using the MAGIC model: (a) overview from 1850-2100 of acid deposition loading and streamwater ANC using a MAGICderived value for Lcrit and an ANCcrit of 20 ìeq l^{-1}, (b) concept of the damage delay time

(DDT) and recovery
delay time (RDT) for
streamwater ANC
(c) responses of soil
chemical variables.

4.4

Conclusions and ongoing priorities

The modelling study of the Afon Hafren has shown that streamwater chemistry is successfully recovering from the effects of acid deposition in response to reductions in emissions under current legislation. It is predicted that stream ANC will have recovered to above 20 ìeq l^{-1} by 2010 and, under current climatic conditions, will remain at safe levels throughout the rest of the 21st century. In this context no further deposition reductions would be necessary. There is fairly good consistency between calculations of critical loads of incoming acidity as estimated by dynamic modelling (MAGIC) and the FAB model. For streamwater, dynamic modelling from 1850-2100 suggest that with respect to critical deposition levels damage delay times and recovery delay times are short (less than 5 years). However, delay times with respect to soil chemistry cannot be defi ned as the soils are poorly buffered with parameters of acidifi cation always below critical levels. Dynamic modelling suggests recovery will be slower (or at least less complete by 2100) than for streams.

For use at ICP IM sites, the work presents a method for dynamic modelling, critical load calculation and, if necessary, defi nition of target loads to ensure recovery to safe levels of a chosen target chemical parameter, either in stream or soil environment. However, the Afon Hafren study illustrates complications that may arise in terms of interpreting dynamics of predicted ANC in the future. In this case stream ANC is predicted to decline slightly some years after current legislation is achieved (Figure 4.1). Therefore, a target load function cannot be defi ned in the same way as for a site that is predicted to show continued and sustained recovery in response to decreasing emissions.

Other complications concern uncertainties over how N dynamics should be treated by dynamic models and uncertainties due to changing climate. The steady-state FAB model suggests that for the Afon Hafren all N deposition above 0.37 keq ha^{-1} yr^{-1} will result in leakage at some stage in the future. Dynamic modelling suggests that signifi cant leakage did not occur in the Hafren until deposition of >1 keq ha^{-1} yr^{-1} occurred. The predicted long-term levelling off in recovery and slight decline in ANC

Figure 4.4 Target Load Functions required for 5 specifi ed future years of streamwater chemistry targets. The functions
are shown in relation to the site critical load functions and to current and projected deposition.

in the 21st Century is in part due to the impact of N leaching. Including N dynamics in MAGIC (Cosby *et al*. 2001) allows N leaching to be governed by the C/N ratio of the soil organic compartment. The conceptualisation allows for N loss to increase into the future even when deposition inputs of N decline, thereby simulating the effects of N saturation. Therefore the dynamic modelling undertaken here represents a worse case scenario. There is much uncertainty regarding future dynamics of N leakage in catchments.

When considering climate change effects, the sensitivity analysis performed by Wright *et al*. (2006) was extended to the Afon Hafren. The results of this analysis reveal that the model is sensitive to future increases in soil decomposition (N release), having an adverse effect on the recovery of streamwater ANC (potentially violating critical levels once again before 2050) and soil base saturation. In the shorter term (up to 2050) increases in seasalt deposition appear likely to also have an adverse effect on ANC but this is unlikely to result in a decrease below 20 ìeq l^{-1}. Some more desirable effects are projected: increasing seasalt inputs is benefi cial to soil base saturation and increasing weathering rates under higher temperatures will enhance recovery in both soils and stream water.

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