Environmental Science and Policy 73 (2017) 124–132

Contents lists available at ScienceDirect

Environmental Science and Policy

journal homepage: www.elsevier.com/locate/envsci

An analysis of the likely success of policy actions under uncertainty: Recovery from acidification across Great Britain

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ARTICLE INFO

Keywords:
HARM
GLUE
Uncertainty
Critical loads
Soil acidification

ABSTRACT

In the context of wider debates about the role of uncertainty in environmental science and the development of environmental policy, we use a Generalised Likelihood Uncertainty Estimate (GLUE) approach to address the uncertainty in both acid deposition model predictions and in the sensitivity of the soils to assess the likely success of policy actions to reduce acid deposition damage across Great Britain. A subset of 11, 699 acid deposition model runs that adequately represented observed deposition data were used to provide acid deposition distributions for 2005 and 2020, following a substantial reduction in SO_2 and NO_x emissions. Uncertain critical loads data for soils were then combined with these deposition data to derive estimates of the accumulated exceedance (AE) of critical loads for 2005 and 2020. For the more sensitive soils, the differences in accumulated exceedance between 2005 and 2020 were such that we could be sure that they were significant and a meaningful environmental improvement would result. For the least sensitive soils, critical loads were largely met by 2020, hence uncertainties in the differences in accumulated exceedance were of little policy relevance. Our approach of combining estimates of uncertainty in both a pollution model and an effects model, shows that even taking these combined uncertainties into account, policy-makers can be sure that the substantial planned reduction in acidic emissions will reduce critical loads exceedances. The use of accumulated exceedance as a relative measure of environmental protection provides additional information to policy makers in tackling this 'wicked problem'.

1. Introduction

The many types of uncertainty that can affect policy making and how these can be presented to and then handled by policy makers, have become topics of increasing interest. Schneider and Kuntz-Duriseti (2002) considered uncertainty in climate change policy. They suggested that whilst one approach is to reduce (bound) the uncertainty by collecting more data, more understanding and building better models, the other approach is to reduce the effects of (manage) any uncertainty in understanding by taking it into account in policy making. This second approach can be traced back to ideas about ecosystem resilience and recovery after disturbance developed in the 1970s. Refsgaard et al. (2007) in a review of uncertainty in the context of water management, suggested that uncertainty in its widest sense can usefully be regarded as the degree of confidence a decision maker has about possible outcomes and/or the probabilities of these outcomes. Uusitalo et al. (2015) suggested that uncertainty analysis can provide decision makers with a realistic picture of possible outcomes, in a context where results are going to be better or worse, not true or false, i.e. that environmental problems are 'wicked problems'. Whilst some types of uncertainty are unquantifiable, other types can be quantified through approaches such as sensitivity analysis, the use of multiple models and exploring the impact of parameter uncertainty. Here we take a quantitative approach to uncertainty in the context of recovery from the problem of acidification in Great Britain. We quantify and then combine the uncertainties in outputs from one acid deposition model and one measure of ecosystem health to assess whether current emissions reduction policies are likely to deliver ecosystem protection. We believe that this is the first effort to combine the uncertainties in both these elements in a single assessment.

European policymakers have been concerned about the acidification of sensitive soils and terrestrial ecosystems, driven by emissions of acidic species, sulphur dioxide (SO_2) and nitrogen oxides (NO_x) since the 1970s. These concerns have led to concerted policy actions within

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http://dx.doi.org/10.1016/j.envsci.2017.03.007
Received 21 September 2016; Received in revised form 16 March 2017; Accepted 17 March 2017
Available online 03 May 2017

the United Nations Economic Commission for Europe (UN ECE) and the European Union (EU), designed to reduce emissions and hence, the damaging deposition. The UN ECE agreed the Convention on Long-Range Transboundary Air Pollution (CLRTAP) in 1979 and has since promulgated a series of Protocols to the Convention, initially involving SO_2 and NO_x separately and then combined with ammonia (NH_3) under the Gothenburg Protocol (1999), referred to as the 'Multi-pollutant, Multi-effect Protocol'. A revision of the Gothenburg Protocol was agreed in 2012 (referred to here as RGP, see Amann et al., 2012; Reis et al., 2012). The EU has tackled the need to reduce emissions through a series of directives focussing initially on Large Combustion Plant (1988 and 2001), giving rise to the National Emission Ceilings Directive (NECD). In 2005, the EU put forward its Thematic Strategy on Air Pollution, Clean Air for Europe (CAFÉ) and under this framework is renegotiating the NECD with current commitments extending to 2029, with new commitments after 2030 (for an assessment of the NECD see Hettelingh et al., 2013a). Within these policy contexts, the chosen measure of ecosystem sensitivity was the critical load (CL) (Hettelingh et al., 1995), where the CL is the amount of deposition the chosen receptor can apparently tolerate without damage being likely (Bull, 1992). Where deposition was greater than (exceeded) the CL, damage was assumed to occur. CLs have been developed for a range of receptors (soils, freshwaters and a variety of terrestrial ecosystems) using a number of different methodologies (for the latest UK information see http://www.cldm.ceh.ac.uk/, for details of the most recent changes in methodology across Europe see Slootweg et al., 2015). It has been long recognised that there is variability between representations of CLs and that there are uncertainties in their calculation (see Zak et al., 1997), but CLs remain central to policymaking in this area and are an accepted risk assessment tool (Hettelingh et al., 2013b; Holmberg et al., 2013). The success of any emissions reduction policy is gauged by the resulting reduction in CL exceedance and system recovery (chemical and biological) (Posch et al., 2012), recognising that any system is unlikely to recover to exactly its pre-acidification state (Helliwell et al., 2014).

As it soon became evident that CLs would not be achievable across the whole of Europe in the foreseeable future, the concept of 'gap-closure' was adopted to formulate acid deposition policies (see Amann et al., 2012 and the references therein). Gap closure implies reducing CL exceedance by a given fraction, say 50%, and then using integrated assessment modelling to find an equitable and fair distribution of emission reductions across the European countries to achieve the gap-closure target. Whilst this is a pragmatic approach, the approach cannot use meeting CLs as its optimisation target (and hence cannot guarantee complete ecosystem protection) and so a new index of environmental protection has been defined in terms of reducing 'accumulated exceedance' (AE) which captures both the magnitude and areal extent of exceedance. This index requires the combination of both CL and acid deposition data, both of which are uncertain.

The historical reductions in emissions across the EU-28 countries (by 87% for SO_2, 54% for NO_x and 27% for NH_3 since 1990) (European Environment Agency (EEA), 2015) and measured decreases in deposition, have been reflected by measurable recovery in pH and acid neutralising capacity in many surface waters (Battarbee et al., 2014; Kernan et al., 2010) and reductions in CL exceedance (De Wit et al., 2015; RoTAP, 2012). Forward projections of current emission reduction commitments and the agreement of any additional reductions, however, depend on the application of atmospheric transport and deposition models, whose outputs can then be compared with CLs to assess the likely resulting environmental improvement (gains). Acid deposition models are uncertain because the parameterisations on which they are based and the input parameters that are fed into them, both contain simplifications and assumptions. CL are also uncertain, as described above. It is important, therefore, that policymakers have confidence in the outcomes of this modelling procedure (deposition and CL exceedance) given all the uncertainties inherent in both the atmospheric transport and CL models and can be assured that the higher costs of additional future emission reductions (assuming that the cheaper options have already been adopted) will actually increase protection of sensitive ecosystems and that recovery from acidification will continue. Two questions therefore arise: 1) can we can really be sure that the emissions reductions proposed to reduce AE will produce discernible environmental improvement or will they be lost in uncertainty? and 2) does the change of approach from an absolute target (CL exceeded or not) to a relative one (based on accumulated exceedance), change our perception of environmental improvement? Here we address both these questions. The concerns around the implications of scientific and model uncertainty for policy making that we address here in relation to acidification are relevant across a range of environmental issues.

We address our two questions about the impact of scientific uncertainty on achieving environmental protection, by exploring the impact of uncertainties in one atmospheric transport and deposition model, the Hull Acid Rain Model (HARM, Metcalfe et al., 2005) and one representation of CL (for soils), based on the Skokloster classification, by comparing estimates of accumulated exceedance of CL in 2005 and 2020 and assessing the likelihood of environmental protection across Great Britain (GB). This builds on an initial assessment of the impacts of uncertainty in HARM on CL exceedance across Wales reported by Heywood et al. (2006a). We provide a brief description of HARM and set out our approach to representing uncertainty in HARM and the CL for soils data set. We describe how we have combined estimates of deposition and sensitivity to acidification (CLs) to yield estimates of accumulated exceedance (AE) and how we have assessed the significance of the modelled changes. Our method is illustrated with reference to one 10 km × 10 km grid square in the Peak District in northern England, before going on to present and discuss the results for the whole of GB and consider the wider implications of this more rigorous approach for policy making.

2. Methodology

2.1. HARM and the GLUE framework

HARM is a receptor-orientated Lagrangian statistical model which is driven by emissions of SO_2, NO_x and NH_3 across the UK and the wider European area. Over a number of years, the model has been used to help in the formulation of acidification control policies in the UK. It provides estimates of wet and dry sulphur and nitrogen (both oxidised and reduced) depositions at 10 km × 10 km spatial resolution across the UK. Further details of the model are given elsewhere (Dore et al., 2015; Metcalfe et al., 2005; Whyatt et al., 2007). Here, HARM has been run using 2005 emissions estimates for SO_2, NO_x and NH_3 sources within the UK and the rest of Europe. An illustrative, gap closure type, scenario was then applied to simulate a possible 2020 emission situation involving a 35% reduction in SO_2 emissions and a 33% reduction in NO_x emissions (no reduction was applied to NH_3 emissions). This 2020 scenario was developed before the RGP was agreed, but is broadly consistent with the UK's current Gothenburg commitments (DEFRA, 2015). Our SO_2 emissions lie within the likely ranges for 2020, but our NO_x emissions are a little high. It is also proposed that UK NH_3 emissions will decline by 2020, by around 12% from the figure used here. Because our results are likely to be influenced by the absolute magnitude of the deposition reduction as well as the spatial distribution of any reduction, our illustrative or hypothetical reduction should be within the bounds of current projections.

Policymakers require that any model used for environmental policy formulation should reproduce real world behaviour adequately. In the present context, this means that an acid deposition model should reproduce the observed acid deposition fields (see for example Dore et al., 2015; Fagerli et al., 2003; NEGTAP, 2001; RoTAP, 2012). However, any comparison of model results with observations is never perfect. Inevitably, there is likely to be good agreement for some sites or