Science of the Total Environment 407 (2009) 5605–5619

Contents lists available at ScienceDirect

Science of the Total Environment

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

Dynamic modelling of the response of UK forest soils to changes in acid deposition using the SAFE model

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ARTICLE INFO

Article history:
Received 3 December 2008
Received in revised form 25 June 2009
Accepted 27 June 2009
Available online 5 August 2009

Keywords:
Acidification
Forest soils
Critical loads
Recovery
SAFE model

ABSTRACT

Both observed and modelled data have been examined from the ten UN-ECE Level II forest intensive monitoring sites in the UK to determine the changes and potential impact on soil solution chemistry resulting from changes in acid deposition inputs. The sites represent a range of forest tree types, soil sensitivities and pollutant deposition inputs found in the UK. The dynamic biogeochemical SAFE model was used to explore temporal changes in soil and soil solution chemical parameters that have been used as indicators for potential forest ecosystem and tree damage in national and international assessments of critical loads. The observed data and model results show that there is significant inter-site variation. The model indicates that the historical pollutant inputs have resulted in significant soil acidification at most of the sites. Model predictions generally match current day observations. Recently declining pollutant inputs have reduced and in some cases reversed the trend of increasing soil acidification. A discussion of the results in terms of critical loads, recovery, their wider implications and uncertainty is presented.

1. Introduction

Detrimental changes to soils and water ecosystems from the impact of acid deposition and acidification have led to the development of national and international policies aimed at reducing emissions of acidifying pollutants. For example, in Europe reductions have been agreed as part of the Gothenburg Protocol in 1999, which targets emissions reductions for sulphur dioxide, nitrogen oxides and ammonia of 75%, 50% and 12%, respectively, by 2010 compared with the 1990 baseline (Jenkins and Cullen, 2001). Increasingly, such policies have required an effects-based approach to proposing solutions for environmental problems and implementing emissions reductions in a targeted and cost-effective way. As a part of this process there is a need to provide policy makers with information showing the consequences of changing emissions on the environment and their associated ecosystems. Current policy development, both within the UK and Europe, is largely being assisted by consideration of critical loads, agreed under the Gothenburg Protocol, and the UN-ECE Convention on Long-range Transboundary Air Pollution (CLRTAP). This Protocol is based on effects, in particular it uses critical loads and critical levels and their exceedances as scientific basis for optimised emission reduction policies. The calculation and mapping procedure follows the Manual of the International Co-operative Programme on Modelling and Mapping Critical Loads and Levels and Air Pollution Effects, Risks and Trends (ICP Modelling and Mapping). A critical load is defined as 'a quantitative estimate of the 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 and Grennfelt, 1988). Where the critical load for an ecosystem is exceeded by inputs of acidifying pollutants, ecosystem damage may occur over an unspecified timescale. Comparison of critical loads with inputs of atmospheric pollution on a spatial basis can then be used to predict ecosystems which are at risk of damage. Critical loads are set through the selection of a chemical criteria and a threshold value that once exceeded will give rise to ecosystem damage. For forest ecosystems in Europe the most commonly used chemical criteria in soil solution are pH, aluminium (Al) concentrations and base cation to aluminium (Bc:Al) ratios and in soilsbase saturation (Hall et al., 2001a). However, critical loads are a steady-state approach with no time scale set in which these chemical criteria will be reached and the onset of ecosystem damage or/and recovery may occur. The steady-state approach does not provide information on if, when and for how long the chemical criteria and critical load have been exceeded. Dynamic models describing biogeochemical processes and the rates at which they occur provide a method of assessing the time frame over which changes to soils and waters may occur.

There are a number of dynamic acidification models that have been developed to improve the understanding of the acidification process, assess chemical criteria used in the critical loads, and the potential for

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doi:10.1016/j.scitotenv.2009.06.043

ecosystem impact and recovery. Understanding these attributes and their intricate linkage are important if policies aimed at reducing the detrimental effects to ecosystems are to be effective. The dynamic models in most common use are MAGIC (Cosby et al., 1985), SMART (De Vries et al., 1994) and SAFE (Warfvinge et al., 1993).

The UK submission to the United Nations-Economic Commission for Europe (UN-ECE) work programme underpinning the Gothenburg Protocol and its reviews have largely focussed on the detrimental effects to soils and the ecosystems they support. The approach used to determine and set critical loads in the UK for terrestrial ecosystems has been incremental. Moving from early research work which developed a generalised empirical critical load based on soil weathering rates applicable to a variety of ecosystem types (Hornung et al., 1995; and the references therein), to a single forest ecosystem critical load based on a simple mass balance approach, described in Langan et al. (2004). This latter piece of work suggested that there were significant areas of UK forests that were at risk from critical load exceedance giving rise to potential damage to forest productivity and vitality. However, as noted above, the underlying methodology of assessment utilises assumptions of steady state. In order to evaluate the system dynamics and provide a time frame for any ecosystem response there is a need to use dynamic models. Therefore, the next stage of the process is to consider the dynamics of forest ecosystems to changes in emissions policy and the possible consequences for soil solution and forest vitality.

This paper reports the first application of the dynamic biogeochemical Soil Acidification of Forest Ecosystems (SAFE) model to the UK Level II long-term intensive forest monitoring plots. These plots were established as part of the UN-ECE International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests) (EC, 1998) underpinning the Convention on the Long-Range Transboundary of Air Pollutants. The aim of this paper is to document the results in relation to critical loads and to comment on the possibility of ecosystem damage and recovery according to the changing acid pollutant deposition inputs as predicted by the modelling undertaken.

2. Methods and data

2.1. Modelling approach

The SAFE model is a largely mechanistic, dynamic multi-layer (horizon) soil chemistry model. It includes process-oriented descriptions of cation exchange reactions, sulphate adsorption, chemical weathering of minerals, solution equilibrium reactions involving carbon dioxide, organic acids and Al-species as well as leaching and accumulation of dissolved chemical components. As for nitrification, SAFE assumes all ammonium to be either taken up or nitrified in the top soil layer. A central component to the model is the calculation of base cation release from mineral weathering in the soil. This is done through the use of the PROFILE model (Sverdrup and Warfvinge, 1993a,b) the code of which is embedded within SAFE. In addition to horizon specific, chemical and physical soil characterisation, SAFE requires time series of input data quantifying atmospheric deposition, net uptake of nutrients, litterfall, canopy exchange, net mineralisation and precipitation inputs. These are derived from site measurements, data from present day deposition and generalised European assumptions about past changes in pollutant emissions together with the computer model MAKEDEP (Alveteg et al., 1998). These time series of changes generated with forest growth, pollutant and deposition are used to drive the SAFE model. The input data requirements for the models are given in Table 1. Of course, the included processes only represent a selection of naturally occurring processes in the soil. Among the processes that have not been included are sulphate adsorption and a series of reactions that may change the cation exchange capacity (CEC) of the soil matrix, store sulphur irreversibly or affect the acid neutralising capacity (ANC) balance in certain soils. There is no hydrological model in SAFE and thus it is not possible to model horizontal flow. A full description of the model and its data requirements can be found in Warfvinge et al. (1993) and Alveteg (1998).

2.2. Chemical criteria

In order to relate changes in soil solution chemistry to forest health a number of chemical criteria have been used. Sverdrup et al. (1990) reviewed the available literature to suggest a number of criteria. Other works, notably the earlier work by Ulrich (1984) and the synthesis of de Vries et al. (1994), have provided an international framework for the setting of critical loads for forest soils in relation to selected criteria (Table 2). A simplifying assumption in these studies is that these criteria apply to the rooting zone taken to be to 50 cm depth. A further review of studies by Sverdrup and Warfvinge (1993a,b) proposed a range of critical Bc:Al ratios for different species, including trees and other ecosystems that could be extrapolated from the studies reviewed by the authors. For woodlands growing on soils with an organic top soil (>30 cm) overlying a mineral soil, or peat soils (>50 cm depth), a critical pH criteria has been used. For woodland ecosystems rooting in inorganic or organic top soils a critical pH of 4.0 has been suggested by de Vries et al. (1994) and Sverdrup et al. (1990). At a European scale, Werner and Spranger (1996) suggest this value equates to a critical inorganic aluminium concentration of 200 ueq l^{-1} which the authors indicate is damaging to forest functioning through its effect on the fine roots and nutrient uptake. As with the work reported by Langan et al. (2004), this study uses a combination of these criteria to evaluate the potential risk of critical load exceedance at the sites investigated.

2.3. Level II forest monitoring sites

Within the UK the forest sites identified to have sufficiently detailed data to support running dynamic models were the ten UN-ECE Intensive Forest Health monitoring sites (Level II) (de Vries et al., 2003) operated by the Forest Research between 1995 and the present (Durrant, 2000;Vanguelova et al., 2007a). These plots consist of stands of oak, Scots pine and Sitka spruce with standardised management practices, including thinning and brashing during their growth cycle (Table 3). The plots vary in planting year between 1920 and 1974 and cover a range of production classes from 4 to 22 m^3 ha^{-1} yr^{-1}. The soils at these sites cover a considerable range of parent materials, from base-rich aeolian sand over chalk drift to base-poor mudstones and quartzites (Table 4). The site locations cover a gradient of pollution deposition inputs and represent the different forest ecosystems, soil sensitivities and pollution deposition histories that exist in the wider total forest stock of the UK (Fig. 1).

At each site both survey and monitoring are undertaken (Table 5). Survey information consists of growth indices (stand height, diameter and volume), tree nutrient content by compartment, tree health (foliar chemistry and crown condition), ground vegetation, litter and soil chemistry. The survey data on tree growth and nutrient content of tree tissue were used to calculate annual biomass uptake of cations. Monitoring consists largely of deposition and soil water fluxes and solution chemistry. Soil solution chemistry has been monitored for nine of the sites over differing periods of time (Tables 4 and 5). Soil solutions are collected every two weeks by suction lysimeters (PRENART equipment ApS), following a few months settling in period. At each site arrays of these lysimeters are taken to represent soil solution at two depths (10 and 50 cm). Further details of the sampling and the results are given in Durrant (2000; Broadmeadow et al., 2004; Vanguelova et al., 2007a, 2009). In addition to the Forestry Commission data a significant amount of information for the sites was analysed and collated by Kennedy (1997) in order to determine soil