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Heterogeneity of atmospheric ammonia at the landscape scale and consequences for environmental impact assessment

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ABSTRACT

We examined the consequences of the spatial heterogeneity of atmospheric ammonia (NH_3) by measuring and modelling NH_3 concentrations and deposition at 25 m grid resolution for a rural landscape containing intensive poultry farming, agricultural grassland, woodland and moorland. The emission pattern gave rise to a high spatial variability of modelled mean annual NH_3 concentrations and dry deposition. Largest impacts were predicted for woodland patches located within the agricultural area, while larger moorland areas were at low risk, due to atmospheric dispersion, prevailing wind direction and low NH_3 background. These high resolution spatial details are lost in national scale estimates at 1 km resolution due to less detailed emission input maps. The results demonstrate how the spatial arrangement of sources and sinks is critical to defining the NH_3 risk to semi-natural ecosystems. These spatial relationships provide the foundation for local spatial planning approaches to reduce environmental impacts of atmospheric NH_3.

1. Introduction

Most atmospheric ammonia (NH_3) originates from agricultural activities (Misselbrook et al., 2000; Van der Hoek, 1998). Intensive livestock farming, i.e. large pig and poultry units, represent substantial NH_3 point sources, due to their high stocking density. Other NH_3 sources include biomass burning, fuel combustion and industrial processes such as the production of nitrogen (N) fertilisers (Bouwman et al., 1997). High atmospheric NH_3 concentrations are directly toxic to plants through stomatal uptake as soon as the uptake exceeds the detoxification capacity (Fangmeier et al., 1994). Ammonia deposition (and deposition of other forms of reactive N) can lead to eutrophication and acidification of sensitive ecosystems, causing changes in biodiversity (Cape et al., 2009b; Cellier et al., 2009; Krupa, 2003; Pitcairn et al., 2009). Studies have been conducted to quantify the effect of NH_3 emission sources on surrounding ecosystems, e.g. Fowler et al. (1998) quantified concentrations and deposition fluxes within 300 m of a poultry farm in Scotland using measurements and deposition modelling, showing rapidly decreasing concentrations with distance from the source. Pitcairn et al. (1998, 2002) analysed the impact of such deposition fluxes on woodland flora and Frati et al. (2007) studied the effect of pig farm emissions on sensitive vegetation (lichens). Sutton et al. (1998) compared deposition estimates based on different scales, ranging from field to landscape to national scale and concluded that, due to the spatial variability of NH_3, the quality of an environmental impact assessment is dependent on the spatial resolution of the deposition data used. Dragosits et al. (2002) provide a more detailed analysis of the landscape study in Sutton et al. (1998): Emission, transport and deposition were modelled within a 5 km × 5 km landscape in England at a 50 m grid resolution; however, no NH_3 measurements were made to verify the estimates. Theobald et al. (2001) and Dragosits et al. (2006) focused on strategies to reduce the effect of emission hotspots on ecosystems by locating tree belts around the sources, indicating the importance of relative spatial location of sources and sinks, and assessed possible landscape planning measures to decrease potential effects on sensitive habitats.

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As an approach to assess the risk of environmental impacts by air pollutants, the United Nations Economic Commission for Europe (UNECE) has developed critical thresholds of pollutant concentrations and deposition fluxes: Critical Levels (CLEs) and Critical Loads (CLs). A CLE is a pollutant concentration in the atmosphere above which plants or ecosystems may be directly negatively affected (Posthumus, 1988). Recently, long term CLEs of NH_3 were reviewed and new, lower values proposed and adopted by the UNECE (Cape et al., 2009a; Sutton et al., 2009a; UNECE, 2007): 1 μg NH_3 m^{-3} for the most sensitive ecosystems, i.e. where lichens and bryophytes are part of the ecosystem integrity, and 3 ± 1 μg NH_3 m^{-3} for higher plants in other semi-natural ecosystems. A CL is a pollutant deposition below which no significant harmful effects on the environment are expected to occur according to current knowledge (Posthumus, 1988). Nitrogen (N) CLs have been defined for specific ecosystem types (see UNECE, 2010 for most up-to-date CLs). In contrast to the CLE approach, which is specifically defined for gases such as NH_3, the CL integrates all forms of reactive N and therefore requires estimates of total N deposition. According to Sutton et al. (2009b) these N deposition estimates are inherently more uncertain, and for assessing the environmental risk imposed by NH_3, it is much easier to measure NH_3 concentrations and examine exceedance of the CLE than to verify CL exceedances by measurement. However, until the recent revision of CLEs, exceedance of CLs has been more commonly used for impact assessments of atmospheric N. For atmospheric NH_3, this may reflect that previous long-term NH_3 CLEs were set at much less precautious level than associated values of N CLs (e.g. Burkhardt et al., 1998), which was one reason for the revision of new long term NH_3 CLEs (Sutton et al., 2009b).

For assessing the environmental impact of NH_3 concentrations and deposition by modelling, it is essential to estimate NH_3 emissions accurately (Dragosits et al., 2002; Hellsten et al., 2008). Hallsworth et al. (2010) highlighted the problem of modelling NH_3 dispersion at relatively coarse scales, such as 5 km resolution, due to the high spatial variability of NH_3 emissions and showed that 5 km modelling underestimated the impact of NH_3 concentrations on semi-natural areas close to intensive agricultural areas. However, at UK national scale, standard assessments of the impact of N deposition are based on 5 km resolution modelling (Dore et al., 2007; Matejko et al., 2009). Dore et al. (2012) compared CL exceedances in the UK using data at 1 km and 5 km resolution. In contrast to results of Hallsworth et al. (2010) for CLEs, CL exceedances were not highly sensitive to grid resolution. This was attributed to the contribution of N wet deposition (which shows less local variability than dry deposition) and that all ecosystem types were included (not only nature reserves protected under the Habitat Directive, as in Hallsworth et al., 2010). However, for an individual nature reserve located a few kilometres from a major road, the standard 5 km grid data were inadequate to accurately assess local N deposition (Dore et al., 2012).

This study provides a contribution to the landscape scale analysis conducted across Europe within the NitroEurope Integrated Project (NEU) (Sutton et al., 2007), in which a landscape is defined as a spatially heterogeneous area covering several square kilometres and contains interacting ecosystems (Forman and Godron, 1981). In rural landscapes, anthropogenic processes in the form of farm management determine to a large extent N dynamics and much of its environmental impact within the landscape (Cellier et al., 2011). The NEU landscape analysis aimed to quantify N flows at the landscape scale using measurement and modelling approaches. In this study, we analysed NH_3 dispersion and its environmental impact in a 6 km × 6 km rural landscape in southern Scotland. The landscape has a diverse emission pattern with a large number of NH_3 emission hotspots, and large areas of sensitive ecosystems as potential sinks. A detailed landscape inventory of all farms and fields at field-level resolution was conducted to coincide with an intensive spatial monitoring programme of NH_3 concentrations. Ammonia dispersion and deposition was modelled at a 25 m resolution, and the environmental impact of the local NH_3 sources assessed and compared with national 1 km resolution estimates (Hallsworth et al., 2010). The results have general implications for the sustainable management of landscapes that combine both intensive livestock agriculture and ecosystems of relevance for environmental protection.

2. Site and methods

2.1. Study area

The study landscape is situated in southern Scotland, which has a temperate climate, with an annual mean temperature of ~8 °C, a typical rainfall of ~1000 mm and predominant southwesterly winds. The 6 km × 6 km area (Fig. 1) is dominated by agricultural grassland (48%), followed by moorland (21%), rough grass (13%) and woodland (10%). The moorland area with low NH_3 emissions is in the northwestern part of the landscape and is partially grazed by sheep at very low stocking densities, partly legally protected as a Site of Special Scientific Interest (SSSI), with another part undergoing peat cutting. The southeastern part is dominated by agricultural land, such as sheep and beef cattle pastures interspersed with poultry farming houses containing nearly 1.5 million laying hens. Most of the layers are kept in cage systems with manure removal by belt systems two to three times per week (farm locations circled in black in Fig. 1, with other livestock houses shaded in black). However, a number of the houses have deep-pit systems, and in most of them layers are managed as free range birds.

2.2. Landscape inventory and emissions

Detailed land cover/land use and farm activity data were obtained by a local survey carried out by Scotland's Rural College (SRUC) and the Centre for Ecology & Hydrology (CEH). Management activities were recorded for each farm building and agricultural field throughout 2008, including type and numbers of livestock housed and grazed, manure management, ventilation type and emission height, crop type and the application of mineral and organic fertiliser. Land cover/land use and farm activity data were processed with a Geographical Information System (ESRI, ArcGIS) and emissions calculated for each individual field and livestock house. Field emissions were calculated by applying UK average emission factors (EFs) of the UK NH_3 emission inventory to applications of mineral and organic fertiliser and to excreta of grazing livestock (Misselbrook et al., 2009), using fertiliser application rates provided by the farmers. Typical N contents were applied to the different types of organic fertiliser (Defra, 2010). Grazing excreta were calculated using grazing records and daily N excretion rates of the specific type of animal (Misselbrook et al., 2009). Similarly, average NH_3 EFs of the UK emission inventory were applied to calculate housing and manure storage emissions, with housing records on systems and durations provided by the farmers. However, after initial analyses, housing EFs were partly adjusted to account for specific local management practices (see Section 3.3). All spatial datasets were converted to a 25 m grid resolution for atmospheric dispersion modelling (see Section 2.4).

2.3. Spatial NH_3 concentration measurements

Monthly average concentrations were measured from April 2007 to December 2008 at 31 locations using ALPHA passive diffusion samplers (Tang et al., 2001) at a sampling height of 1.5 m above ground. Measurement locations were distributed across the study area in collaboration with farmers and landowners in the landscape. Sites were selected to cover NH_3 concentrations over different land cover types and farms. More sites were placed in NH_3 emitting areas to capture concentration gradients around emission hotspots and diffuse sources, taking the main wind direction into account. The nearest site to an emission hotspot was located 70 m downwind of a poultry house to avoid saturation of the samplers. To assess measurement precision and uncertainty, samplers were exposed in triplicate at each location. The sampling rate of the ALPHA samplers was calibrated against the DELTA denuder reference system (Sutton et al., 2001b), using the UK National Ammonia Monitoring Network methodology (NAMN, Sutton et al., 2001a). ALPHA samplers were stored in a cold room (4 °C) until analysis in the laboratory with an AMFIA NH_3 flow injection analyser, based on analysis by selective ion membrane transfer and subsequent conductivity measurement (Wyers et al., 1993).

2.4. Atmospheric dispersion and deposition modelling

Atmospheric dispersion and dry deposition of NH_3 within the study landscape was simulated using the LADD (Local Area Dispersion and Deposition) model (Hill, 1998). Loubet et al. (2009) recently reviewed LADD and other models available for simulating NH_3 dispersion. The advantages of LADD are that it operates in 3D (with