Agroforestry systems for ammonia air quality management

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Declaration

The candidate confirms that the work submitted is his own, except where work which has formed part of jointly-authored publications has been included. The thesis contains two chapters in press with another two intended for publication in peer-reviewed journals. Details of each proposed publication, including an outline of the candidate and co-authors contributions, are given below. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others. No part of this work has been submitted for any other degree or professional qualification.

The candidate, as lead author, carried out the data analysis and writing of the papers and final thesis. Co-authors provided support and guidance on the scope and design of the project, the analyses performed and contributed to the editing of paper manuscripts. More specifically:

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Lay Summary

Air pollution can lead to environmental impacts. Over the past decades there have been some success stories reducing pollutant emission, namely sulphur dioxide (SO_2). However, impacts on ecosystems from atmospheric nitrogen (N) pollution (referred to as N deposition) are still seen as a major threat for European biodiversity. The fertiliser effect of nitrogen can disturb the system of ecosystems leading to changes in species composition and changes in structure and function. Across Europe over 70% of Natura 2000 protected sites are at risk from N deposition with over 70% of the Natura 2000 area in Europe (EU28) exceeding environmental limits (called a critical load) for nitrogen. Ammonia (NH_3) from agricultural sources is a key nitrogen pollutant contributing to the threat to these sites due to the close proximity of agricultural activities and protected sites.

Modelling using an atmospheric transport model showed that agricultural livestock production in the UK is the dominant nitrogen source for N deposition across the UK Natura 2000 network. Nearly 90% of all sites had livestock as their dominant source. 76% of all Special Areas of Conservation (SAC) sites exceeded their critical load for N deposition, representing 74% of the entire SAC area.

Legislation to regulate pollutant emissions to air and protect biodiversity are often not integrated, and there has been no common European approach for determining the impacts of N deposition on individual Natura 2000 sites. Sources of N deposition present difficulties in assessing and attributing impacts, because N deposition can result from local sources (1-2 km), or very far away sources (>1000 kms).

Managing nitrogen losses on the farm and improving the efficient use of nitrogen are key components for overall reduction in ammonia emissions. Many nitrogen management options are available to abate ammonia from agricultural activities. On the one hand, technical and management measures include controlling emissions from manure storage and spreading, livestock feeding strategies, and improving housing systems. Trees, on the other hand, are effective scavengers of both gaseous and particulate pollutants from the atmosphere, making tree belts potentially effective landscape features to support ammonia abatement strategies.

Using a deposition and turbulence model the recapture efficiency of tree planting around ammonia sources was estimated. Using different tree canopy structures, tree depths and differing leafiness of the canopy, recapture efficiency for ammonia by the trees ranged from 27% (trees planted around housing systems) up to 60% (under-storey livestock silvopastoral systems). Model results from scaling up to national level suggest that tree planting in hot spot areas of ammonia emissions would lead to reduced N deposition on nearby sensitive habitats. Scenarios mitigating emissions from cattle and pig housing yielded the highest reductions. Increased capture by the planted trees also had the effect of reducing long-range transport effects, including a decrease in nitrogen deposition in rainfall of up and a decrease in export from the UK.

Agroforestry measures for ammonia abatement were shown to be cost-effective for both planting downwind of housing and in silvopastoral systems, when costs to society were taken into account. Comparing the cost per kg of NH_3 abated showed that planting trees is a method of ammonia emission mitigation comparable with other (technical) measures.

Agroforestry for ammonia abatement offers multiple benefits for the famer and synergistic effects for society as a whole including i) carbon sequestration, ii) visibility screening around housing units, iii) improved animal welfare for silvopastoral systems, iv) reducing critical load exceedance on protected sites v) price advantage of 'woodland chicken' products, vi) supporting pollution regulation requirements for emission reduction, vii) supporting national afforestation policies.

The results of this work support the notion that in the emerging discussion about the values of ecosystem services and the role of nature-based solutions to tackle persistent environmental challenges, tree planting has a large potential in rural and urban environments.

Abstract

Air pollution can lead to environmental impacts. Over the past decades there have been some success stories reducing pollutant emission, namely sulphur dioxide (SO_2). However, impacts on ecosystems from atmospheric nitrogen (N) pollution are still seen as a major threat for European biodiversity. Across Europe over 70% of Natura 2000 sites are at risk from eutrophication with over 70% of the Natura 2000 area in Europe (EU28) exceeding critical loads for nutrient nitrogen deposition. Agricultural ammonia is a key contributor to the threat to these sites due to the close proximity of agricultural activities and protected sites.

Source attribution modelling using an atmospheric transport model showed that agricultural livestock production in the UK is the dominant nitrogen source for N deposition across the UK Natura 2000 network. Nearly 90% of all sites had livestock as their dominant source, contributing 32% of the total nitrogen deposition across the whole network. 76% of all Special Areas of Conservation (SAC) sites exceeded their critical load for nutrient nitrogen, representing 74% of the entire SAC area. The extent of exceedance is also notable with many sites experiencing depositions of >50 kg N/ha/yr over the critical load. The situation for acidity critical load exceedance is less severe, but 51% of sites are still exceeded.

Legislation to regulate pollutant emissions to air and protect biodiversity are often not integrated, and there has been no common European approach for determining the impacts of nitrogen deposition on individual Natura sites, or on conservation status. Off-site sources of air pollution present difficulties in assessing and attributing impacts, because deposition can result from local sources (1-2 km), or very far away sources (>1000 kms).

Managing nitrogen losses on the farm and improving the efficient use of nitrogen are key components for overall reduction in NH_3 emissions. Many nitrogen management options are available to abate ammonia from agricultural activities. On the one hand, technical and management measures include controlling emissions from manure storage and spreading, livestock feeding strategies, and improving housing systems. Trees, on the other hand, are effective scavengers of both gaseous and particulate pollutants from the atmosphere, making tree belts potentially effective landscape features to support ammonia abatement strategies. Using a coupled deposition and turbulence model the recapture efficiency of tree planting around ammonia sources was estimated. Using different canopy structure scenarios, tree depths and differing leaf area density (LAD) and leaf area index (LAI) were adjusted for a main canopy and a backstop canopy. Recapture efficiency for ammonia ranged from 27% (trees planted around housing systems), up to 60% (under-storey livestock silvopastoral systems). Practical recapture potential was set at 20% and 40% for housing and silvopastoral systems respectively. Model results from scaling up to national level suggest that tree planting in hot spot areas of ammonia emissions would lead to reduced N deposition on nearby sensitive habitats. Scenarios for on-farm emission control through tree planting showed national reductions in nitrogen deposition to semi-natural areas of 0.14% (0.2 kt N–NH_x) to 2.2% (3.15 kt N–NH_x). Scenarios mitigating emissions from cattle and pig housing yielded the highest reductions. The afforestation strategy showed national-scale emission reductions of 6% (8.4 kt N–NH_x) to 11% (15.7 kt N–NH_x) for 25% and 50% afforestation scenarios respectively. Increased capture by the planted trees also generated an added benefit of reducing long-range transport effects, including a decrease in wet deposition of up to 3.7 kt N–NH_x (4.6%) and a decrease in export from the UK of up to 8.3 kt N–NH_x (6.8%).

Agroforestry measures for ammonia abatement were shown to be cost-effective for both planting downwind of housing and in silvopastoral systems, when costs to society were taken into account. Planting trees was also cost-effective from a climate change perspective. Comparing the cost per kg of NH_3 abated showed that planting trees is a method of ammonia emission mitigation comparable with other (technical) measures. The costs for planting trees downwind of housing were calculated at €0.2-0.8/kg NH_3 abated, while the cost of the silvopastoral system were €2.6-7.3/kg NH_3.

Agroforestry for ammonia abatement offers multiple benefits for the famer and synergistic effects for society as a whole including i) carbon sequestration, ii) visibility screening around housing units, iii) improved animal welfare for silvopastoral systems, iv) reducing critical load exceedance on protected sites v) price advantage of 'woodland chicken' products, vi) supporting the Industrial Emission Directive (IED) requirements for emission reduction, vii) supporting national afforestation policies. The results of this work support the notion that in the emerging discussion about the values of ecosystem services and the role of nature-based solutions to tackle persistent environmental challenges, tree planting has a large potential in rural and urban environments.

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Chapter 1. Introduction

1.1 Overview

Air pollution can lead to many forms of environmental impacts on terrestrial and aquatic ecosystems including the effects of acidification, eutrophication, and the direct toxic effects on plants. While there have been some success stories with emission reductions of sulphur dioxide (SO_2) over the past decades, emissions of nitrogen pollution have been reduced much less with ammonia emissions reduced by only small amount. Many protected sites are still at threat from air pollutants across the EU with nitrogen deposition, and particular ammonia, still remaining the significant pollutant threat.

There are many nitrogen management options available in order to abate ammonia from agricultural sources. These include manure management, livestock feeding strategies, and improving housing systems. This research will investigate a further option, namely the potential for trees to capture ammonia emissions at source. Trees are effective scavengers of both gaseous and particulate pollutants from the atmosphere making tree belts potentially effective landscape features to support ammonia abatement strategies.

The thesis initially examines the extent of critical load exceedance over the Natura 2000 network, and establishes the main sources that contribute to nitrogen deposition on protected sites. It will then quantify the effect of trees around hot-spots of ammonia by modelling farm emissions at the local scale. The study will look at tree belt planting based on two options - planting tree belts downwind of animal housing and planting trees for livestock managed under the trees (silvo-pastoral systems). Further national modelling will be undertaken to upscale the effect of tree belts across the UK. Finally cost benefit analysis will be carried out to estimate the cost effectiveness of planting trees to abate ammonia taking into account any social benefits. This abatement method will also be compared with other ammonia mitigation methods.

In summary this thesis will demonstrate that ammonia is a target pollutant for policy makers if sensitive ecosystems are to be protected. It will show that planting tree belts around hot spots of farm emissions is a suitable and alternative method in the effort to reduce nitrogen deposition to sensitive ecosystems.

The introduction chapter will initially describe the main sources of emissions of nitrogen and will outline the processes of deposition and the fate of nitrogen in the environment focusing on the nitrogen cascade approach. Some effects and ecosystem impacts are also given together with a review of the policy responses (and some examples) for tackling nitrogen pollution impacts. The final half of the introduction focuses on the mitigation techniques that are available to the farming industry and then describes in detail the processes and practice of implementing tree planting for ammonia mitigation. The modelling approach rounds off this chapter describing the main model used in the study.

1.2 Background

The increasing relationship between science and legislation has already yielded positive environmental benefits. For example, many ecological effects of 'Acid Rain' have been substantially improved upon by a reduction in the emissions of sulphur pollutants (SO_x) from coal-fired power stations. This has led to the subsequent reduction in pollutant concentrations in air, rain, freshwaters and soil, and there is now evidence that ecosystems are in the process of recovery (Caporn, 2006). However, more recent transboundary air pollution problems have been identified including the eutrophication process that results from nutrient nitrogen deposition. Atmospheric N pollutants contribute to a host of environmental problems including human health effects through particulate matter, greenhouse gas emissions, and eutrophication and acidification effects on semi-natural ecosystems that can lead to species composition changes (Pitcairn, 1998; Sheppard, 2008). Many protected nature sites (e.g. Natura 2000) across the UK and the rest of Europe experience critical load and level exceedances with agricultural emissions of ammonia being recognised as a major source.

Remedies to reduce nitrogen have until now largely been limited to national and international policies like the National Emission Ceilings Directive in Europe (NECD - 2001/81/EC) and the Convention on Long-Range Transboundary Air Pollution (UNECE, 1999). This has been delivered by the setting of national emission ceiling targets and the setting of effects-based policies e.g. the setting of an environmental limit or threshold value to protect human health or ecosystems. Most successes have come from the regulation of combustion sources for nitrogen oxides (NO_x) through directives like the Large Combustion Plant Directive, and the implementation of catalytic converters and engine efficiency improvements in vehicles. Delivering similar successes in NH_3 emission reduction in the agricultural sector has been much harder to attain in most countries due to the lack of in-country regulations with the exception of Denmark and the Netherlands. This has been partly down to the burden of individual farmers having to bear the costs of NH_3 reduction measures like building low emission housing systems, manure storage systems and manure application, with little regulatory requirement.

As SO_2 and NO_x emissions have decreased NH_3 has begun to contribute a larger share of pollution. In fact by 2020, it is estimated that ammonia will be the largest single contributor to the nutrient nitrogen and acid deposition, and secondary particulate matter formation in Europe (Reis *et al.*, 2015).

1.3 Atmospheric Nitrogen Pollution – Emissions, Pollutant Processes, Impacts and Policy Responses

1.3.1 Nitrogen Emissions

Emissions of nitrogen are primarily derived from combustion processes and agricultural sources. Emissions of NO_x comes mainly from energy generating sectors, where in the UK there has been a steady downward trend since peaks in the 1970s (Figure 1.1). In the power generation sector reductions in NO_x have followed the similar downward trend as sulphur with less use of coal and the introduction of gas generation leading to reduced NO_x emissions (Defra, 2012). Around 50% of UK NO_x is emitted from transport sources and similar reductions have been achieved through engine efficiency and 'end-of-pipe' improvements (three-way catalyst). The EU has a programme to update legislation setting new emission targets for new car and van fleets. For example the latest regulation (Regulation (EC) 459/2012) sets out limits (known as Euro 6) to reduce NO_x emissions from light passenger cars (60 mg/km) and commercial vehicles (75 mg/km).

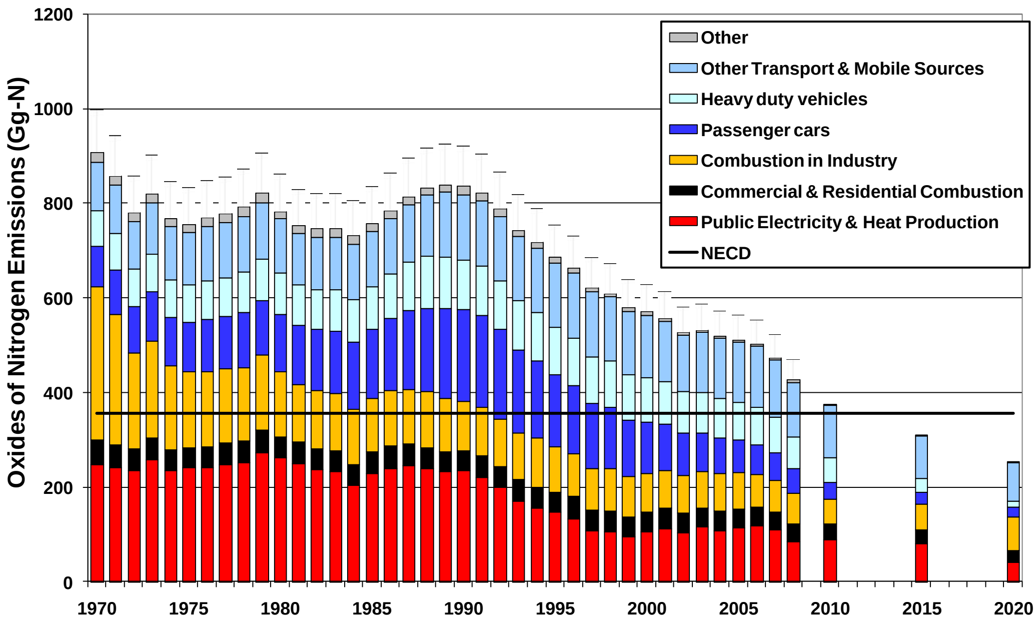


Figure 1.1. UK emissions of Nitrogen Oxide (NO_x-N Gg-N) (Defra, 2012)

Air emissions of ammonia (NH_3) across Europe are dominated by the agricultural sector (Sutton *et al.* 1995 ; Misselbrook et al 2000). This reduced form of nitrogen is highly reactive and deposits readily to vegetation (Sutton *et al.*, 1993; Erisman *et al.*, 2007).

The breakdown of emissions of NH_3 by sector (Figure 1.2) shows that livestock farming provides around 75% of the total NH_3 emission in the UK (Defra, 2006).

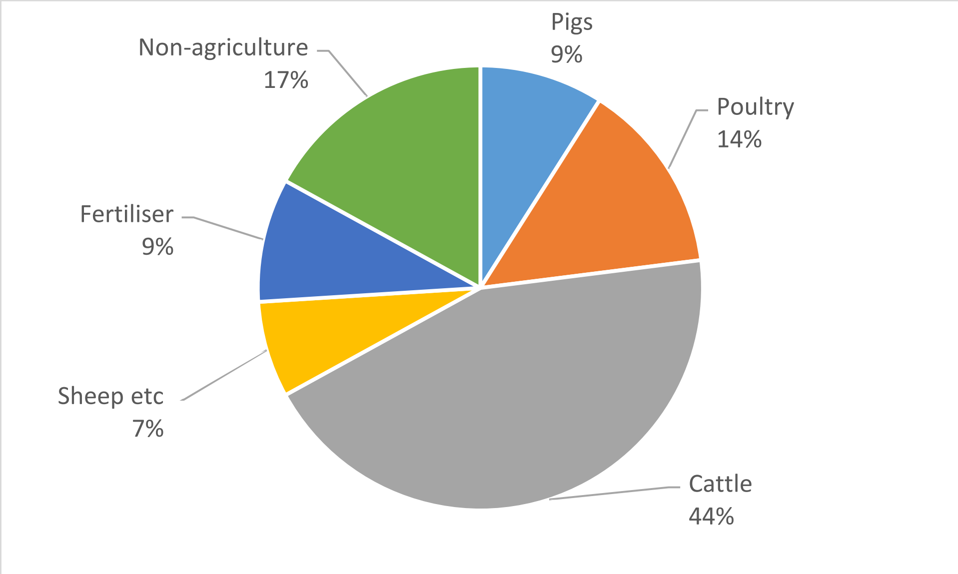


Figure 1.2. Ammonia emissions in the UK by source (Defra, 2002)

Ammonia emissions in the UK have reduced since the 1990s due to declining livestock numbers, as well as improved technologies in the housing of pig and poultry farms (e.g. wet scrubbers), which are now regulated under the Industrial Emissions Directive (IED) 2010/75/EU ; Dore et al, 2008). Figure 1.3 shows the estimated emission trends in NH_3 from 1990 to 2010, with additional projections for 2015 and 2020. The UK is a signatory to the Gothenburg Protocol and the EU National Emission Ceilings Directive (NECD) with targets for NH_3 emissions set to 297 Gg of NH_3 (244 Gg-N) for both by 2020. However, while emissions for ammonia are on target for 2020 (NECD), the rate of reduction is slow into the future (Misselbrook et al 2009).

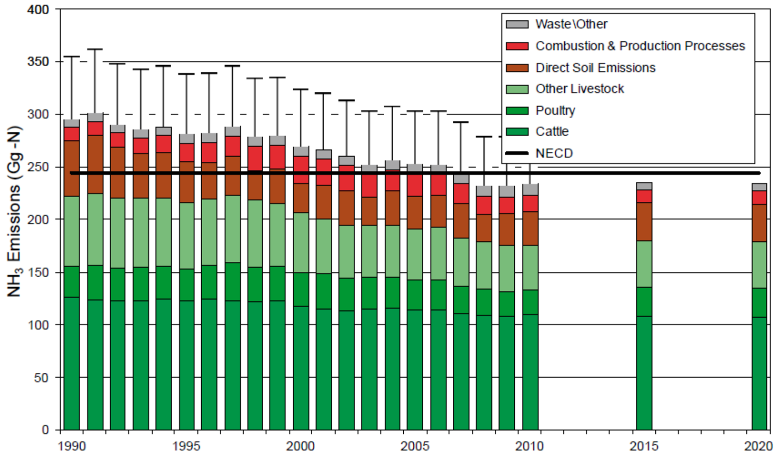


Figure 1.3. NH_3 emission in the UK 1990-2020 (Defra, 2012)

There are six agricultural management activities identified as key sources of ammonia: emissions from housing, grazing, storage and manure spreading, hard standings and fertiliser use (Misselbrook *et al.*, 2011). A recent UK inventory (2011) provides a breakdown of these activities for the UK (Figure 1.4) showing housing and manure application as the highest emission activities.

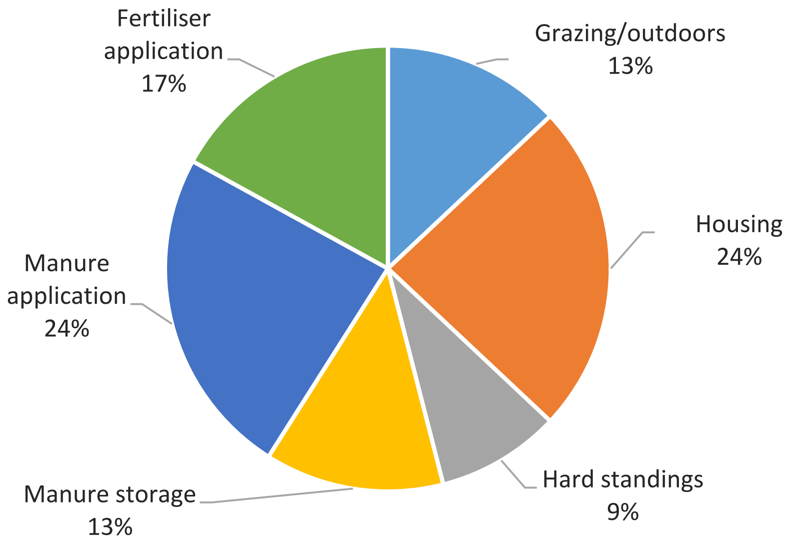


Figure 1.4. UK breakdown of ammonia emission by management category (Misselbrook *et al.*, 2011)

Emissions from livestock in the UK are calculated using a nitrogen flow model (Webb and Misselbrook, 2004) using emission factors and empirical models for each management category. These are then worked up based on animal statistics for the country to provide national emissions for each management category.

Nitrogen contained in the diet of livestock animals is either retained to produce meat or diary or lost (excreted) in the form faeces and urine from the animals. The urine (or uric acid in poultry) form of N is then hydrolysed into NH_3 by the enzyme urease, while the organic manure N is mineralised to become part of the total amonniacal N (TAN) pool which is a combination of NH_3 and NH_4^+ (Eory *et al.*, 2015). N is then volatilized from the manure/urine surface in the form of NH_3. This loss of TAN from the animals can range from 60-90% depending on animal species, feed composition and management (van Vuuren *et al.*, 2015).

1.3.2 Nitrogen Processes and Cascade

Once released into the atmosphere reactive nitrogen N_r (defined as all nitrogen compounds except for N_2), has the ability to be transformed into other forms of reactive nitrogen giving rise to a notion of pollutants 'cascading' through the environment bringing about issues such as eutrophication, acidification, greenhouse gas production, and poor air quality. The conceptual framework of a "nitrogen cascade" (Figure 1.5), was originally described by Galloway *et al.* in 1998. Forms of reactive nitrogen include ammonia (NH_3), and ammonium (NH_4^+), nitrous oxide (N_2O), nitrate (NO_3^-), nitrite (NO_2^-), and organic compounds (e.g., urea, amines, proteins, and nucleic acids). They key processes for the production of ammonium in the atmosphere are:

One-way reaction:

$$NH_3 + H_2SO_4 \ in \ particle \ \rightarrow NH_4^+ \ in \ particle$$

[*No NH_3 bound in this way can volatilise*]

Two way reaction:

$$NH_3 + HNO_3 \ (gas) \ \leftrightarrow NH_4NO_3 \ containing \ particle$$

[NH_3 can volatilise again (depending on temperature, realtive humidity, concentrations)]

These forms of N are all interconnected and are constantly in flux in the environment. N_r accumulates in the environment at all spatial scales (e.g. local, regional and global) (Galloway *et al.* 1995). The anthropogenic creation of N_r has increased sharply since the 1960s and is now greater than production from natural systems. The main causes of this increase have been down to the Haber-Bosch process which converts N_2 to NH_3 for food production; combustion of fossil fuels producing NO_x; and the cultivation of legumes, rice and other crops to promote the conversion of N_2 to organic N though biological fixation. The illustration below shows the movement of human-produced N_r as it cycles through various environmental reservoirs in the atmosphere, terrestrial ecosystems, and aquatic ecosystems before it returns to the atmosphere as non-reactive N_2 following denitrification. The nitrogen cascade illustrates the multiple effects N_r has on the environment from human health impacts to terrestrial eutrophication and soil acidification. The rates of nitrogen cascade vary within environmental systems with some systems slowing the cascade over time resulting in an accumulation of Nr. This Nr accumulation can enhance the effects of Nr on the environment (Galloway, 2003).

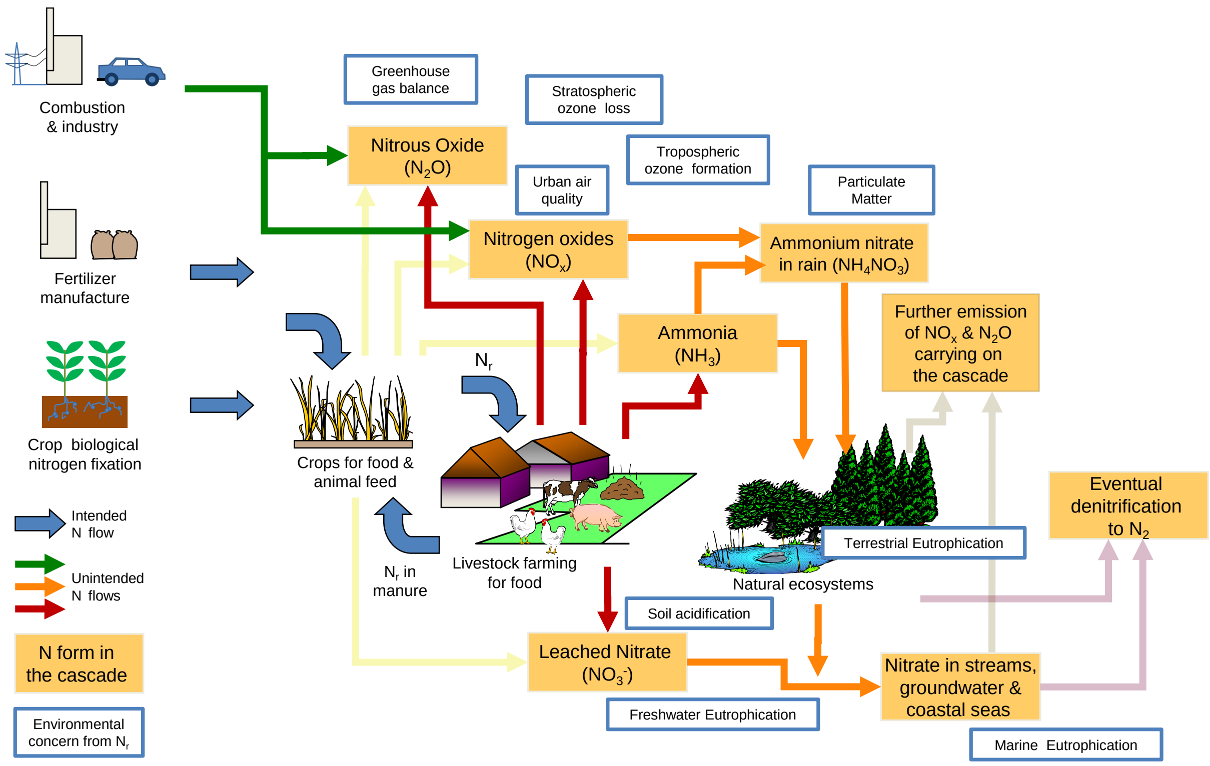


Figure 1.5. The Nitrogen Cascade (adapted from The European Nitrogen Assessment, 2011) highlighting combustion and agricultural emissions. The orange boxes represent the main pollutant forms of N_r. Five environmental concerns are highlighted as blue boxes. Blue arrows represent intended anthropogenic N_r flows while all the other arrows (green, orange, red) are unintended flows (or losses). The purple flow is the eventual conversion of N_r back to N_2 through dinitrifcation.

The coloured arrows (green, yellow, red etc.) of Figure 1.5 show the losses or unintended flows of N_r molecules as they are transferred from one environmental system to the next. The subsequent environmental impacts (blue boxes) result from the multiplicity in the pollutant form of N_r as it cascades through the system. Combustion sector processes (green arrows) show the emissions of NO_x and subsequent flows to other N_r forms and associated impacts e.g. air quality and terrestrial eutrophication. The emissions of ammonia from agriculture show a whole host of N_r losses in various pathways giving rise to multiple effects including urban air quality, greenhouse gas balance, particulate matter, soil acidification and terrestrial eutrophication.

Atmospheric pollutant deposition occurs through the processes of wet and dry deposition (Figure 1.6). In the atmosphere NH_3 can undergo chemical reactions to form ammonium aerosol (NH_4^+), which can be transported over long distances (1000s km) eventually being removed by wet deposition in precipitation (rain or snow) (Fowler et al 1998; EMEP, 2007). Since cloud and rainfall scavenge these aerosols, parts of the UK with the highest rainfall tend to have the largest wet deposition (Rodhe & Grandell, 1972; Smith, 1975). Dry deposition of NH_3 is the removal of gases and aerosol phase (e.g. $NH_4^+NO_3^-$) directly to the vegetative surfaces (or any surface) (Smith *et al*, 2000). Dry deposition of NH_3 occurs over much shorter distance (<1-2 km).

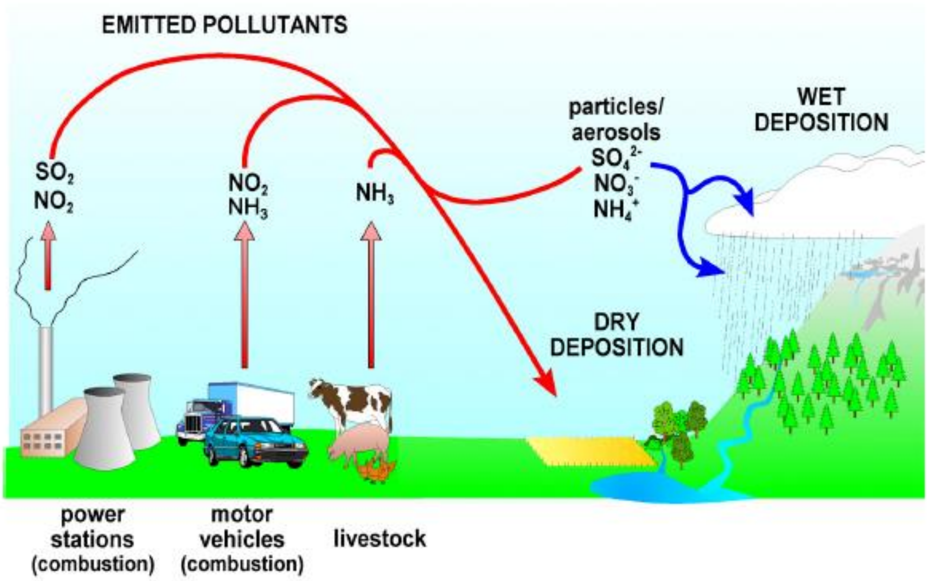


Figure 1.6. Deposition processes of emitted pollutants by UK industry. (adapted from NEGTAP, 2001)

NO_2 gas can also be dry deposited to the vegetation surfaces, but the gas reacts in the atmosphere to form nitric acid (HNO_3) (Hertel, 2006), which in turn can react to form nitrate (NO_3^-) aerosols with NH_3 or dry deposit as HNO_3. Long range transport of pollutants are not seen as a local landscape issue, but national governments have to take account of longer range transport of pollutants and their impacts at a national and international level.

1.3.3 Nitrogen Effects and Ecosystem Impacts

Deposition of nitrogen to ecosystems can lead to eutrophication and acidification effects (Falkengren-Grerup, 1986). Observed changes in ecosystems can range from acidification of soils to the increase in nitrogen loving plants at the expense of plants tolerant of low nitrogen supply (van Breemen & van Dijk, 1988). These changes are not always apparent as species composition changes can occur slowly over a number of decades. Observed environmental effects include:

- Loss of species adapted to low N availability e.g. many slower-growing lower plants, notably lichens and bryophytes. (Pearce 2002; Bobbink *et al.*, 1998)
- Changes in species composition with the loss of high conservation value species, which can also impact on ecosystem function. Bogs are particularly at risk if they lose Sphagnum mosses. (Press *et al.*, 1986; Gunnarsson 2000)
- Competition from invasive species - often grasses pose a threat for many communities. (Tomassen 2004)
- Pollution of ground water and drinking water due to nitrate leaching (Zhang 1996)
- Losses of both inorganic and organic N from terrestrial systems may contribute to freshwater, coastal and marine eutrophication (Hornung *et al.*, 1995).

Many sensitive semi-natural areas across Europe are often above their nitrogen threshold. Critical loads and levels are the terms used to describe an environmental limit or threshold above which some form of degradation is likely. The critical load relates to the quantity of pollutant deposited from air to the ground (flux), whereas the critical level is the gaseous concentration of a pollutant in the air (UNECE, 1996). A simple description of a critical level can be illustrated by way of example: namely, a critical level of ammonia for lichens and bryophytes is 1 μg/m3. This threshold figure provides regulators, planners and conservation practitioners with a clear understanding of the maximum concentration of ammonia that the lichen can withstand. The Critical Load for a habitat is generally defined as: "*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). Critical Levels are described as a threshold limit <u>above which</u> impacts to ecosystems could occur, explicitly as: *"and is defined as the "concentration, cumulative exposure or cumulative stomatal flux of atmospheric pollutants* *above which direct adverse effects on sensitive vegetation may occur according to present knowledge"* (Posthumus, 1988).

Critical Loads for nutrient nitrogen include an empirical approach where a range of deposition loads are set for particular ecosystems across Europe (Achermann & Bobbink 2003). For example, a nitrogen poor habitat like a raised blanket bog has a critical load threshold of 5-10 kg N/ha/yr while the critical load for a relatively nutrient rich habitat e.g. a rich fen, has a higher critical load threshold of 15-30 kg N/ha/yr. A sample list of the habitat specific Empirical Critical Loads for Nitrogen are shown in Table 1.1.

Table 1.1. Empirical critical loads of nutrient nitrogen (kg N ha^{-1} yr^{-1}) for some typical UK habitats (Achermann & Bobbink, 2003)

Ecosystem type	kg N ha^{-1} yr^{-1}	Indication of exceedance
Broadleaved deciduous woodland	10-20	Changes in soil processes, nutrient imbalance, altered composition mycorrhiza and ground vegetation
Dry heaths	10-20	Changed species composition, increase of nitrophytic species, increased sensitivity to parasites
Calluna-dominated wet heath (upland moorland)	10-20	Decreased heather dominance, decline in lichens and mosses
Rich fens	15-30	Increase in tall graminoids, decrease in bryophytes
Moss and lichen dominated mountain summits	5-10	Effects upon bryophytes or lichens
Raised and blanket bogs	5-10	Change in species composition, N saturation of *Sphagnum*

Deposition of sulphur, as sulphate (SO_4^{2-}), and nitrogen, as nitrate (NO_3^-), ammonium (NH_4^+) and nitric acid (HNO_3^-), can cause acidification and both sulphur and nitrogen compounds must be taken into account when assessing acidification of soils. Critical loads for acidity (comprising of nitrogen and sulphur) are primarily based on the soil characteristics of an ecosystem and methodology ranges from the empirical approach to a steady-state mass balance. The Critical Load Function (Posch, 1997) is used to determine the links between deposition of sulphur and nitrogen and a critical load of acidity. It is defined in three components parts (Min N, Max N, Max S) with the area under the graph represent the critical load (UNECE, 2004).

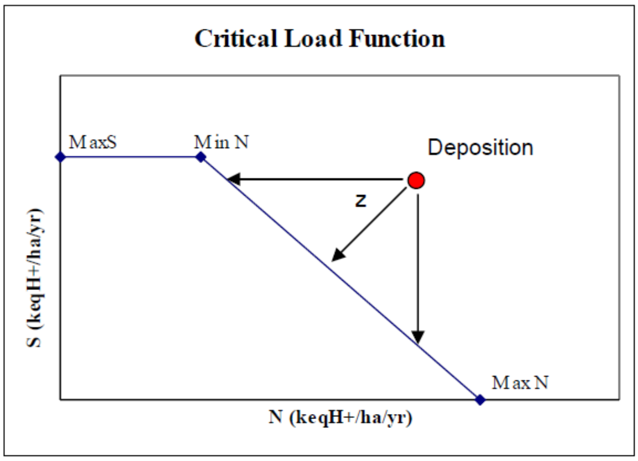


Figure 1.7. Critical Load Function showing an exceedance sue to the deposition of nitrogen and sulphur. z represents the smallest reduction of both pollutants to reach the critical load. (CLRTAP, 2004).

The critical load function provides for a very useful policy tool for assessing any potential remedial action required to meet or better the critical load (i.e. whether S or N deposition or both need to be reduced to avoid exceedance of the critical load). In general, but not always the most practicable method, is to reduce both pollutants by the smallest amount, represented by the shortest distance (z).

In addition to Critical Loads, Critical Levels for certain pollutants can also be set by vegetation types. Critical Levels are set at 30 µg/m^3 for NO_2 for all vegetation types. NH_3 has two values – one for lichens and bryophytes set at 1 µg/m^3 and for higher plants set to 3 µg/m^3.

The critical load approach has been applied for the natural areas of Europe (Hettelingh et al, 2008). Figure 1.8 shows the sensitive areas (in red) to both nutrient nitrogen (left) and acidity (right).

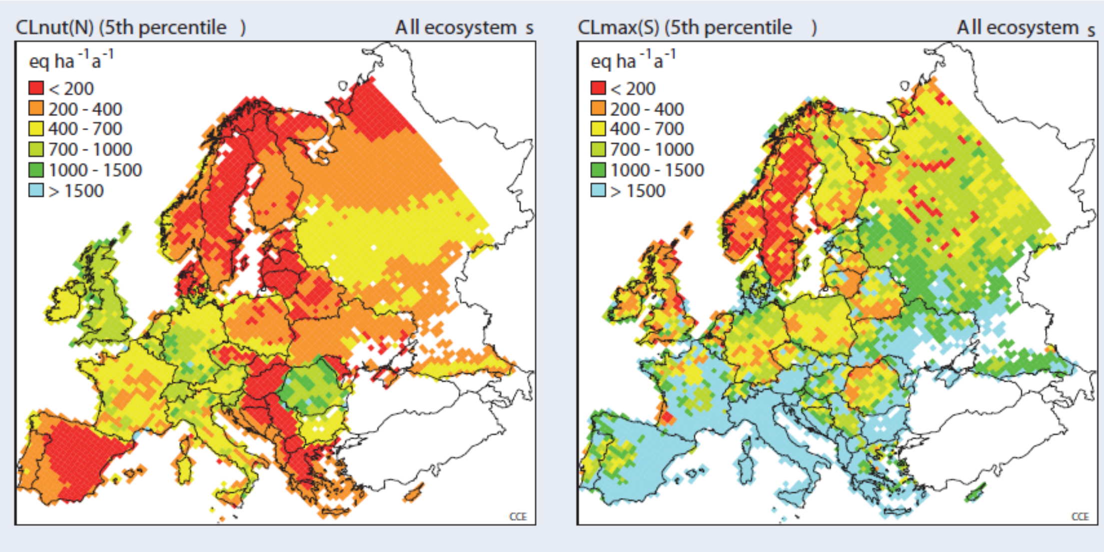
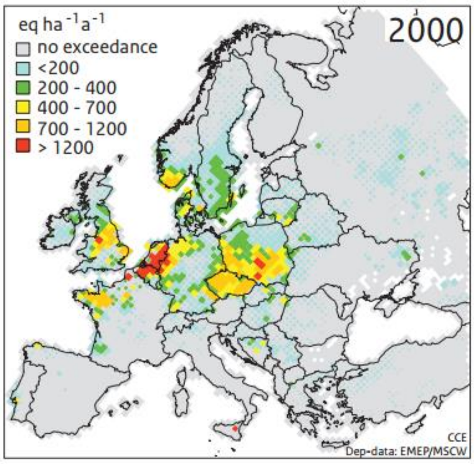


Figure 1.8. European critical loads for eutrophication (left) and acidification (right) which protect 95% of natural areas in 50x50 km^2 EMEP grid. Red shaded areas illustrate grid cells where deposition needs to be lower than 200 eq/ha/yr (equivalent to 2.8 kg N/ha/yr for Nutrient N) to reach this protection target (Hettelingh *et al.*, 2008)

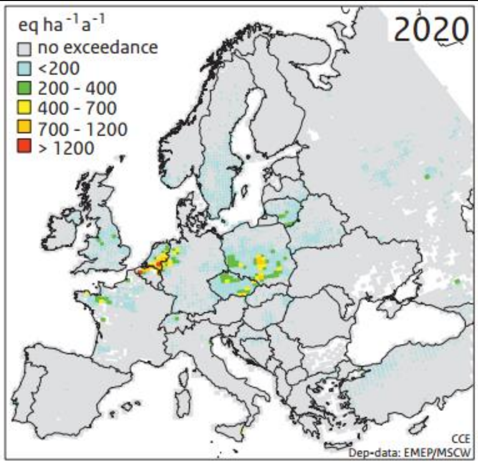
Additional pollutant deposition or concentrations above the Critical Load or Level is termed a Critical Load/Level Exceedance and is used as the main test of a habitat impact in pollution control and regulation. Acidification and nitrogen deposition are significant threats to sensitive semi-natural habitats in the UK and will remain so in 2020 despite significant falls in emissions. It is estimated that in 2020, 39% of sensitive habitats in the UK will still exceed the critical load for acidity, with 48% still exceeding the critical load for nutrient nitrogen (Hall *et al.*, 2006).

Across Europe, mapping of exceedances (Figure 1.9) is carried out individual Member State National Focal Centres and reported to the Coordination Centre for Effects (CCE). The CCE is responsible for the development of modelling methodologies and databases for the integrated assessment of effects of air pollution (under climate change) on biodiversity in European natural areas, including Natura 2000 sites.

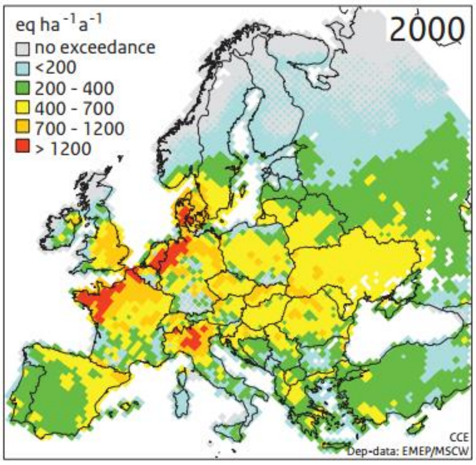
Work carried out by the CCE has shown that the European area at risk of acidification is estimated to decrease from 11% in 2000 to 6% and 1% in 2020 according to current legislation (CLE) and a maximum feasible reduction (MFR) scenario. Some countries including the Netherlands still have 60% of their ecosystem area at risk of acidification in 2020. For ecosystems at risk from eutrophication, this area for Europe is estimated to decrease from 49 % in 2000 to 47 % and 17 % in 2020 for CLE and MFR, respectively.



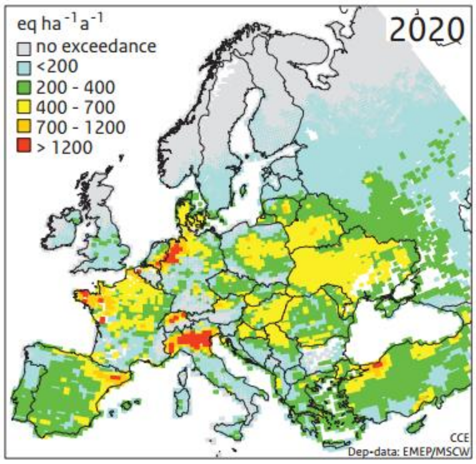
Acidity Critical Load Exceedance 2000.
Red areas showing an exceedance above 1,200 eq/ha/yr.



Acidity Critical Load Exceedance 2020 – Gothenburg Protocol-Current Legislation.



Nutrient Nitrogen Critical Load Exceedance 2000.
Red areas showing an exceedance above 1,200 eq/ha/yr .



Nutrient Nitrogen Critical Load Exceedance 2020 – Gothenburg Protocol-Current Legislation.

Figure 1.9. Critical load exceedance maps for Europe comparing years 2000 and 2020 based on current legislation (Hettelingh et al, 2014).

1.3.4 Policy Responses for Nitrogen Air Pollution

As we have seen, impacts of air pollution to society and the environment can be large. With regard to N pollution a key objective for Governments is to reduce negative human health effects, biodiversity loss and degradation. At the interface between science and policy, an understanding of the ‘acceptable’ limits of pollution can be defined via a framework of cost-benefit analysis, where the incremental benefits of regulation are compared with the incremental costs. Figure 1.10, based on the Drivers, Pressures, State, Impacts, and Responses (DPSIR) framework, illustrates the relationships between science, policy and society. The DPSIR model was defined by the European Environment Agency (EEA) as an extension of the Pressure/State/Responses (PSR) model developed by the Organisation for Economic Co-operation and Development (OECD). The model is a casual framework describing the relationship between society and the environment. It is a cyclical model in that it can be used gauge the effectiveness of responses.

The Drivers are the socio-economic and socio-cultural influences driving human activities, which often increase pressures on the environment. Society’s demand for energy and food provide two examples of drivers giving rise to atmospheric pollutants. The Pressures are the resulting emissions that go on to create depositions and concentrations. The State of the environment is the resulting health and condition of ecosystems and human health (e.g. the pH of the soil or the concentration of a gas in the air). Impacts are the negative effects on human health or environmental degradation. Responses (i.e Policy Responses) are the governments’ responses to tackling not only the impacts, but also the drivers, the state and even the pressures.

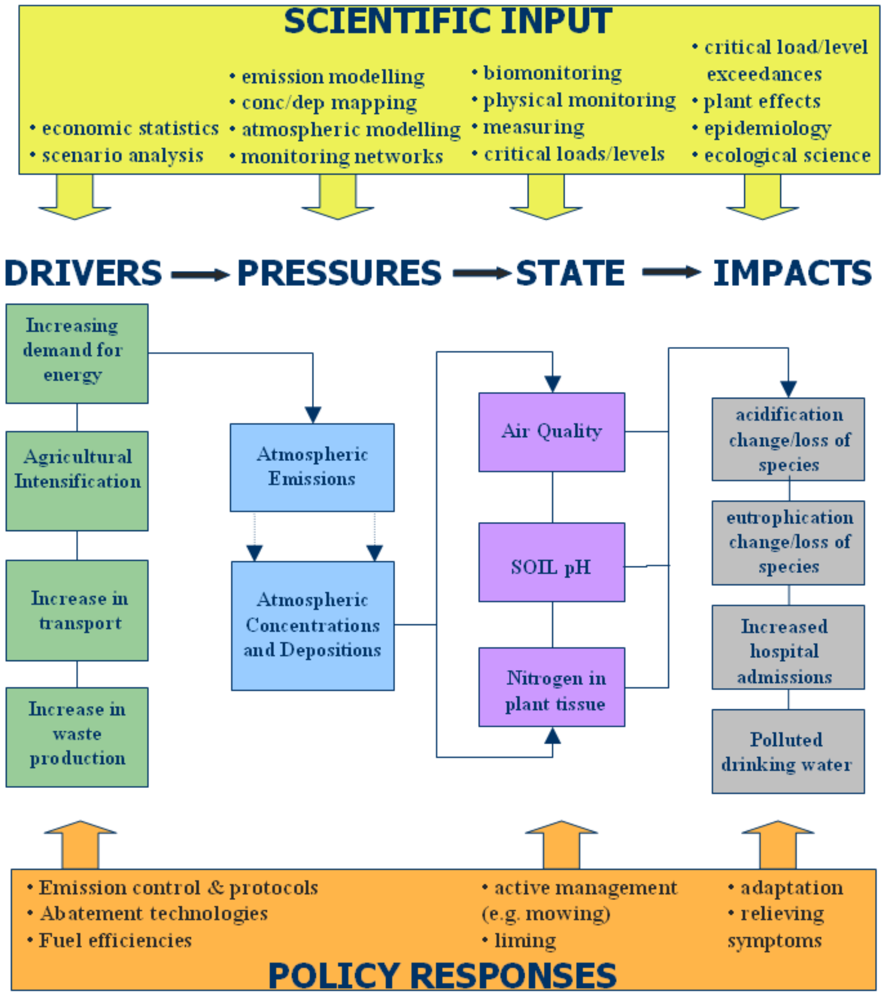


Figure 1.10. Adaptation (Walmsley, 2002) of the DPSIR model showing the interaction of science and policy

The increasing relationship between science and policy has already yielded positive environmental benefits. For example, ecological effects of 'Acid Rain' have been reduced substantially by a reduction in the emissions of sulphur pollutants from coal-fired power stations. This has led to the subsequent reduction in pollutant concentrations in air, rain, freshwaters and soil, and there is now evidence that ecosystems are in the process of recovery. However, more recent transboundary air pollution problems coming to light have been identified including eutrophication from nutrient nitrogen and ground level ozone impacts (Defra, 2012). These issues have yet to be addressed as successfully, with further policy actions to reduce effects being required.

1.3.5 Examples of Policy Responses to Air Pollution

The evidence emerging from scientific research into the pollutants discussed so far have contributed to a body of legislation employed to positively address a variety of concerns including human health impacts and ecosystem degradation. However, pollution control legislation can be traced as far back as the 14th Century where workers were prohibited from using sea-coal (a soft coal) in their furnaces (Mister, 1970). In the 19th century the Improvements Act 1847 contained clauses on factory smoke (Great Britain Parliament, 1847).

This section provides a chronological look at the key legislation that has been adopted by governments in response to environmental pressures. In the UK this legislation is primarily driven by European directives and is the main focus of this section.

The Clean Air Act of the United States, a federal law covering all the states, is the cornerstone for US air quality regulation and first came into law in 1970 (United States Congress, 1970). The US Clean Air Act was amended in 1990 to accommodate large emission sources and important new enforcement powers. Environmental Impact Statements (EIS) in the United States came into law in 1970 under the National Environmental Policy Act (NEPA) of 1969 (United States Congress, 1969). Federal agencies are required to produce EISs for all major projects or legislative proposals that may significantly affect the environment.

In the United Kingdom, the Clean Air Act of 1956 (Great Britain, Parliament, 1956) grew out of a need to control domestic coal burning within cities. In particular, it was domestic coal burning that gave rise to the Great London Smog of 1952 leading to a 50-300% rise in mortality rates (Bell and Davis, 2001). The Clean Air Act of 1968 (Great Britain, Parliament, 1968) introduced the control of industrial sources and air pollution (mainly sulphur dioxide) emissions from tall stacks.

The 1979 Geneva Convention on Long-Range Transboundary Air Pollution (CLRTAP) was established within the framework of the United Nations Economic Commission for

Europe (UNECE, 1979). The convention has 51 parties of which Canada, United States, the Russian Federation, the UK and other countries of the EU are members. The Convention has included eight protocols that identify specific measures to be taken by Parties to cut their emissions of air pollutants including SO_2, NO_x, NH_3, Volatile Organic Compounds (VOCs), O_3, heavy metals, Persistent Organic Pollutants (POPs). The 1999 Multi-pollutants and multi effect protocol (Gothenburg protocol, UN ECE, 1999) set out a 2010 ceiling for emissions of sulphur, NO_x, VOCs and ammonia, which were negotiated on the basis of scientific assessments of pollution effects and abatement options. The Protocol was amended in 2012 to include national emission reduction commitments to be achieved by 2020 and beyond (Reis *et al.*, 2012). Annex IX of Protocol also set out a list of measures that signed Parties are obligated to apply, as a minimum, the ammonia control measures specified in annex. These include measures to reduce emissions from manure storage and spreading, animal housing installations and fertiliser application. The Gothenburg protocol was also used as the foundation for the European National Emissions Ceiling Directive of 2001.

During the mid 1980s the EU passed the Environmental Impact Assessment (EIA) directive (Council Directive, 85/337/EEC) enforcing a need to assess the potential impacts that new developments may have on their surroundings. The EIA procedure ensures that environmental consequences or likely significant effects of projects are identified and assessed before authorisation is given for the project to go ahead. The requirements of the EIA directive have been transposed into UK legislation through the Town and Country Planning (Environmental Impact Assessment) Regulations (1999) (Great Britain, Parliament, 1999) for all projects that are subject to approval through the planning system.

The EU Air Quality Framework Directive was followed by the 1st daughter directive in 1999 (Council Directive, 1999/30/EC), which for the first time introduced air quality limits for vegetation for sulphur dioxide (SO_2) and nitrogen oxides (NO_x). These limits were to be reached by 2001. With the introduction of the National Air Quality Strategy in 1997 (DETR, 1997), local authorities are since then committed to monitoring air quality and ensuring that air pollution from a wide range of sources is kept below the limits set out in the National Air Quality Strategy.

The National Emission Ceilings Directive (NECD) (Council Directive 2001/81/EC) came into force on 27 November 2001, with the aim of reducing emissions of pollutants that cause acidification, eutrophication and ground-level ozone in order to protect the environment and human health. The long-term objective is to ensure that pollutant levels remain below their critical loads and critical levels. A set of environmental objectives has been set for 2020 against a 1990 base to achieve the following:

- Acidification: areas where critical loads are exceeded to be reduced by at least 50%;
- Ground-level ozone (health): load above critical level for human health to be reduced by two-thirds and load in any area not to exceed a specified absolute limit; and
- Ground-level ozone (vegetation): load above critical level for vegetation to be reduced by one-third and load in any area not to exceed a specified absolute limit.

Towards those ends, the common position sets out annual emission limits for a number of pollutants. Table 1.2 shows the UK's annual limits (based on calendar years) for the year 2010 and 2020.

Table 1.2. UK Emissions for 2006 in relation to NECD and Gothenburg 2020 Targets

Pollutant	UK Emissions in 2006 (kt)	UK Emissions ceiling NECD target in 2010 (kt)	UK Gothenburg Protocol target in 2020 (kt)	Reduction (from 2006 level) to meet Gothenburg 2020 target
NO_x	1,595	1,167	711	55%
NH_3	315	297	283	10%

In 2001, the revised Large Combustion Plant Directive (LCPD) (2001/80/EC), came into existence in Europe and was aimed at reducing acidification, ground level ozone and particles by controlling emissions of SO_2, NO_x and dust (particulate matter (PM)) from large combustion plants. The LCPD focuses on combustion plants above 50MW thermal (MWth) running on either solid, liquid or gaseous fuels. The Directive takes into account advances in combustion and abatement technologies.

Since the inclusion of intensive farming in Europe in the Integrated Pollution Prevention Control Directive (2008/1/EC) pig and poultry installations above a certain size have to be assessed and permitted. For the pig industry, permits are required for farms with more than 2,000 production pigs over 30 kg and/or 750 sows, while the threshold for poultry units is 40,000 birds. IPPC has now been assimilated along with LCPD into the Industrial Emissions Directive (IED, 2010/75/EU).

Finally, and probably most key in protecting habitats and species, is the Habitats Directive (Council Directive, 92/43/EEC). Under the directive each Member State has set up a national network of protected sites to form a European network of Sites of Community Importance (SCIs) also known as Natura 2000 sites. These now include Special Areas of Conservation (SAC) and Special Protection Areas (SPA). Furthermore, Member States are required to take measures to maintain or restore habitats and species (listed under Annex I & II) to a *favourable conservation status*. The Habitats Directive does not adequately address off-site source emissions contributing to the transport of air pollutants over many kilometres. Emissions of nitrogen primarily from combustion and agricultural processes clearly present off-site pressures on the Natura 2000 network. Many sensitive Annex I habitats are naturally adapted to low nitrogen supply, and the eutrophication effects from nitrogen pollutants from the atmosphere contributes to ecosystem impacts. This fertilisation effect results in the loss of the most sensitive species, reduction in biodiversity, and the establishment by invasive species that prefer high rates of nitrogen supply.

Importantly for development plans and projects, the Habitats Directive under Article 6.3 introduces for the first time the precautionary principle in that developments can only be permitted once it has been ascertained that there will be no significant effect on the integrity of the protected site. The onus is therefore on the developer to prove that there will be no likely significant impact on the protected habitat, and an appropriate assessment report is required if a potential impact is envisaged. The specific text (92/43/EEC, para 48(i)) reads:

> *"**Assessment of implications for European site 48.(1)** A competent authority, before deciding to undertake, or give any consent, permission or other authorisation for, a plan or project which-*
>
> *(a) is likely to have a significant effect on a European site in Great Britain (either alone or in combination with other plans or projects), and*
> *(b) is not directly connected with or necessary to the management of the site, shall make an **appropriate assessment** of the implications for the site in view of that site's conservation objectives."*

In relation to the sources of nitrogen emissions, some may be many kilometres away (100s km) from the potentially affected sites. This long-range transport potential of nitrogen pollutants can trigger appropriate assessments where source and site are many kilometres from each other. In addition, local impacts are also very relevant for local sources of ammonia from intensive agricultural units (<2 km). Moreover, since these sources are almost always in rural areas, the potential for impacting a Natura 2000 site is increased.

1.4 Ammonia Abatement Techniques

It is at the local level where the detail of policy informed by science takes effect as it is translated into practice. Managing nitrogen losses on the farm and improving the efficient use of nitrogen are the key components for overall reduction in NH_3 emissions. For example on mixed livestock farms, between 10% and 40% of the nitrogen loss is related to NH_3 emissions (Oenema *et al.*, 2012). As we have seen, the need for nitrogen management is set out in the revised Gothenburg Protocol of CLRTAP in which Annex IX lists the measures for controlling NH_3 emissions from agricultural sources.

Importantly, an integrated approach is required as controlling emissions from all aspects of farming is vital if it is to be cost-effective. Focusing on one area of the system will often not be cost effective. For example controlling emissions from manure/slurry spreading will have less benefit overall if there are large losses from housing and storage. Annex IX importantly emphasises this by stating that "Each Party shall take due account of the need to reduce losses from the whole nitrogen cycle".

Techniques have developed over time with certain EU countries taking the lead and are currently practising these methods (e.g. The Netherlands and Denmark). Best Available Techniques (BAT) have also been set out in the EU for pig and poultry farming under the IED Directive (preceeded by the IPPC Directive). The Reference Document on Best Available Techniques for Intensive Rearing of Poultry and Pigs (BREF 07.2003) lays out BATs for on-farm processes and activities including nutritional feeding, feed preparation, rearing (housing), and collection, storage and spreading of manure.

Including the above from BREF, focus can be placed on five broad areas where ammonia abatement has already been well researched and proven as an effective method. These are:

- <u>Livestock feeding strategies</u> promotes the use of low protein livestock feed to reduce the volatilisation potential of NH_3 in faeces.
- Decreasing ammonia emissions from <u>animal housing</u> can involve decreasing the surface area fouled by manure using slatted floors; increased use of straw; rapid separation of faeces and urine; lowering the indoor temperature and ventilation; air scrubbing by removing NH_3 from the air through forced ventilation in combination
- Preventing emissions from <u>manure storage</u> facilities mainly involves the use of solid or floating covers or allowing the formation of a crust.

- Manure application techniques involve the application of manure either by injection or incorporating into the soil (ploughing in). Band spreading on the surface of the soil using a trailing shoe or hose can also achieve significant reductions. Slurry dilution is another method to decrease emissions often via irrigation systems.
- NH_3 emission from fertiliser application (urea) can be reduced by using urease inhibitors (urea fertiliser only); incorporating the fertilizer into the soil and irrigating after spreading are further techniques; the most effective method, up to 90% reduction, can be through the switching from urea to ammonia nitrate.

The above strategies can all been described as Category 1 methods as they are seen as practical to the farmer and there is some quantitative date to calculate emission reductions. UNECE, 2014 has described three broad groups of strategies for the abatement of ammonia emissions:

- Category 1 techniques and strategies that are well researched, considered to be practical or potentially practical, and there are quantitative data on their abatement efficiency, at least on the experimental scale;
- Category 2 techniques and strategies: These are promising, but research on them is at present inadequate, or it will always be difficult to generally quantify their abatement efficiency. This does not mean that they cannot be used as part of an NH_3 abatement strategy, depending on local circumstances;
- Category 3 techniques and strategies: These have not yet been shown to be effective or are likely to be excluded on practical grounds.

Costs for implementing abatement techniques ranges from a net saving of €1 per kg NH_3 -N saved (for some manure spreading techniques) up to around €10 per kg NH_3 -N saved for implementing air scrubbers in housing systems (Bittman *et al.*, 2014). Such cost calculations may also be compared with environmental benefits (van Grinsven *et al.*, 2013).

Trees as abatement techniques have, up till now, not been included in programmes or current frameworks for reducing NH_3 emissions. This is possibly due to the fact that the tree-belt systems that are planted around farms only reduce on-farm emissions: NH_3 is still emitted into the atmosphere and the trees are only operating in a scavenging way. However, this approach is attracting interest from many outside stakeholders like conversation agencies who are interested in using trees to buffer sensitive habitats in the local landscape. A further positive aspect of planting trees is to offset greenhouse gas emissions as trees can sequester carbon as they grow.

1.5 Air Pollution and Trees – processes and practice

1.5.1 Processes of deposition

The transport of pollutants to the plant surfaces from the planetary boundary layer occurs by turbulent diffusion with the rates determined by the wind speed and the aerodynamic roughness of the surface (Smith *et al.*, 1999). Turbulence causes the formation of eddies responsible for the transport of pollutants to the plant surface. Asman (2008) nicely described the effect of eddies near the surface as "the flutter of leaves of trees, irregular movements of dust particles, or ripples and waves on water surfaces."

Deposition of gases can be described as being the product of the gas concentration and a deposition velocity (V_g) which is the reciprocal of a number of resistances the pollutant gas has to overcome before depositing to the leaf surface (Smith *et al.*, 1999):

$$V_g(z) = \frac{1}{r_a(z) + r_b + r_c}$$

where:

$V_g(z)$ = deposition velocity from a height z

r_a (z)= aerodynamic resistance

r_b = lamina boundary layer

r_c = bulk canopy resistance

The first resistance encountered is the aerodynamic resistance (r_a) which is a function of the friction velocity (a measure for the wind speed and turbulence) and the atmospheric stability. The next resistance is the laminar boundary layer (r_b) which occurs very close to the leaf surface where the transport of the pollutant gas is no longer by turbulence but by molecular diffusion. The final resistance is known as the bulk canopy resistance (r_c) or surface resistance of the whole canopy. Once overcoming these resistances gases can be taken up by the stomata and is dissolved into the apoplastic solution. Or at the plant surface deposition of NH_3 (which is highly soluble) occurs at the cuticle especially if the leaf is covered by a layer of water, or the gas is adsorbed to surfaces waxes (Jones *et al.*, 2007). There is a substantial literature on dry deposition processes. For example, in the case if NH_3 exchange processes can also be bidirectional (Massad *et al.*, 2010; Flechard *et al.*, 2012), and other parametrisations are needed for agricultural land. However, the deposition velocity approach provides a general model that is applicable in many situations.

1.5.2 Tree as scavengers of air pollutants

What makes trees particularly effective scavengers of air pollutants is their effect on turbulence (Beckett et al 2000, Nowak 2000). Having a higher roughness length (and lower aerodynamic resistance r_a) aids mechanical turbulence and promotes dry deposition to the surface. Dry deposition rates to trees exceed those to grassland by typically a factor of 3–20 (Gallagher *et al.*, 2002, Fowler *et al.*, 2004). This implies that the conversion of grassland and arable land to trees or targeted management of existing wooded areas, can be used to promote the removal of ammonia from the atmosphere.

Figure 1.11 shows the key processes by which trees can have a beneficial effect as landscape structures to mitigate NH_3 air pollution. They can be used to 1. reduce emissions from slurry lagoons by reducing the wind speed over its surface; 2. recapture emissions by the trees themselves through increased turbulence and deposition velocities; 3. increase the dispersion above the canopy through increased mixing thereby reducing deposition to nearby sensitive habitats.

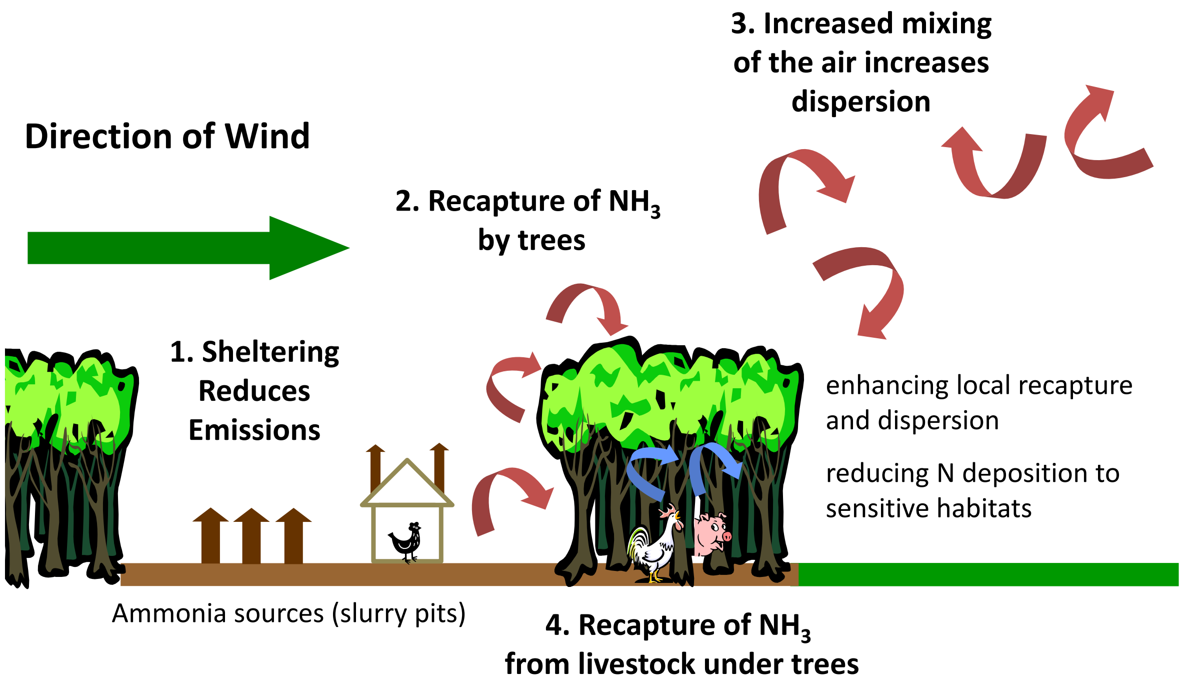


Figure 1.11. Effect of trees on capturing and dispersing ammonia emissions by sheltering of storage pits, and recapture downwind of animal housing (Bealey *et al.*, 2011)

As the plume from the source approaches the tree-belt part of the plume is pushed upwards and does not impact with the tree-belt itself. Instead, it flows over the top where turbulence is increased leading to additional dry deposition. As the rest of the plume enters the tree belt the air flow (wind speed) is reduced and NH_3 capture occurs. This means that at low wind speed the capture of NH_3 is greater. Conversely at higher wind speeds the residence time of the plume in the tree belt is shorter, so that the amount dry deposited becomes less (Asman, 2008). As the plume passes through the tree belt gases only have to be transported over small vertical distances of the order of 0.1 m, resulting in a negligent aerodynamic resistance. The main resistance influencing deposition to the leaves or needles is the laminar boundary layer resistance, and stomatal resistances.

In general, capture also increases with increasing atmospheric stability which are represented by low wind speeds. Under stable conditions the vertical mixing of the plume is reduced and concentrations are usually higher.

When considering emissions of NH_3 reactions in the atmosphere to ammonium NH_4^+ should also be taken into account. In other words how much of the emitted NH_3 is deposited as wet or dry NH_4^+. For receptors close to sources (e.g. <1 km) dry deposition is driven by the gaseous form (NH_3) as conversion to NH_4^+ has not had time to occur. Furthermore, the dry deposition velocity of NH_3 is about five times higher than for particulate NH_4^+. (Ferm, 1998)

Several previous studies have shown the effectiveness of trees in capturing pollutants. However these tend to mainly focus on particulates (e.g. $PM_{10}/_{2.5}$) in relation to improving urban air quality. For example Nowak *et al.* (2013) modeled $PM_{2.5}$ removal by trees in ten US cities and associated health effects. McDonald *et al.*, (2007) modeled the potential of urban tree planting to mitigate PM_{10} across two UK conurbations. Novak *et al.* (2006, used meteorological and air pollution data to show the removal of O_3, PM_{10}, NO_2, SO_2, CO by urban trees and shrubs across the United States. Urban parks in Tel Aviv were also found to have a mitigation effect on the concentration of NO_x, CO and PM_{10} (Cohen *et al.*, 2014). Some studies have looked at the suitability and pollutant capture efficiency of particular trees. For example, Becket *et al.*, 2001 showed in wind tunnel experiments that coniferous species, and broadleaf trees with hairy leaves, had a greater effectiveness at capturing particles than other broadleaf trees.

Studies examining the usefulness of trees to capture ammonia are limited. The capture of ammonia by surrounding vegetation has been studied by Patterson *et al.* (2008a), who observed lower NH_3 concentrations were measured when potted trees were present downwind of the poultry house fans compared with when the trees were removed (16.4 versus 19.3 ppm). Further work by Patterson et al. (2008b) also showed that the foliar N concentrations of Spike hybrid poplar and Norway spruce were greater near the exhaust fans compared to control plants at 40 m or more. Spike hybrid poplar was found to retain greater foliage N than Norway spruce. Both species were able to capture NH_3 near the housing's fans.

In a modelling study, Dragosits *et al.* (2006) estimated how tree belts can reduce deposition to sensitive ecosystems, with trees surrounding the sensitive habitats being more effective than trees around the sources for their scenarios. Wind tunnel experiments by Famulari *et al.*, (2015) showed that significant NH_3 was recaptured using 2m tall *Picea Abies* (Norway spruce) placed in 5 rows in a wind tunnel (Figure 1.12). NH_3 and CH_4 were released simultaneously, and concentrations of both were measured upwind and downwind of the source, through and beyond the trees. The depletion of methane with increasing distance from the source was due to dispersion, whereas the depletion of ammonia depended on interaction with the trees stomata, as well as other surfaces. Capture under wind speeds of 5 ms^{-1} was around 30% for low concentration releases (180 ppbV) while recapture was 15% for a high concentration releases (750 ppbV) also at 5ms^{-1}, with more NH_3 being recaptured under wetter conditions (up to 43%).

Figure 1.12: Side view of the wind tunnel with 28 trees and 16 growth lights on. (Bottom) Front view of the wind tunnel with 28 trees and 16 growth lights on.

Modelling research undertaken by Asman (2008) on the entrapment of ammonia by shelterbelts showed that capture of dry deposited gaseous ammonia increased with the height of the shelterbelt and the stability of the atmosphere (favouring neutral conditions), but decreased further away from the source to the shelterbelt. At 200 m away from a source the model predicted that a maximum 37% of the emission of a ground level point source of ammonia can be dry deposited before the plume reaches a shelterbelt that is located 200 m downwind. Asman estimated a further 11% could be removed by a 10 m high shelterbelt. Experimental approaches to measure ammonia recapture were carried out in a Scots pine plantation in Scotland by Theobald *et al.* (2001). The field experiment looked at the difference between the ammonium in the throughfall of the fumigated and non-fumigated parts of the woodland. From these measurements it was estimated that 3% of the emitted ammonia was deposited to the leaf surfaces. Importantly these measurements did not include the foliar uptake of ammonia, so the actual recapture would have been larger. From the tracer-ratio measurements, ammonia recapture of up to 46% of the ammonia entering the woodland was predicted. The author notes that the woodland was in general very open and that the estimated 3% recapture could be larger if the woodland had been better designed (e.g. less open).

While initial modelling using the MODDAS model (Theobald *et al.*, 2004) showed a recapture of ammonia emissions up to 15%. It is evident that the fraction of NH_3 recaptured varies greatly according to the local site configuration.

Theobald *et al.*, (2004) produced designs of suitable tree shelter belt configurations to maximise NH_3 recapture (Figure 1.13). The intake zone captures low-level ammonia close to the building, and the design of the canopy in the recapture zone funnels the ammonia underneath the canopy. The backstop zone prevents ammonia passing out the downwind edge of the tree belt and forces the ammonia up through the canopy ensuring maximum recapture. Conifers have the ability to capture larger amounts of ammonia (or other pollutants) broadleaved trees. Freer-Smith et al., 2005 showed that conifers with their smaller leaves and more complex shoot structures were able to capture larger amounts of particulate matter than broadleaves. This capture efficiency is also magnified in the winter months because the needles of conifers are not shed during the winter. However, some research has shown that deciduous trees are better at absorbing gases due to having a larger stomata to surface area than conifers (Adrizal, 2008).

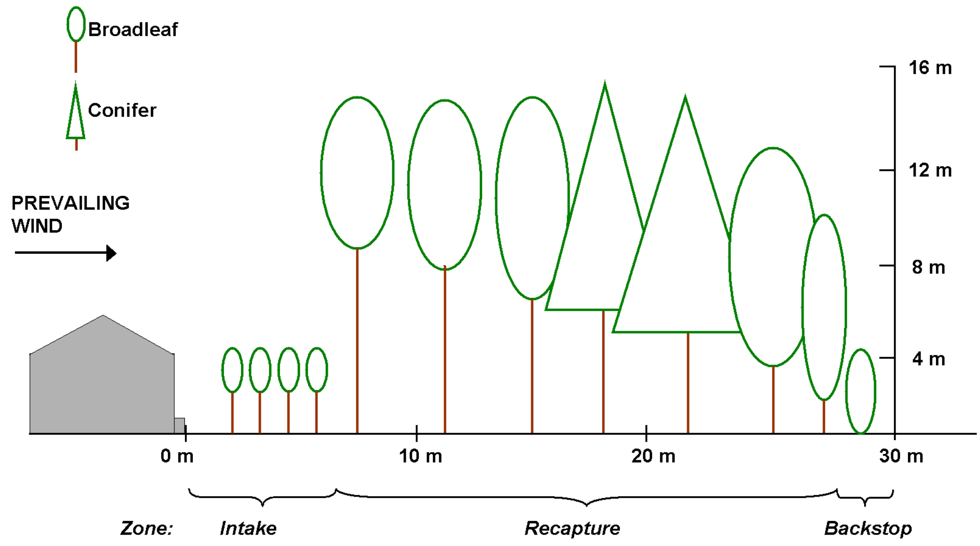


Figure 1.13. Tree belt downwind of side-ventilated livestock housing (Theobald *et al*., 2004)

Adding extra N to forest systems in the short term can lead to increases in nitrogen cycling and forest productivity, but these effects can disappear after further years as added N is sequestered in the wood and the soil organic matter (Aber, 1989), and the capacity for nitrogen uptake drops off leading to N leaching. The trees themselves can be sensitive to other air pollutants, frost and insect attack and drought (Nihlgard, 1985). As added N inputs exceed the biological demand and the storage capacity of the soil nitrate will eventually leak as the system become nitrogen saturated. The N leaching can affect the ion balance and lead to soil acidification and nitrate can leach into freshwaters leading to eutrophication.

The fate of adding N to a forest system has been studied numerous times. Using labelled ^{15}N Tietema *et al.,* 1998 showed that about 10-30% (ranging from 3-42%) of added N was taken up by the trees, with 10-15% being retained in the mineral soil. In the organic layer 20-45% was retained at low N inputs (0-30 kg N ha^{-1} yr^{-1}) but less was retained (10-20%) at high N inputs (30-80 kg N ha^{-1} yr^{-1}). Uptake of ammonium by trees (and other vegetation) and the soil microbes can lead to acidification of some sensitive soils and a reduction in the availability base cations. Furthermore, increased nitrification rates enhances the onset of nitrate leaching.

Tree Species Selection

Some work has specifically looked into the choice of species although this has focused on particulate capture in urban environments. Theobald *et al.*, 2004 looked at the selection of suitable species for planting for ammonia mitigation. The selection of suitable species was assessed examining a number of key criteria: functionality – canopy structure for maximum recapture; site conditions – species selection depending on soil type, topography etc.; tree species ecology – choice dependent on relative growth rates and suitability for mixed plantings; and management – ability to manage the woodland e.g. access for machinery.

The most important criterion for tree species selection is for the tree to maintain healthy growth. Other criteria can also include species which have a high nitrogen requirement and tolerance of nitrogen. Trees which can grow rapidly and have a high leaf area index (LAI). Some mix of evergreens are also desirable for year round recapture and species that can withstand coppicing of pruning are preferred. Some species identified as not suitable included cherry (low LAI), chestnut (large leaves) and oak (slow growth rate). However many other woodland tree species that are suitable included beech (shade tolerant), field maple (suitable for most soils and can be coppiced), birch (suitable on most soils, can be coppiced) and Scots pine (high LAI) and Sitka spruce (high LAI and fast growing).

Modelling carried out in this study focused on using beech (*Fagus sylvatica*) and Sitka spruce (*Picea sitchensis*) in tree belt systems, although in practice more mixed planting should be encouraged.

1.6 Thesis Overview

This thesis examines the potential for NH_3 air pollution to cause environmental impacts and how these impacts may be reduced. Specifically, it explores the suitability of using tree belts to capture ammonia around livestock housing and manure storage systems. Planting trees near emission sources can be seen as a way to increase the removal of ammonia from the atmosphere, thereby reducing the potential impacts on nearby sensitive ecosystems. This can be looked at as on-farm (or farm-gate) reduction of emissions as ammonia is captured at source.

1.7 Research Questions

Chapter 2 - Existing legislation controlling ammonia emissions does not adequately or systematically address the impacts of nitrogen on the Natura 2000 network, or the wider objectives of the Habitats Directive. This chapter addresses: What are other EU countries experiences in regulating nitrogen pollution sources, and what are the policy measures to combat exceedance?

Chapter 3 – Models source apportionment across the Natura 2000 network for nitrogen and sulphur deposition. It assesses the exceedances and the source matrix of sites and the national picture. What is the level of critical load exceedance across the network and what are the dominant sources for policy makers to focus on to reduce exceedance?

Chapter 4 – Explores how agroforestry systems can be implemented as an ammonia abatement option at the local scale and specifically it asks: How much 'on-farm' ammonia can be recaptured by planted trees according to different scenarios?

Chapter 5 – Scales up to estimate the outcome in terms of NH_3 mitigation from adopting agroforestry systems at the national scale. For example it asks: How much can agroforestry systems reduce ammonia emissions on a national level. What is the effect on semi-natural areas and how many hectares of trees are required to obtain realistic reductions in on-farm emissions?

Chapter 6 – examines the benefits and costs of implementing trees and how they compare with other techniques. It shows that this technique can offer many win-win solutions. It discusses the question: What are the comparative costs and additional benefits of agroforestry systems?

Chapter 7 – Provides an overall discussion and summarises the conclusions of this thesis. It finally discusses the question: What is the efficacy of planting trees for ammonia abatement?

Chapter 2. Review of approaches to air quality management of Natura 2000 sites across Europe

2.1 Aims and Objectives

Chapter 3 discusses the approaches to air quality management across the Natura 2000 network in Europe. It primarily focuses on nitrogen deposition with special attention given to agricultural activities and emissions of ammonia. Chapter 2 showed that livestock emissions of ammonia are the most dominant source across the network, and all but 10% of Special Areas of Conservation (SAC) did not have livestock ammonia as their dominant source. This chapter also examines the current policy frameworks (extending to air, agriculture and nature conservation policies) for reducing exceedance on the Natura network - future options are presented.

2.2 Background

From Chapter 2 we can see that critical loads exceedance for eutrophication effects, and to a lesser extent acidification, is prevalent across the UK Natura network and that nitrogen deposition represents a major threat to biodiversity. In fact, across Europe over 70% of Natura 2000 sites are at risk from eutrophication with over 70% of the Natura 2000 area in Europe (EU28) exceeding critical loads for nutrient nitrogen (see Figure 2.1, CCE, 2014) - the figure for acidification is 5% exceedance.

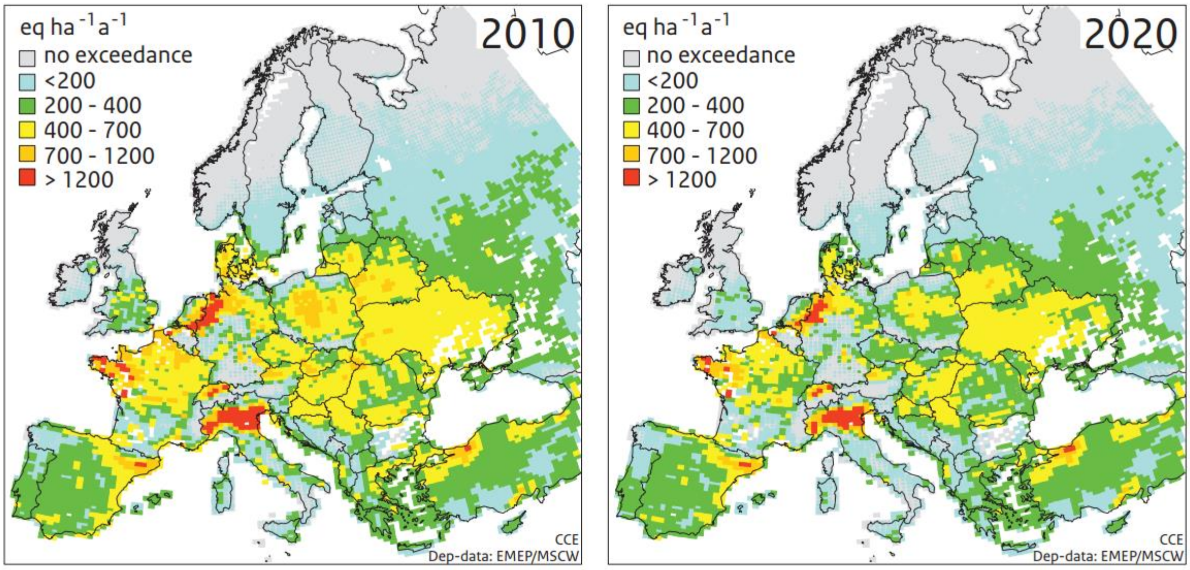


Figure 2.1 Average Accumulated Exceedance (AAE in eq ha–1 yr–1) of critical loads for eutrophication are exceeded by N deposition for 2010 and 2020 based on Gothenburg Protocol emission limits (CCE 2014)

The magnitude of emission reduction required by 2020 to eradicate critical load exceedance of nutrient nitrogen is a round 80% (Figure 2.2).

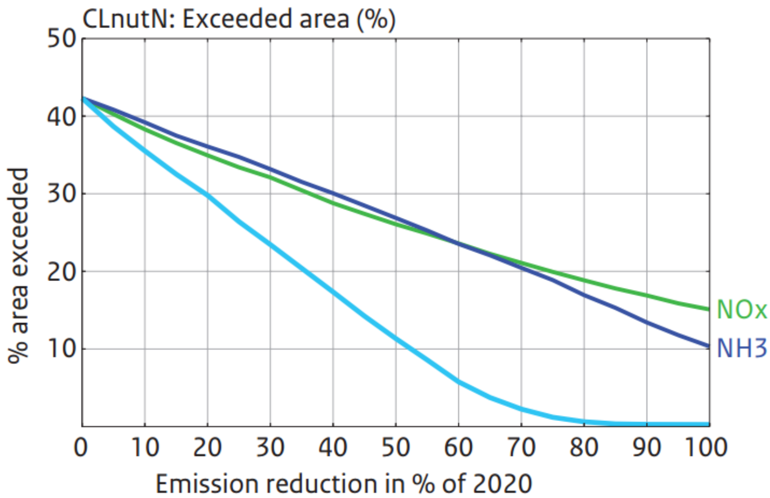


Figure 2.2 European ecosystem area exceeded (in %) of CLnutN as function of uniform emission reductions (RGP 2020=100%) of NO_x (green lines), NH_3 (blue) and total N (turquoise) (CCE 2012).

2.2.1 Habitats Directive

The Natura 2000 network of habitats and species is protected by the Habitats Directive (92/43/EEC) and the Birds Directive (79/409/EEC). Annex I habitats and Annex II species (non-birds) are key features for Special Area of Conservation (SAC), while birds are protected under a network of Special Protection Areas (SPA). The Habitats Directive is the cornerstone of Europe's nature conservation strategy where each Member State is required to maintain and restore habitats to a 'favourable conservation status'. Favourable status is when the natural range of a habitat, and the area it covers, are stable or increasing, and the specific structure and functions are present and likely to continue into the future.

The main objectives of the Habitats Directive are:

> *"…to contribute towards ensuring biodiversity through the conservation of natural habitats and of wild fauna and flora in the European territory of the Member States to which the Treaty applies" (Article 2.1); and*
>
> *"…to maintain or restore, at favourable conservation status, natural habitats and species of wild fauna and flora of Community interest" (Article 2.2)*

Article 3 of the Directive required the establishment of protected sites across Europe that would contribute to protecting habitats and species identified in Annex I and II. Member States are required to establish conservation measures for each SAC through management plans that correspond to the ecological requirements of Annex I habitats and Annex II species that are present on the site (Article 6.1). Many Annex I habitats are adapted to low nitrogen inputs, and the fertilisation effect from nitrogen polluting compounds from the atmosphere can negatively affect sensitive ecosystems. Effects of excess nitrogen inputs leads to loss of the most sensitive species (e.g. lichens and bryophytes), and an increase in more invasive type species that prefer high rates of nitrogen (e.g. coarse grasses). This results in a gradual decrease in biodiversity - i.e. a net loss in species (Stevens *et al.*, 2010).

Article 6.3 of the Directive ensures strict protection measures from new plans or projects where they can only be permitted if they are shown to have no significant effect on a Natura 2000 site. This precautionary approach puts the onus on the applicant to show their project has no significant effect on site integrity.

Significantly, plans and projects in the Directive have not been restricted to on-site activities (e.g. water abstraction), but include activities situated off-site that still exert an effect on the site. For example, emissions of reactive nitrogen compounds from industrial and agricultural installations represent impacts from off-site activities. The Habitats Directive does not directly address nitrogen impacts and until now there has been no common European approach for determining the impacts of nitrogen deposition on individual sites or on conservation status. Off-site sources present difficulties in assessing and attributing impacts because deposition can result from local sources (1-2 km), or very far away sources (>100kms).

2.2.2 Air Pollution Legislation

Supporting legislation to control emissions of air pollutants is provide by a suite of Directives and protocols which range from regulatory frameworks to environmental impact assessments. The Convention on Long-Range Transboundary Air Pollution (CLRTAP) has included eight protocols that identify specific measures to be taken by Parties to cut their emissions of air pollutants including SO_2, NO_x, NH_3, Volatile Organic Compounds (VOCs), heavy metals, Persistent Organic Pollutants (POPs). The 1999 Multi-pollutants and multi effects protocol (Gothenburg protocol, UN ECE, 1999), amended in 2012, includes national emission reduction commitments to be achieved by 2020 and beyond. Annex IX of the Protocol also sets out a list of ammonia control measures that signed Parties are obligated to apply. These include measures to reduce emissions from manure storage and spreading, animal housing installations, and fertiliser application. In Europe, the CLTRP is assisted by way of the National Emissions Ceilings Directive (NECD, 2001/81/EC) which sets limits of pollutant emissions for each Member State.

The Industrial Emissions Directive (IED, 2010/75/EU) came into force in 2010 and replaced 7 Directives with the aim of taking an integrated approach to controlling industrial emissions. The most important of these are the Large Combustion Plant Directive (LCPD, 2001/80/EC) and the Integrated Pollution Prevention and Control

Directive (IPPC, 2008/1/EC). LCPD aimed to reduce emissions of SO_2, NO_x and particulates from large combustion plants, while IPPC concerned the issuing of permits by regulators for emissions to air. The integrated IED now also includes the issuing of permits of pig and poultry farms over a certain size. Other policies also play their part in regulating emissions including the Nitrate Directive (91/676/EEC) and the Environmental Impacts Assessment Directive (EIA). The EIA Directive requires the assessment of effects of most projects causing NO_x and NH_3 emissions on SACs and SPAs.

2.3 Assessing Plans and Projects on Natura 2000 sites

Article 6 is one of the most important articles in the Habitats Directive as it defines how Natura 2000 sites are managed and protected. Plans and projects can only be permitted if they are shown to not adversely affect the integrity of a Natura 2000 site, unless there is some form of overriding public interest why it should proceed. As described above the focus has often been on on-site activities while off-site activities, including the polluting effect of local and transboundary air pollution sources, have been harder to assess and hence have received less attention. Nitrogen deposition primarily from combustion and agricultural processes clearly present off-site pressures on the Natura 2000 network. Furthermore, Chapter 2 showed that agricultural sources, and their associated ammonia emissions, are often the dominant source in the rural landscape. This is not surprising as for many countries Natura sites are often interspersed between agricultural lands.

The process for assessing plans and projects has been set out in the Habitats Directive and subsequent guidance.

> *Article 6.3 - Any plan or project not directly connected with or necessary to the management of the site but likely to have a significant effect thereon, either individually or in combination with other plans or projects, shall be subject to appropriate assessment of its implications for the site in view of the site's conservation objectives. In the light of the conclusions of the assessment of the implications for the site and subject to the provisions of paragraph 4, the competent*

> *national authorities shall agree to the plan or project only after having ascertained that it will not adversely affect the integrity of the site concerned and, if appropriate, after having obtained the opinion of the general public.*

The assessment process broadly adopted across the EU is set out in four stages – Stage One: screening; Stage Two: Appropriate assessment; Stage 3: Alternative solutions; and Stage Four: Final decision. An initial decision on whether the plan or project is *likely to have a significant effect* is the first test. Assessing significant effects is hard to untangle but the conservation objectives for the site and designated features should be considered. However, the 'likeliness' test is treated as a screening process where an environmental limit (often a critical load or level) for a habitat is compared with the modelled deposition from the project or plan. *In combination* effects from other existing source also need to be taken into account with cumulative impacts ascertained. If no likely significant effect can be ruled out an appropriate assessment is triggered involving a full detailed assessment of the plan or project and the site. The appropriate assessment should focus on the site's conservation objectives and importantly determine whether a favourable status can be maintained. Any potential mitigation methods should be examined at this point too. It is important that emphasis should be given to abating impacts at source rather than abating the impacts at the receptor, e.g. the Natura 2000 site (EC, 2002). The final stage of the appropriate assessment is to show whether the plan of project will not adversely affect the integrity of the site. Again this refers directly to the site's conservation objectives of the Annex I habitats, or the Annex II species for which the site was designated.

Integrity can be defined as: "the ability of a site to maintain a coherent structure as a habitat or for supporting a complex of habitats and species" (EC, 2000). Ways of assessing this can include answering questions like "Does the project or plan have the potential to interfere with the balance, distribution and density of key species that are the indicators", or "Does the project or plan have the potential to reduce the diversity of the site?

Under Article 6.4 the competent authority is required to arrive at a conclusion regarding the consequences of the plan or project in relation to the integrity of the site concerned. If it is concluded that the plan or project would have no adverse effect, then the plan or project can proceed. If an appropriate assessment identifies that any activity cannot be proven to have no adverse effect, then the competent authority must refuse permission for the proposed plan or project.

Approaches to the assessment of plans and projects under Article 6.3 are comparable across a number of countries where the process has been well formalised (Bealey *et al.*, 2011). In particular the UK, Denmark, Germany and The Netherlands have similar procedures for assessing nitrogen deposition. Each country adopts the empirical critical loads approach in assessing nitrogen deposition exceedances. To achieve this Annex I habitats have been allocated a 'best fit' site relevant critical load where it can be compared with on-site deposition. These thresholds are used in both the screening Stage one process helping to determine a likely significant effect, and in appropriate assessments to determine adverse effects.

The critical load approach is now generally well established as a mechanism for assessing likely effects. It is the main tool used by regulators and conservation practitioners in their staged process (Bealey *et al.*, 2011). However, issues arise where the process contribution is small (e.g. < 1% of the critical load) or where an exceedance already exists. Taken together with other contributing sources can lead to many small '<1%' additions to the background adding up to 20-30%. So while a <1% contribution could be described as 'de minimis' - or, small enough to be ignored - the cumulative effect of multiple projects cannot be ignored. When it comes to judgements on 'no adverse effects' some countries - for example, Germany and the UK - have set a percentage of the critical load which can range from 1% and 3% to 10% or even higher in some cases.

In the UK an acceptable process contribution of 20% (in combination) of the critical level/load has been used in the assessment of impacts from existing installations from intensive livestock sector, but no per cent threshold has been set yet for generic application. In Germany the critical load threshold test of 3% is backed-up by a process contribution limit of 0.3 kg $ha^{-1}yr^{-1}$. The livestock sector in Denmark has also set limits: not on critical loads, but on allowable contributions of nitrogen deposition per farm. The allowable contribution for one livestock unit is set at 0.7 kg N $ha^{-1}yr^{-1}$ as long as there are no other livestock farms within a certain distance. If there is one other livestock farm within this distance, a total of 0.4 kg N $ha^{-1}yr^{-1}$ is allowed, down to 0.2 kg N $ha^{-1}yr^{-1}$ if there are two or more other livestock farms (Bjerregaard, 2011). This approach fulfils the required 'in combination' test laid out in the Directive and has comparable limits with the critical load indicator approach – 10% of a typical critical load of 10 kg N $ha^{-1}yr^{-1}$ is 1 kg N $ha^{-1}yr^{-1}$. In addition to these allowable deposition limits and percentages of the critical load an exception is often used whereby if the site is in unfavourable status than any extra deposition is seen as incompatible with the conservation objectives of the site.

Meanwhile, in France air pollution is primarily seen as a human health issue, and ecosystem impacts of air pollution are not considered important or are at least significantly underestimated (Alard, 2013). Despite this, in 2013 a national plan was adopted laying out plans to reduce the use of mineral fertilizer and increase organic nitrogen efficiency. The main driver in France appears to be the cost involved for the farmer.

There are many broad similarities when it comes to establishing conservation objectives and measures across the Member States. However, there are some substantial differences in how nitrogen deposition is used in setting conservation objectives, including the setting of critical loads and levels for Annex I habitats. Some countries do not see nitrogen as a threat as impacts are not widespread (e.g. Scotland) (Whitfield *et al.*, 2013). Often country agencies find it difficult to separate out the impacts of nitrogen deposition and that of grazing (or under grazing). In the Netherlands, nitrogen deposition is indirectly taken into account via the designation process and that ecological requirements are met.

2.3.1 The Netherlands: Integrated Approach to Nitrogen | Programmatische Aanpak Stikstof (PAS)

The approach taken in The Netherlands warrants special mention as it provides an integrated approach to nitrogen deposition and impact assessment for plans and projects. Although nitrogen deposition was falling at the national level, it was often impossible for individual applicants to demonstrate that their plan or project would not adversely affect the integrity of the site concerned; hence the need for an integrated approach. The Integrated Approach to Nitrogen (Programmatische Aanpak Stikstof (PAS), Dutch Ministry of Economic Affairs, 2015) promotes government agencies and developers working together to reduce nitrogen emissions and also facilitate economic development. It has three main aims: to provide scope for economic development, make nature sites more robust by restoring the nature quality, and reduce nitrogen emissions. There are 69 Natura 2000 sites which have been identified as having sensitive habitats and species to nitrogen. PAS has two main strategies to cope with nitrogen impacts and comply with the Natura 2000 targets:

- Taking measures at source to ensure a lasting reduction in nitrogen deposition: for example, low-emission housing;
- Implementing restoration measures for nitrogen sensitive nature areas e.g. turf cutting, mowing, hydrological measures etc.

By approaching the problem at both the source and receptor end, PAS is able to accommodate, to a certain extent, further economic growth and new installations. All new or expanded installations can make use of the mitigation options packaged up in measures for each Natura site (consisting of ecological restoration strategies and source measures). Implementation of these measures is a statutory requirement for the authorities concerned because the measures are part of the appropriate assessment (Article 6 Habitats Directive).

Under PAS, set amounts of nitrogen deposition allowed are given for each site and these cannot be exceeded. The overall net nitrogen that is deposited to a site must not increase, and must aim to fall. Nature targets should be met, nitrogen deposition must continue to fall, and restoration measures must ensure restoration or at least no further deterioration. This differs somewhat from the use of threshold values and the use of critical loads. Instead PAS allows economic development to take up this reduction. This "headroom for development" assumes economic growth of 2.5% per year combined with existing nitrogen policies. The allocation of nitrogen deposition is split into 4 segments: deposition reserved for autonomous growth (e.g. extra traffic on existing roads), deposition for plans or projects below a limit 0.14 kg of N hectare^{-1} year^{-1}, deposition for priority projects (e.g. motorway expansion), and deposition for single applications. The headroom for development/deposition is calculated for each nitrogen sensitive hectare in each Natura 2000 site based on anticipated reduction in nitrogen from source measures and restoration measures. On top of this are a suite of mitigation options to further reduce emissions.

PAS is driven by an online calculation software tool AERIUS (Figure 2.3). AERIUS supports the process for issuing permits, the monitoring of the PAS and spatial planning in relation to nitrogen.

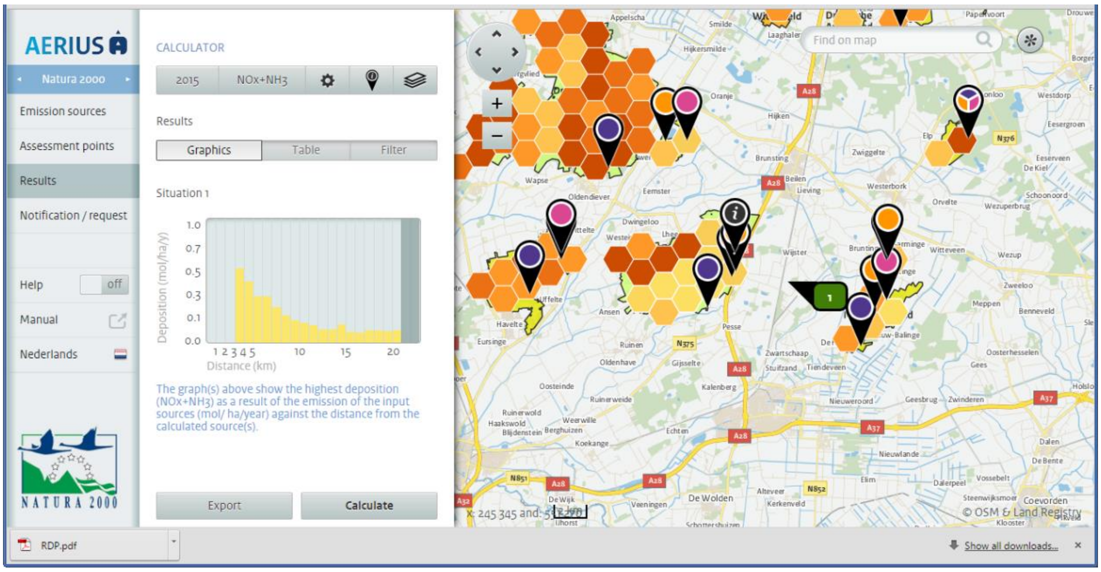


Figure 2.3. AERIUS calculator

The integration of nitrogen policies set out in PAS is highly innovative although options for both restoration and mitigation have been explored separately and PAS is now integrating these options. One of the key benefits of the AERIUS system is that it can manage the impacts of critical load exceedance by auditing sources and contributions across the country while at the same time linking critical loads and levels at protected sites per hectare in a hexagonal grid. Costs are expected to be high as authorities will have to pay for incorporating measures such as like translocating problematic farms, while restoration costs are not cheap. AERIUS itself is a very impressive piece of software but has also come at a cost of several million Euros.

2.4 Current and future policy options for tackling nitrogen deposition impacts on Natura 2000 sites

It is important to note that many SACs and SPAs across Europe still exceed their critical loads even with such a precautionary approach laid out in the Habitats Directive (CCE, 2014). Exceedance of critical loads and levels at a site infers that an adverse effect on the integrity of the site will follow. This paradox of high protection alongside continuing high exceedance can be attributed to a number of reasons:

- Plans and projects under Article 6.3 of the Habitats Directive in most Member States are only supported by regulatory requirement under IED in which an appropriate assessment is triggered. For agricultural sources this only covers pig and poultry installations which represent a small percentage of the overall ammonia emissions. This means that big emitters like cattle are effectively unregulated. This contrasts highly with measures in The Netherlands under PAS where every source contributing greater than 1 mol N/ha/yr (0.14 kg) is taken into account.
- In combination effects are challenging to model even when emissions in an area are known. Full auditing of existing sources is often not apparent as information on animal practices can be hard to collect and farm locations may differ from the actual location of barns for example. This means that gaining a picture of what is happening in the landscape and accounting for all sources can be an impossible

task. Large database systems (e.g. AERIUS in The Netherlands) seek to address these issues: all emission data is stored in a database.

- The large spatial distribution in nitrogen deposition, split between local and long range, makes it hard to regulate as sources can be 100s of kilometres away. Source attribution modelling provides a mechanism for assessing receptor site deposition, particularly for long range sources where their deposition contributions are often small but collectively add up to large amounts. This is contrasted by issues with modelling local sources of ammonia emissions which are highly spatially variable relying on suitably fine resolution data and modelling (50 m). Without high resolution data, multiple hot-spots of ammonia emissions are effectively unaccounted for in the landscape.

2.4.1 Current Policy Options

Figure 2.4 shows the key legislation associated with agriculture production including the regulatory legislation used to control losses to air and water illustrated by the 'hole-in-the-pipe' model.

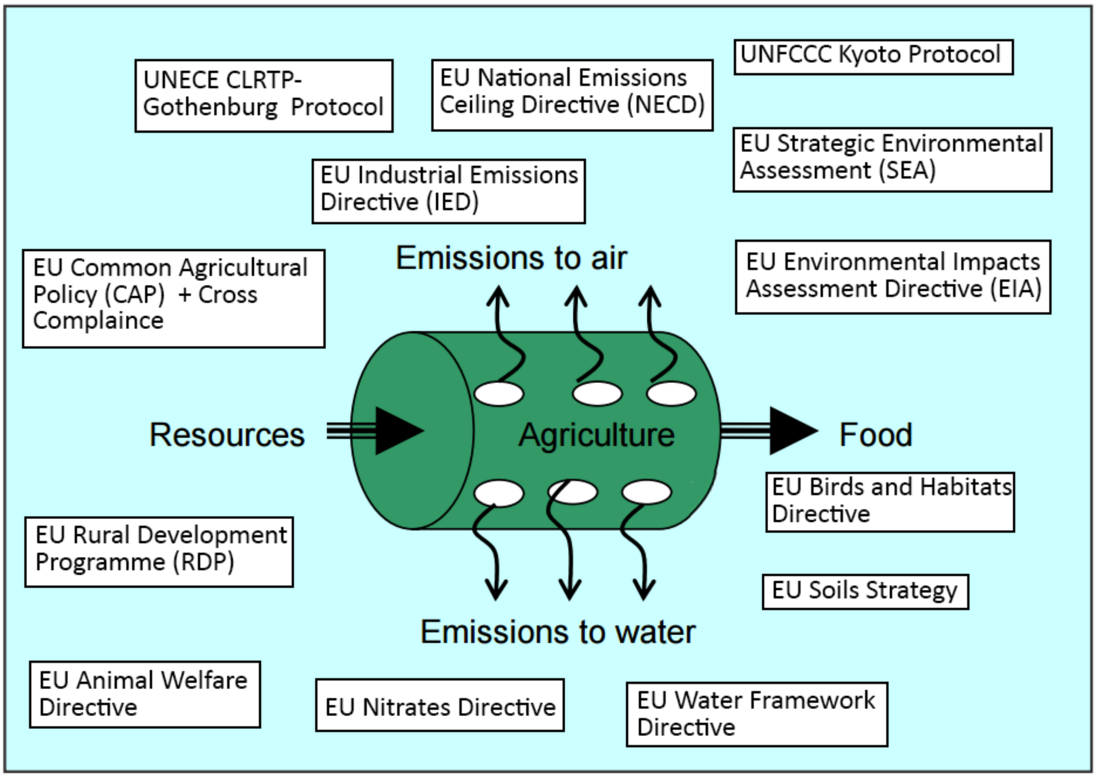


Figure 2.4. Agriculture symbolised as 'a hole-in-the-pipe-model', showing resources going in and food out, with losses to the air and water. Key legislation associated with reducing agricultural emissions and impacts to air and water are shown (adapted from Oenema, 2012).

Air Pollution Policy

Current policy options to support the Habitats Directive have been touched upon in the Background section of this chapter and also in Chapter 1. The backbone of reducing eutrophication and acidification effects on ecosystems lies with the UNECE Gothenburg Protocol, adopted within Europe in the form of the NECD. Target emissions are set for air pollutant species for each member which should provide in-country reductions of reactive nitrogen (N_r) and also transboundary exports and imports between countries. In addition to the national target emissions, annexes of the protocol lay out mitigation options for reducing both NO_x and NH_3. To achieve a reduction in ammonia emissions, a much wider uptake of best practice across the EU is required, including measures that focus on areas like fertiliser management (e.g. urea substitution); low emission manure application and storage; livestock feeding strategies; and low emission 'state of the art' housing facilities. Under The Clean Air Policy Package adopted in 2013 (EU COM/2013/0918), new ceilings for ammonia emissions have been set up to 2030. It will strive to achieve the objective of 'no significant impacts', and compliance with the limits values of the WHO air quality guidelines (by 2050).

The UNECE and NECD are not set up to protect impacts on protected sites from individual applications for new plans or projects, even though they specify national targets,. The IED, however, comes into force at this point controlling air pollution release by permitting installations above a certain size, and further specifying that these installations must obtain a permit to operate based on implementing Best Available Techniques (BAT) to reduce emissions. Currently IED for agriculture only regulates a small percentage of the overall ammonia emitted into the atmosphere. As previously noted in this Chapter, pig and poultry places are the only livestock that are regulated and much larger emitters like cattle are currently unregulated. Cattle are the main source of ammonia emissions in Europe.

Environmental Assessment Policy

The Environmental Impact Assessment Directive (EIA - 85/337/EEC) also provides an assessment approach to new plans and projects. Under Annex I of the Directive, industrial processes listed include roads, building new power generation stations, and agricultural practices. The thresholds for agriculture almost mirror that of IED and, like that Directive, also only include pig and poultry. The EIA Directive includes a list of project categories that are subject to assessment. Strangely, the thresholds here are higher than that of IED so this Directive always takes precedent - 85,000 places for broilers, 60,000 places for hens, 3,000 places for production pigs (over 30 kg) and 900 places for sows.

Annex II provides more options for assessing impacts from all agricultural practices although Member States can set thresholds. In Denmark, EIA is used to screen all applications for new or extending livestock production units. Screening includes the carrying out of impact assessment of N-deposition to SACs. In the UK, many agricultural activities under EIA are generally not assessed for impacts on Natura 2000. However, agricultural practices under Annex II have recently been used to call for assessments to be carried out in Wales. The Welsh Government has endorsed an approach where pig and poultry units close to a sensitive areas (e.g. Natura 2000 site) but are below the Annex I numbers, and exceed the Annex II threshold limit (in this case "the area of new floorspace exceeds 500 square metres"), are now open to assessment. This opens up the process for much smaller farms requiring EIA assessments, which in turn could help towards capturing the 'in combination' element of the Habitats Directive, where many small contributing farms can add up to give exceedances of critical loads/levels. However, since Member States are allowed to set their own threshold criteria for Annex II, many activities that impact on Natura 200 sites are likely to be overlooked for assessment.

The Strategic Environmental Assessment (SEA) Directive (2001/42/EC) is linked with EIA. SEA focuses on large scale plans or programmes and provides certain conditions where an EIA is required. For example there is a requirement of Member States to inform others of transboundary impacts from a proposed plan. The focus on the SEA Directive is the specification of environmental assessment for large scale plans and programmes. A list of conditions apply that require an EIA under this Directive. This includes the requirement to inform other Member States of possible transboundary impacts of proposed plans or programmes. Categories listed under Annex II include agricultural activities including pig and poultry installations. Cattle and other sources are not specified. SEA is also mandatory for planning and programmes in the view of likely significant effects under the Habitats Directive. SEA offers the potential for assessments of agricultural activities in relation to protecting Natura 2000 sites.

Importantly, SEA are now required as part of a Rural Development Programmes (RDP). This directive therefore has the potential to review the impacts of nitrogen emissions more widely, including both NO_x emissions from roads and NH_3 emissions from agriculture. For example, where a regional plan specifies an area as being targeted for agricultural activities rather than urban or other development, then it could be argued that this choice should be assessed in relation to the protection of the Natura 2000 network.

Agricultural Policy

Cross Compliance (Council Regulation 73/2009 and Commission Regulation 1122/2009) is another significant set of requirements that land owners have to meet in order to receive support payments. Cross compliance is an important element of receiving payments under the Common Agricultural Policy across Europe. It includes two elements:

- 'Statutory Management Requirements' based on current EU legislation cover 18 legislative standards on the environment, food safety, animal and plant health and animal welfare. Of these the Habitats Directive and Birds Directive are included as well as three other environmental legislations: the Nitrates Directive (91/676/EEC), the Directive 80/68/EEC on the protection of groundwater, and the Sewage Sludge Directive (86/278/EEC).
- 'Good agricultural and environmental condition' obligates the farmer to keep land in good agricultural and environmental condition – e.g. soil protection, maintenance of soil organic matter and structure, avoiding the deterioration of habitats, and water management.

Inspections of farms are made each year, but this is often only around 1% of those claiming under the Basic Payment Scheme.

Cross compliance is meant to set a common standard across Europe; namely, that agricultural activities are carried out in a sustainable and legal way. It is formed out of a principal that 'the polluter pays'; in other words, responsibility to maintain the standard is on the famer and non-compliance can result in payments being cancelled. However, it is unclear how effective cross compliance has been in protecting Natura 2000 sites. In principle, it should catch the much smaller farms that are currently unregulated and therefore could be seen as a powerful regulatory instrument.

A large obstruction for implementing ammonia mitigation options has clearly been the costs for the farmer in, for example, updating housing systems or buying new land spreading equipment (although in general the costs involved for manure spreading or covering manure storage are very cost effective compared to other measures). However the recent review of the CAP has paved the way for financial support to farmers for ammonia measures available through its 2nd pillar, the Rural Development Programme (RDP). RDP, with a budget of €85 billion for 2014-2020, has set out objectives for rural development which contribute to the Europe 2020 strategy for smart sustainable growth. Member States will have to build their RDPs based upon at least four of the six Union priorities, one of which (5(d)) sets out reduce greenhouse gas and ammonia emissions from agriculture. The Scottish government, for example, has outlined €23 million for investments and €54 million in Agri-Environment-Climate (AEC) schemes, to reduce greenhouse gases and ammonia emissions. This will allow land managers to apply for annual management costs and capital projects for a wide range of environmental purposes. The AEC scheme will contribute to the objectives of the EU Birds and Habitats Directives and help towards achieving favourable conservation status for Natura 2000 sites.

2.5 Future Policy Options

Tackling the impacts of nitrogen deposition in Natura 2000 sites can be addressed in terms of regulating long range and short range transport of nitrogen.

The regulation of long range transport has been supported with the onset of the UNECE Gothenburg Protocol, and NECD in Europe. Recent reviews of both instruments provide for a reduction in the future ceilings of all nitrogen pollutants by 2020. Reductions based on 2005 for both NO_x and NH_3 show that reductions across Europe, for 2020, range from 20-55% for NO_x (average EU 42%) and 1-25% for NH_3 (average EU 6%). The level of ambition for reductions in NH_3 is rather low given the importance that ammonia has now and in the future for eutrophication, acidification and particulate threats. The potential for further reductions of NH_3 based on a maximum reduction from the RAINS model (MMR) is also the greatest. This highlights that current commitments for reducing NH_3 are the lowest of all pollutants. Accordingly, UNECE and NECD, while achieving general reductions in ammonia emissions, are not closely linked to protecting the Natura 2000 network. Protection from biodiversity loss has not been based on the European Biodiversity Strategy or biodiversity targets. Targets have largely been developed within the air pollution discipline; for example, the Working Group on Effects within CLRTAP was set up in 1980 to research and monitor pollutant effects. And targets for assessing impacts and favourable status have relied heavily on the use of critical loads. While the critical loads approach provides a very useful tool to support air pollution policy, linking them to biodiversity protection is difficult.

International policy instruments do not directly address the problems associated with short range transport to Natura 2000 sites. Actually, hot spots of spatially variable ammonia in the landscape require more local targeted responses. The permitting of pig and poultry farms under IED alone are only, at best, regulating 5% of NH_3 emissions; accordingly, the inclusion of cattle (of a certain herd size) in IED could be a highly effective addition. However a recent review in 2013 left out the inclusion of large cattle farms into IED due to the potential high increase in permits and running costs, resulting in higher meat prices for consumers.

Spatial planning of ammonia sources, the implementation of mitigation options, and the location of polluting activities in respect of Natura 200 sites has a significant role to play in reducing the impacts of nitrogen deposition and concentrations on the Natura 2000 network. Ammonia emissions and subsequent concentrations are closely matched. Predicting hot spot areas is certainly possible even without modelling. For example, planning where to spread manure and install new sheds should be relatively straight forward when taking into account nearby protected sites. Spreading manure or siting new sheds on areas away from sensitive sites can be managed as part of landscape planning with the use of buffer zones around protected sites.

Buffer zones can help to allow greater deposition and dispersion between a source and receptor, while the planting of tree belts in these zones can enhance both these effects considerably. Planting trees around sources can provide suitable reductions in ammonia emissions of around 20% if planted at depths of 50 m or more between source and receptor (see Chapter 4). This strategy of enhancing the deposition of ammonia at the 'farm gate', and not contributing to NECD emission reductions, has the added benefit of reducing wet deposition and country export as more ammonia is deposited locally (see Chapter 4). In England, provisions have been made to promote the planting of trees as part of an Agri-environment scheme to reduce ammonia emissions. Under the Countryside Stewardship scheme, land managers can receive grants for woodland creation to abate ammonia emissions (based on ammonia targeted areas set for certain clusters of protected sites e.g. SAC).

Improving awareness of nitrogen deposition impacts can further be broadened through the Natura 2000 Biogeographical Process. Set up in 2011 as a series of seminars for each of the five biogeographical regions, the aims of the Biogeographical Process are to assist practitioners in managing Natura 2000 to improve conservation status, whilst sharing experiences and best practice. Nitrogen deposition was highlighted at the Atlantic Natura seminar, in December 2012, as a major threat to the conservation status of many habitat types. In response, the 'Nitrogen Deposition and the Nature Directives Workshop - Impacts and Responses: Our shared experiences', was held in December 2013. The workshop built on the established evidence base of nitrogen impacts on biodiversity and the report of Hicks et al (2011).

Finally the important interactive effects between nitrogen deposition and climatic factors should not be overlooked. A changing climate will influence the effects of nitrogen deposition as rising temperatures will increase ammonia emissions. However the science community is only starting to understand the interactive effects of reactive nitrogen and climate change.

2.6 Conclusions

Nitrogen deposition impacts are widespread across Europe and that general awareness of the problem is growing. Agricultural emissions of ammonia are seen as a main pollutant threat to the Natura 2000 network as agricultural activities are often situated adjacent to, or surrounding, protected sites. The favourable conservation status of protected sites will continue to decline unless Member States take a proactive stance in tackling the issue. Furthermore, the EU 2020 Biodiversity Strategy targets will not be met. Current compliance issues exist for Member States as the objectives of nature conservation policy are not being taken into account with that of air pollution and agricultural policies. Greater integration of these polices is essential to reverse critical load exceedance and the unfavourable status of Natura sites. Some Member States have gone far in implementing such integration: the Netherlands the PAS approach demonstrates an excellent approach towards assessing, permitting and managing critical load exceedance. Building IT systems to manage and audit sources can also go a long way towards capturing in-combination effects and ensuring critical loads and levels are not exceeded. Allowing development where there is head room for extra nitrogen deposition is positive from an industry viewpoint, while setting out mandatory mitigation measures and restoration programmes ensures integrity of the site can be preserved. Similarly depositions and concentrations are kept on a downward trend.

While IED will continue to regulate pig and poultry farms, most farming activities are unregulated, with diary and beef farming being an obvious omission. Funding is now available for Member States to roll out mitigation options in the light of CAP reform and ammonia measures being added to RDP. Incorporating spatial planning at both local and regional levels can be used to pinpoint locations of hot spots of ammonia where appropriate measures can be used. Integrating buffer zones and tree belts in the landscape can further be exploited to reduce deposition to Natura sites.

Associated with CAP is the application of cross-compliance: current awareness (and enforcement) is lacking between Member States with respect to farmers and potential adverse effects on Natura sites. Further guidance is required to make the links more robust, and suitable low cost methods of abatement should be promoted. For example, the combined benefits of deploying ammonia mitigation options may be significant since measures like manure injection into the soil can increase nitrogen use efficiency, reduce the need for extra mineral fertilisers, and importantly fulfil the requirements of other legislation like the Nitrates Directive or the Water Framework Directive. Similarly, planting trees as scavengers of ammonia has added benefits of increasing carbon storage and meeting afforestation targets across Member States. If there was a full application of all presently available technical emission control measures (the maximum feasible reduction case - MFR) critical load exceedance could be radically reduced across Europe (Figure 2.5).

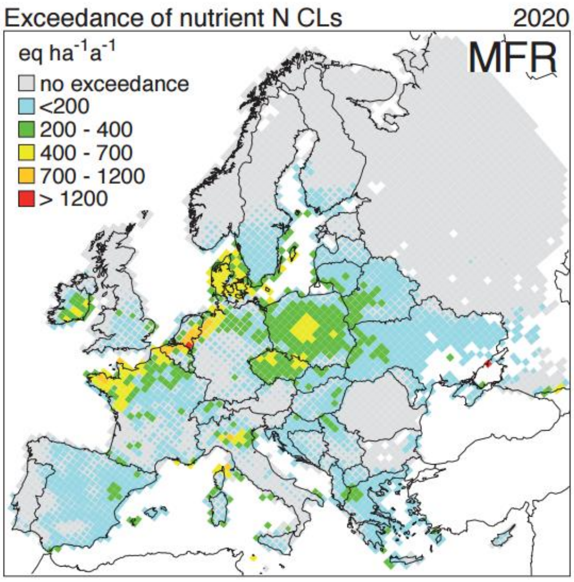


Figure 2.5 Critical load exceedance for nitrogen depositions under the Maximum Feasible Reduction (MFR) emission scenario

Finally, awareness gathering workshops organised through networks like the Natura 2000 Biogeographical Process are highly welcome. Exploring future policy development through shared experiences is important for managing the nitrogen deposition threat across the Natura 2000 network.

The next chapters focus on the potential of planting tree belts around agricultural hot-spots to mitigate against ammonia. It estimates the ammonia reduction potential of planting trees around livestock buildings and manure storage, together with the implementation of a silvopastoral system of livestock grazing under trees.

Chapter 3. Source Attribution of Eutrophying and Acidifying Pollutants on the Natura 2000 Network in the UK

3.1 Introduction

Oxides of sulphur and nitrogen emitted out of stacks and exhausts of combustion sources, and ammonia emitted from animal housing can have negative effects on ecosystems when they are eventually deposited onto habitats. Effects of eutrophication and acidification cause species composition changes, soil acidification and loss of biodiversity (Bobbink *et al.*, 2010; Stevens *et al.*, 2006).

Current EU legislation to protect European sites relies heavily on the Habitats Directive (Council Directive 92/43/EEC) and the Industrial Emissions Directive (IED). The provisions of the Habitats Directive require Member States to take measures to maintain or restore at favourable conservation status the natural habitats and species of Community importance. Article 6.3 of the directive provides a mechanism by which plans and projects can only be permitted if they are shown to have no adverse effect on the integrity of a European site (e.g. Special Area of Conservation - SAC). Under the regulations which implement the IED Directive, the regulators are required to take account of the Habitats Directive when issuing permits to industrial processes covered by the regime.

Potentially, emissions from industrial processes may exert a range of different types of pollutant impact on European sites over short (<5 km) or long ranges (>100 km). The combustion of fossil fuels, in particular, may release large quantities of sulphur and oxidised nitrogen through tall stacks which can travel long distances before they are deposited in dry or wet form (precipitation) to vegetation surfaces. This means that habitats can be impacted by sources located hundreds of kilometres away. Meanwhile, emissions of highly soluble gases like ammonia originating from agricultural activities constitute major local sources and travel much shorter distances (<2 km) before depositing. Understanding the relative contribution and the distance of transport from these different sources can help regulators and policy makers decide on what and where remedial action may need to be taken to control and mitigate against releases from these activities.

To investigate these varying influences from sources on European sites a method of source attribution (or source apportionment) is applied where the pollutant 'footprints' of many individual sources can be compiled to provide a pollutant-receptor matrix quantifying the contribution of a source or sector to deposition at each site. Source attribution is often produced by using dispersion models and driven by emission inventories (Viana *et al.*, 2008). In this study a Lagrangian dispersion model was used to produce source attribution footprints for over a hundred different sources.

Similarly, effects-based risk assessment tools like the 'critical loads and levels' concept (see Chapter 1) have long been used to assess the health of ecosystems and help in the development of pollutant abatement strategies. By linking habitats at a particular site to a relevant critical load it is possible to assess the status of a sites 'health' by comparing site specific critical loads with pollutant deposition.

Combing the source apportionment principles with site specific critical loads produces a useful policy tool showing the key sources that are contributing to critical load exceedance at Natura 2000 sites.

3.2 Methodology

The FRAME (*Fine Resolution Multi-pollutant Exchange*) model, incorporating emission point sources and sectors, was used to provide source footprints of nitrogen and sulphur deposition across the UK (Vieno et al, 2009). The modelling was carried out for a current year 2012. A detailed description of FRAME is provided in Chapter 1.

3.2.1 Modelling

FRAME is a Lagrangian atmospheric transport model used to assess the long-term annual mean deposition of reduced and oxidised nitrogen and sulphur over the United Kingdom.

The model has been applied to assess environmental impacts and their changes (Matejko et al., 2009 ; Dore et al., 2012) as well as the impact of policy to reduce pollutant emissions (Dore et al., 2007) and to generate source-receptor relationships for use in integrated assessment modelling. (Oxley et al., 2013). A detailed description of the FRAME model is contained in Singles *et al.* (1998).

FRAME was used to estimate the contribution to deposition of sulphur and nitrogen across the United Kingdom for emissions data from the UK National Atmospheric Emissions Inventory (NAEI - http://naei.defra.gov.uk) (see Table 3.1), split into 160 different sub-sectoral emission categories. This included 22 individual point sources and background 'area' emissions of SO_2, NO_x and NH_3 split into 11 SNAP sectors (Selected Nomenclature for Air Pollution, European Environment Agency, 2013), international shipping and European emissions. The 11 SNAP sectors are - energy production and transformation; commercial institutional and residential combustion; industrial combustion; industrial processes; solvent use; road transport; other transport; waste treatment and disposal; agriculture; and natural. The top 22 point sources were isolated as they are of interest to regulators. They were made up of power stations (12), refineries (5), steel works (3), auto generators (1), and other industrial combustion (1). The input of ammonia emissions to the model used the National Ammonia Reduction Strategy Evaluation System (NARSES, Misslebrook *et al.*, 2010) model split into 5 sectors – livestock; fertiliser; non-agricultural abatable sources (e.g. remaining point sources, transport, solvents, industry, power generation); non-agricultural non-abatable sources (wild birds and animals, pets, non-agricultural horses, all human-related emissions); and non-agricultural sources from the waste sector (including anaerobic digesters). Regional land masks were applied to the emission maps to separate sources by country - England, Scotland, Wales and Northern Ireland - to give the devolved government regulators a clearer idea of the source location of the pollutant depositing on each protected site.

Table 3.1. Emissions scenarios for the FRAME runs

Type	Pollutants	Emission Year	Country	SNAPs	Runs
Top 22 Point Sources	SO_x, NO_x	2012	1 (UK)	-	44
Point Sources	SO_x, NO_x	2012	4	11	88
Point Sources	NH_x	2012	4	5	20
Industrial Point Source	NH_x	2012	1 (UK)	1	1
Offshore	SO_x, NO_x	2012	1 (UK)	1	2
International Shipping	SO_x, NO_x	2011	-	1	2
European Import	SO_x, NO_x, NH_x	2012	-	-	3
				TOTAL	**160**

Running the model

In order to create individual footprints the FRAME model was run for a base scenario (all sources), then run again abating each source one by one. A 25% emissions reduction was applied to each emissions source to reduce the influence of non-linearities in atmospheric chemical reactions which could occur if a single source was completely removed. This is in line with the approach adopted at a European scale for the generation of source-receptor data for policy applications with the EMEP model (http://www.emep.int). The final footprints were then scaled by a factor of 4. By running the model successively and abating individual emission sources at each run, the difference between the source and the base run is that source's footprint

The footprints of dry and wet N and S deposition corresponding to each source were calculated according to equation (1) by the difference between the deposition maps for the baseline simulation (id=0) and the simulation with the emissions removed (id=n):

FP(id=n) = DEP(id=0) – DEP(id=n) ……………….. (1)

Where:
FP is the footprint
DEP is the FRAME modelled deposition data,
id=0 is the baseline simulation.

For each 5 km grid square across the domain, pollutant compounds for SO_x, NO_y and NH_x deposition were calculated for all footprints. In addition the output deposition data was split into more detailed chemical species to provide an approximation of how much of each ʻsource attribution type (e.g. livestock, fertiliser, shipping, etc.) is a short or long range input. The short/long range split was made as follows:

1. Wet NH_3 deposition (short range)
2. Wet NH_4^+ deposition (long range)
3. Dry NH_3 deposition (short range)
4. Dry NH_4^+ deposition (long range)
5. Wet HNO_3 deposition (long range) + Wet NO_3^- deposition (long range)
6. Dry NO_2 deposition (short range)[1]
7. Dry HNO_3 deposition (long range) + Dry NO_3^- aerosol deposition (long range)
8. Dry SO_2 (short range)
9. Dry SO_4 aerosol (long range)
10. Wet SO_2 (short range)
11. Wet SO_4 (long range)

[1] there is no Wet NO_2 deposition (short range) as it is insoluble

In addition deposition data was output to three different ecosystem types – forest, moorland (representing short semi-natural vegetation) and a grid average (an average of arable, grassland, urban, forest and moorland). It is important to have this distinction as deposition varies depending on the habitat's surface roughness. Forests having a higher surface roughness and thereby larger depositions. In particular dry deposition rates for ammonia are much higher to forests and to moorland than they are for typical grid-averaged rates due to the low canopy resistance for these vegetation types. By contrast the deposition velocity of ammonia to agricultural fields is very low (Flechard et al., 2011)

Post Processing and Calibration

Firstly, the individual source deposition footprints were normalised such that their sum is the same as that from the baseline simulation. This will account for any model non-linearities that would otherwise lead to errors in the source attribution methodology. Analysis of the source attribution data however showed that the difference between the sum of the footprints and the baseline simulation was relatively small with an average of 4% depending on the deposition component.

Secondly, where FRAME deposition data are to be used for calculations of critical loads exceedance, a standard technique is to apply a calibration procedure. This approach is based on the convention that the official data set of mapped deposition of nitrogen and sulphur (CBED - Concentration Based Estimated Deposition, APIS 2015) for the United Kingdom is obtained from measurements of precipitation concentrations and gas concentrations which are interpolated across the country and combined with deposition velocity estimates and maps of annual rainfall for the UK. The CBED data set (Concentration Based Estimated Deposition) is averaged over the three-year period 2011-2013. This provides a more robust estimate of typical annual average deposition than for a single year due to the influence of variable annual meteorology (precipitation and general circulation) on annual average deposition for a single year (Kryza et al., 2012). For this work, the calibration procedure used is described in equation (2):

DEP(CAL,2012) = DEP(UNC,2012) * (DEP(CBED,2011-2013)/DEP(UNC,2012)) …………….. (2)

where:

DEP(UNC,2012) refers to uncalibrated FRAME deposition data for the emissions simulation year 2012,

DEP(CBED,2011-2013) is the CBED deposition data for the period 2011-2013 and

DEP(CAL,2012) is the calibrated deposition for the year 2012.

The use of the calibration procedure can also be applied to the footprint data generated according to equation (1). For calibration of footprints, the formula applied is described in equation (3).

FP(CAL,2012,id=n) = FP(UNC,2012,id=n) * DEP(CAL,2012) / Σ FP(UNC,2012,id=1,160) ……….. (3)

Where Σ FP(UNC,2012,id=1,160) corresponds to the sum of the uncalibrated footprints. This calibration procedure ensures that the calibrated footprints, when combined, will generate the official CBED deposition totals for the year 2012.

3.2.2 Aggregation of output and calculating critical load exceedance

Using a data processing software package (FME), the footprint deposition files were superimposed over the Natura 2000 network boundaries for Special Areas of Conservation (SAC) in conjunction with the relevant critical loads for the designated features at each site. Since processing and presenting 160 individual source footprints is difficult, further aggregation of the source footprint were made. Regional footprints were aggregated to 38 UK sectors, for example Livestock for England, Scotland, Wales, and Northern Ireland became Livestock UK. The 22 point sources were also aggregated to one ‘point source’ contribution for the UK.

Exceedance statistics per site were calculated by comparing the most sensitive habitat (lowest critical load) with the total deposition at that site. The total deposition was based on the ecosystem type used in the model. For example if the most sensitive habitat was a forest or woodland then the modelled forest deposition dataset was used to compare critical loads.

The resulting matrix of source attribution by sector and exceedance statistics for each site, provide a status of the whole UK's Natura 2000 SAC network.

3.3 Results

This section will present the results in four sections. The first section will look at some example footprints to show the spatial distribution of deposition from sources varies for both nitrogen and sulphur. National statistics on the different pollutant species are also presented showing the split between short range and long range sources, wet and dry, and for nitrogen reduced and oxidised. The second section looks at the dominant sources across the Natura 2000 network. The third section examines the level of exceedance across the network. The forth section reviews sample case studies focusing on particular SAC sites which demonstrate the variability across the UK of different source characteristics.

3.3.1 Footprints

Figure 3.1 to Figure 3.8 show example source footprints for both nitrogen and sulphur deposition. Figure 3.1and Figure 3.2 footprints show the dry and wet components of nitrogen deposition for Livestock emissions for England. It is clear to see the difference in spatial distribution between dry and wet. Dry deposition of ammonia deposits locally within a few kilometres the sources and this is shown by the fact that deposition is strictly confined to England. While wet deposition of ammonium (in rainfall) travels further, principally as particulate ammonium, and other countries like large parts of Wales and Southern Scotland are affected. Shipping emissions of sulphur and nitrogen compounds, and their deposition footprint profiles, are shown in Figure 3.3 and Figure 3.4. Figure 3.3, shows the deposition of sulphur clearly indicating the shipping lanes around the coast of the UK (English Channel and up the Atlantic coast to the Western Isles of Scotland). Deposition of NO_x (Figure 3.4) shows a more uniform distribution across the country when compared with sulphur deposition, which is due to a higher component of wet deposition over dry. Figure 3.5 and Figure 3.6 show footprints of a single point source showing dry and wet deposition of nitrogen (NO_x). Dry deposition is not confined to the immediate few kilometres of the stack, but are still depositing 200 to 300 km from the source - although the amounts deposited are very small. However long range wet deposition deposits much further crossing into other countries falling on upland areas of high rainfall. Transport sources are shown in Figure 3.7 and Figure 3.8 with 'road transport' (e.g. buses, cars, HGVs, LGVs) emissions depositing close to road networks and built up areas of cities and towns. 'Other transport' consists of rail links and airports (take-off & landing), and in-port traffic and berthing (e.g. ferries and naval). In Figure 3.8 railway lines can clearly be seen represented by long chains of deposition in the south west of England.

The footprints demonstrate the long and short range nature of both nitrogen and sulphur, and reinforces the issues that Natura 2000 sites have from air pollution. It further validates the requirement for local, national, and international approaches to nitrogen and sulphur pollution.

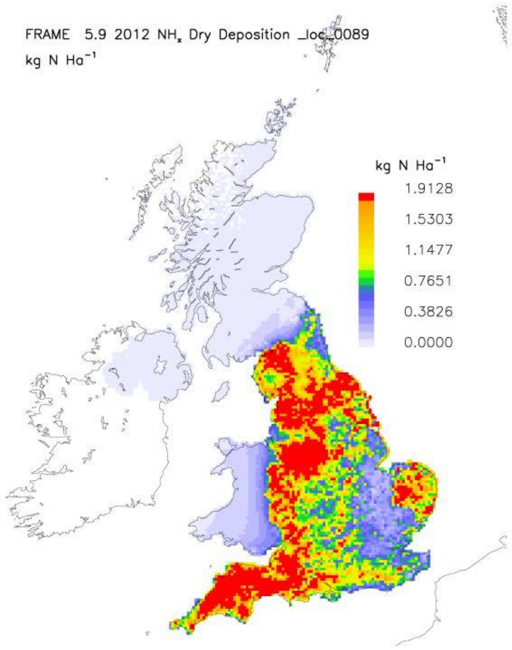


Figure 3.1 Livestock NH_x dry deposition England (kg N ha^{-1})

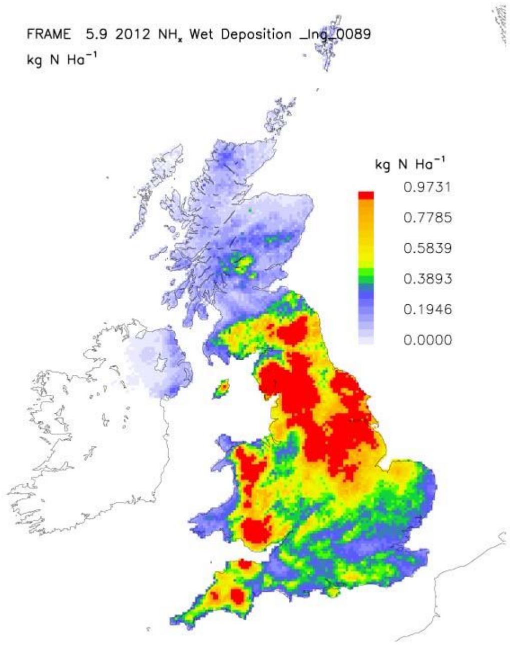


Figure 3.2 Livestock NH_x wet deposition England (kg N ha^{-1})

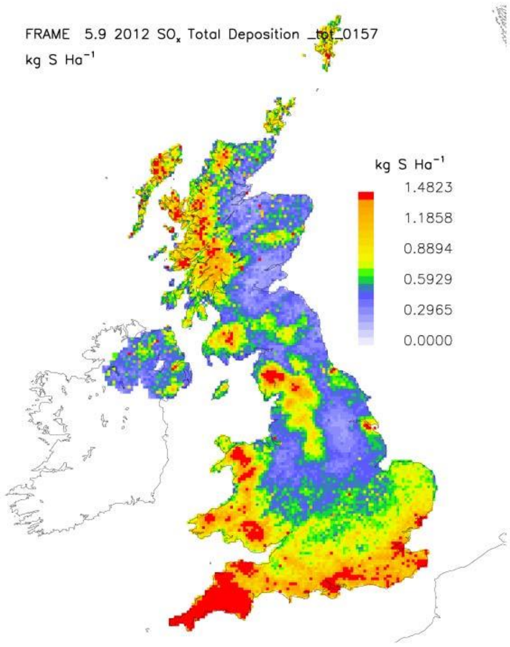


Figure 3.3 Shipping SO_x total deposition UK (kg S ha^{-1})

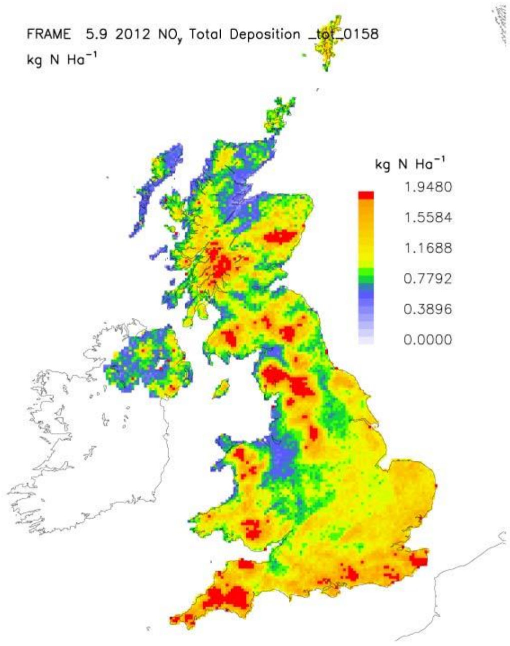


Figure 3.4 Shipping NO_y total deposition UK (kg N ha^{-1})

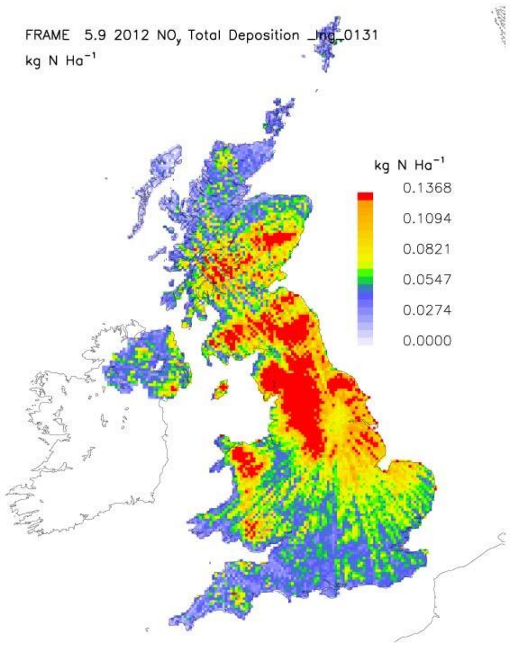


Figure 3.5 Drax coal fired power station NO_x total long range deposition (kg N ha^{-1})

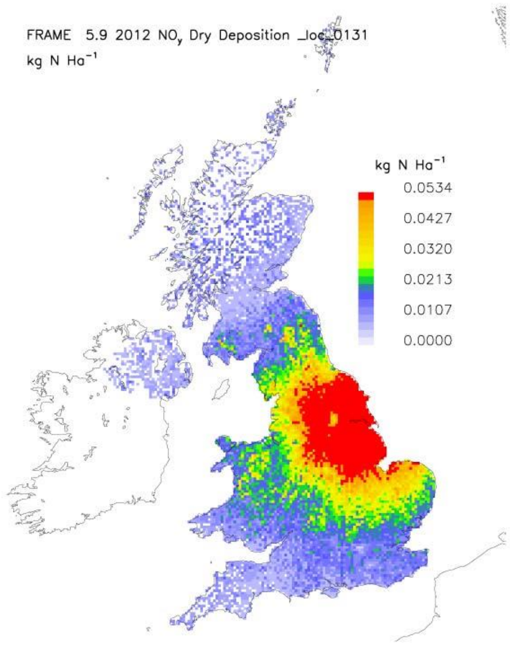


Figure 3.6 Drax coal fired power station NO_x dry deposition (kg N ha^{-1})

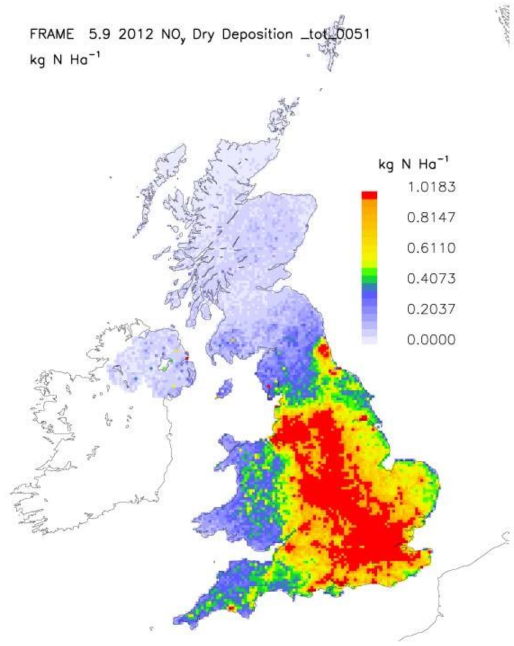


Figure 3.7 Road transport NO_x dry deposition England (kg N ha^{-1})

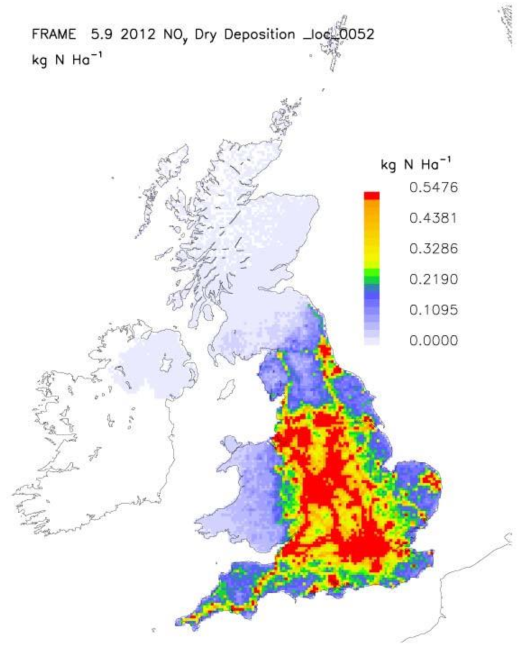


Figure 3.8 Other transport NO_x dry deposition England (kg N ha^{-1})

National pollutant share across the Natura 2000 network

The status of nitrogen and sulphur deposition across the UK is shown below. In the UK, nitrogen deposition to the Natura network comes predominantly from livestock practices (32%) (Figure 3.9) reflecting the fact that SACs are often surrounded by agricultural land and practices. Import from Europe is equally split (9% and 10% between reduced nitrogen (NH_x) and oxidised nitrogen (NO_x). This is interesting as it shows that agricultural practices in Europe are having a similar effect as to combustion sources, which have often been the focus for long range transport.

Of the total nitrogen that is deposited on Natura sites, nearly three quarters (73%) comes from the reduced form of nitrogen (Figure 3.10), with dry deposition being the main fraction (48%) highlighting the contribution from livestock shown in Figure 3.9. The long range contribution of NO_x is 19% indicating that long range transport from combustion sources is still relevant although only a quarter of reduced forms.

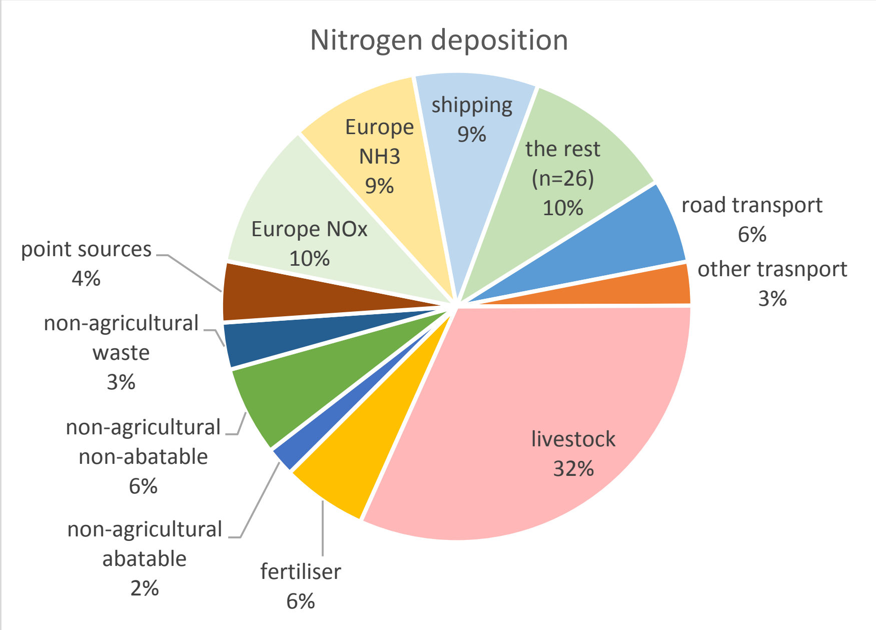


Figure 3.9. Nitrogen deposition source attribution across the whole Natura 2000 network (SAC) showing livestock as the largest contributor to nitrogen deposition (32%). 'The rest' are 26 other source contribute individually less than 5%.

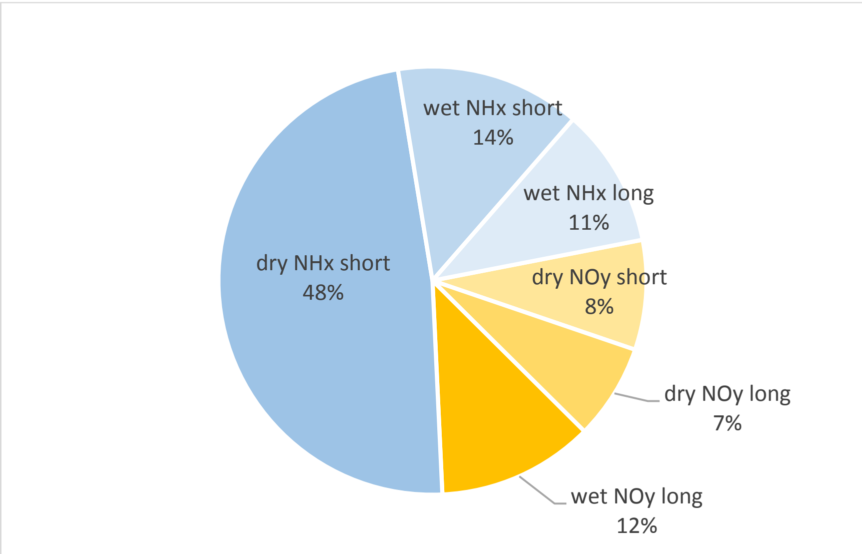


Figure 3.10. nitrogen deposition across the Natura 2000 SACs showing separation of short and long range pollutants for NH_x and NO_y

For sulphur deposition the distribution is shared between five main source-sectors – point sources, Europe import, shipping, commercial & residential and industrial (Figure 3.11). Point sources contribute over a third of the deposition to SACs (37%) with shipping and Europe being the other main sulphur sources. Notably, the majority of long range sulphur deposition (60%) is in the form of wet deposition highlighting the importance of long range transport (Figure 3.12).

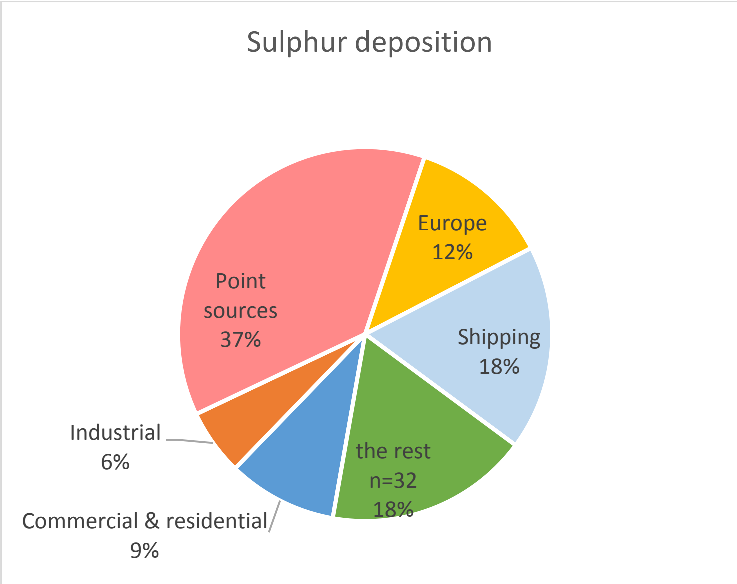


Figure 3.11. Sulphur deposition source attribution across the whole
Natura 2000 network (SAC) 'The rest' are 26 other source contributing individually less than 5%.

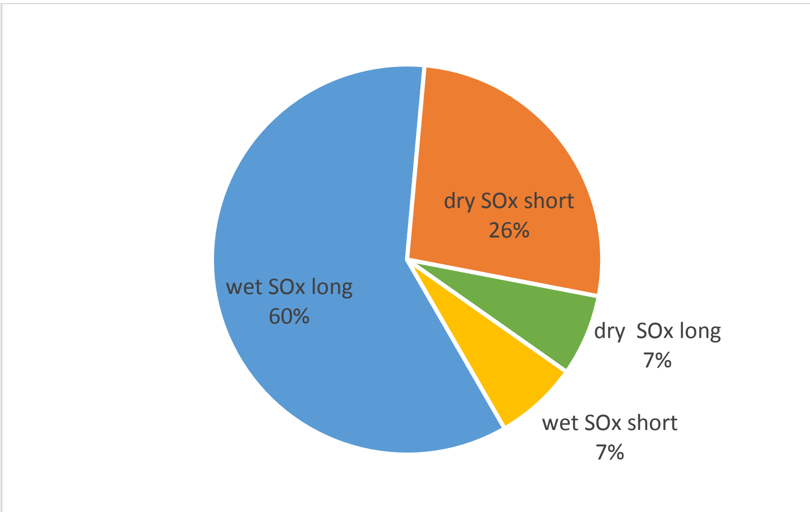


Figure 3.12. Sulphur deposition across the Natura 2000 SACs showing separation of wet/dry and short/long range elements

3.3.2 Dominant sources across the Natura 2000 network

Nitrogen

In the previous section livestock emissions were shown to contribute the most deposition to the Natura network (33% contribution). However it is useful to gauge exactly how many sites have livestock as their most dominant source, and how large the contribution is. Livestock is overwhelmingly the dominant source at 564 of the 631 Natura SAC sites (Figure 3.13). This represents 89% of the total SACs across the UK (Figure 3.14). The percentage contribution for livestock ranges from 14-54% of the total. The next most dominant source is emissions of ammonia from Europe (29 sites, 5% of SACs). There were only three sites where the dominant source is road transport and these were in urban areas (e.g. London). This is somewhat surprising as there are many SAC next to or surrounded by busy road networks. This perhaps shows the close proximity of livestock activities near to Natura sites. Although NO_2 concentrations may be high near roads NO_2 has a relatively low deposition velocity to natural ecosystems in comparison to NH_3 and HNO_3, and that NO_x is emitted primarily as NO in the model (which is not dry deposited) although NO is rapidly oxidised to NO_2

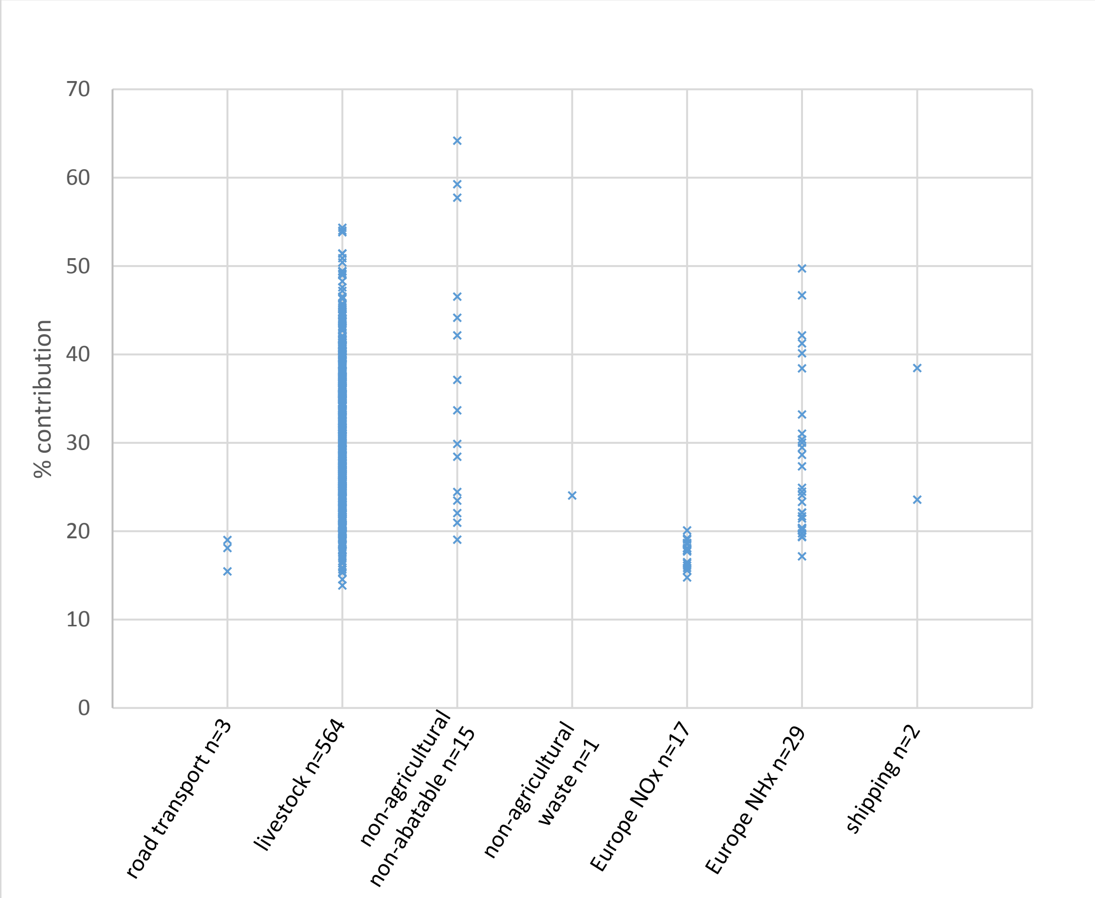


Figure 3.13. % contribution per site for most dominant sources - total nitrogen deposition Natura 2000 (SAC) n=631

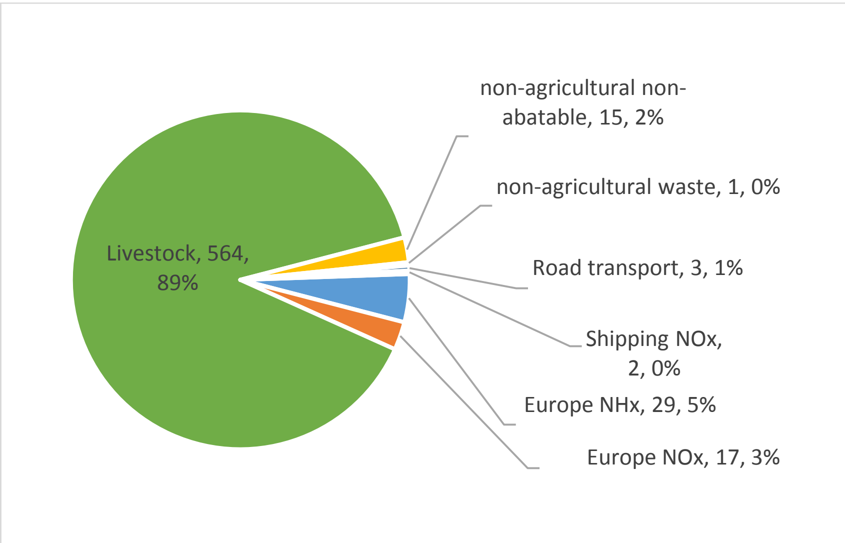


Figure 3.14. Most dominant source sector contributions of nitrogen deposition to Natura 2000 SAC n=631. Displayed as the number of sites and as a % of the total number of sites (631).

SAC maps of dominant sources are shown in Figure 3.15 to Figure 3.18 demonstrate the extent and spatial variability of dominant sources. Figure 3.15 and Figure 3.16 show the difference between deposition based on the grid average calculated as an average of several land classes (forest, moorland, arable, grassland, urban), and deposition to 'moorland' or short vegetation. Differences in the 'moorland' map are at coastal sites, where shipping and import of ammonia from Europe are less dominant than livestock. Upland sites in the Scottish Highlands are influenced more by European import of NO_x. Notably natural sources of ammonia dominate some coastal and island sites (see legend 'Non-agricultural non abatable'). These are large sea-bird colonies sites where emissions of ammonia can be substantial - in some cases more than 100 t NH_3 per colony per year (Blackall *et al.*, 2007). Figure 3.17 and Figure 3.18 show the dominant nitrogen sources split between short and long range transport. Short range sources are similar to the total deposition maps (Figure 3.15) where livestock is still the dominant source. However, long range transport of nitrogen is driven by NO_x, in particular from Europe import and shipping.

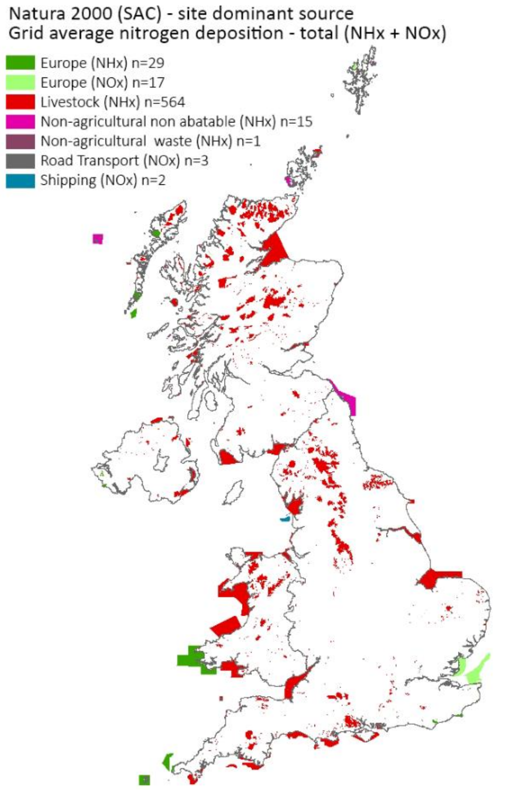


Figure 3.15. Dominant source at each site for total nitrogen deposition (grid average)

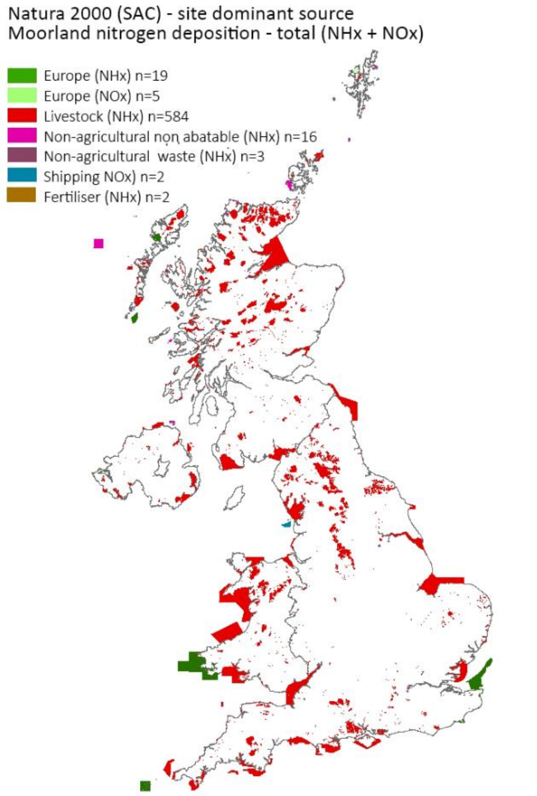


Figure 3.16. Dominant source at each site for total nitrogen deposition (moorland)

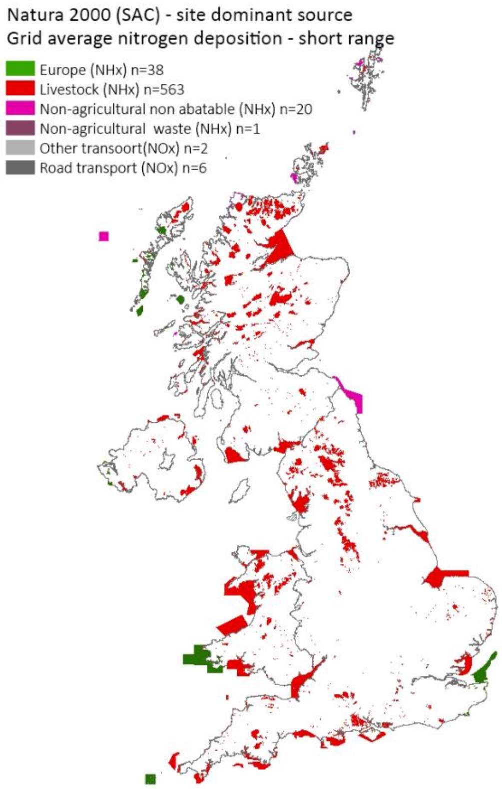


Figure 3.17. Dominant source at each site for short range nitrogen deposition (grid average)

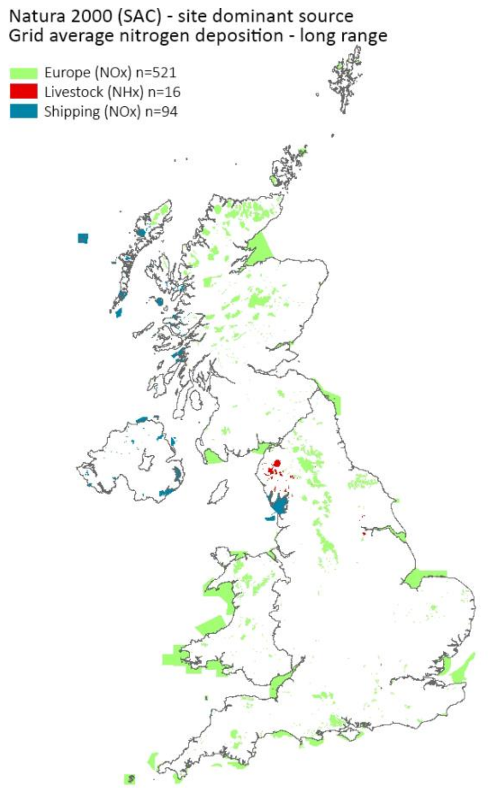


Figure 3.18. Dominant source at each site for long range nitrogen deposition (grid average)

Sulphur

There are three main sulphur sources that affect Natura 2000 SAC sites – point sources, shipping and European Import. From Figure 3.19 point sources are the most dominant source for sulphur deposition at 482 sites contributing up to 70% at one particular site. Shipping is the next most dominant source at 119 sites while European import is most dominant at 30 sites.

Figure 3.20 to Figure 3.22 show the locations across the Natura network of dominant sources. Shipping is most dominant along the English Channel coast and up the Atlantic coast in the west. Europe import only dominates in parts of Northern Ireland along the border with the Republic of Ireland, and in the Shetland Isles. The long and short range split between sulphur deposition shows little change in the overall picture. In the north of Scotland 'other transport' are the dominant short range source which may indicate inshore shipping and ferry emissions. These sites change to a dominant source of international shipping for the long range sulphur map (Figure 3.22).

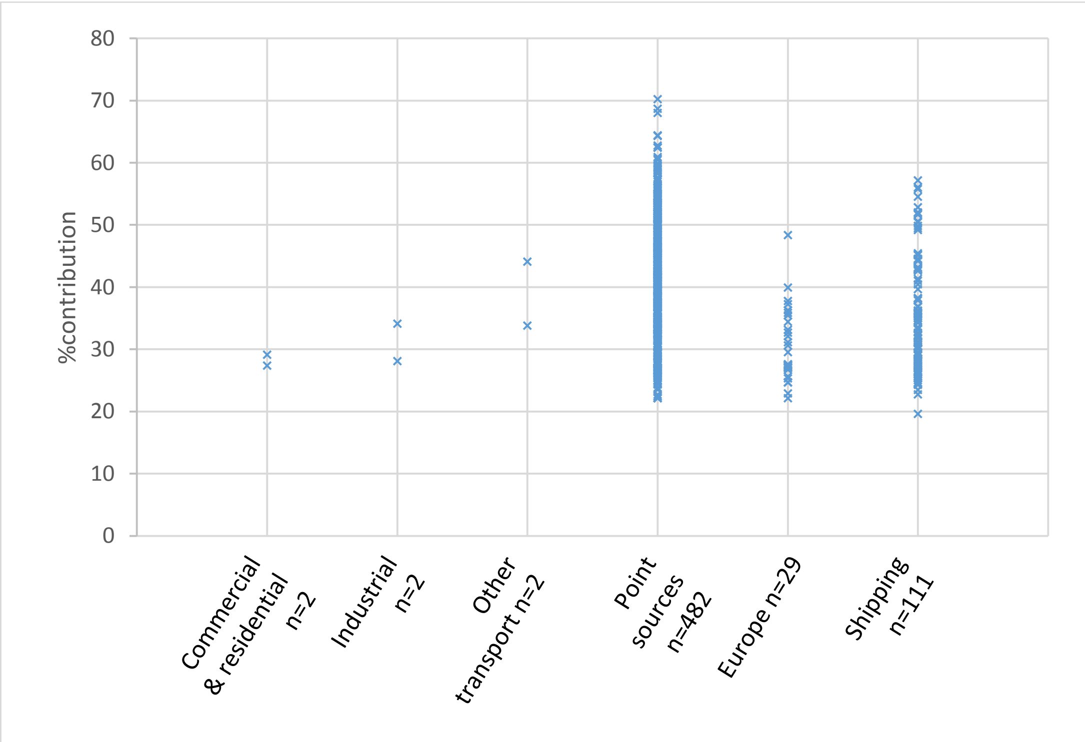


Figure 3.19. % contribution per site of dominant sourced - total sulphur deposition Natura 2000 (SAC) n=628

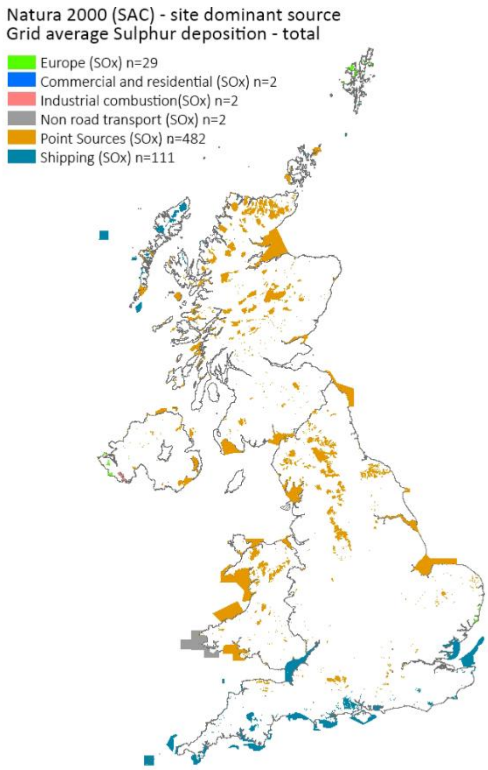


Figure 3.20. Dominant source at each site for total sulphur deposition (grid average)

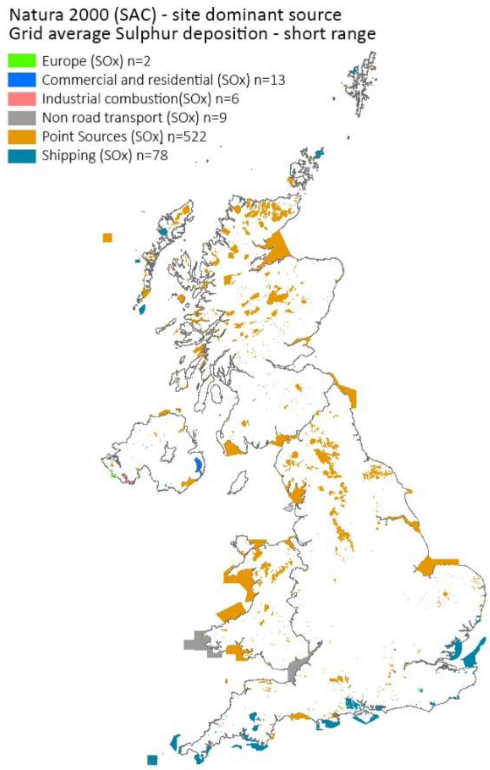


Figure 3.21. Dominant source at each site for short range sulphur deposition (grid average)

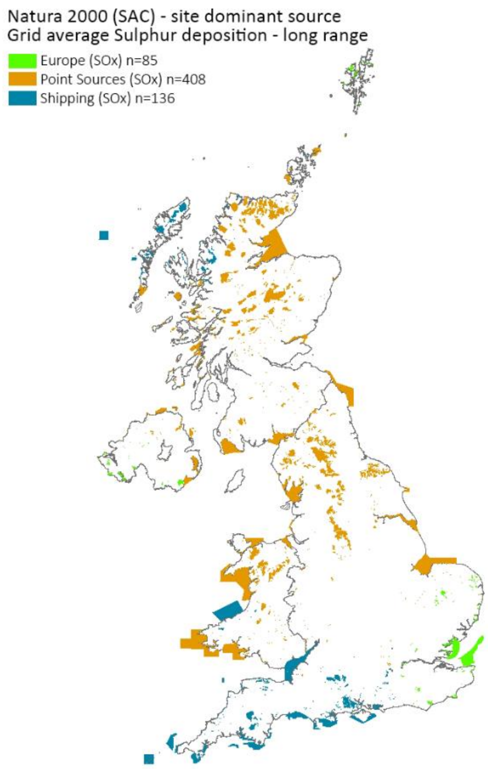


Figure 3.22. Dominant source at each site for long range sulphur deposition (grid average)

3.3.3 Critical Load Exceedance

Exceedance statistics were calculated for the whole network comparing the source-sector deposition with the critical load of the most sensitive Annex 1 feature at each site showing minimum and maximum critical load exceedances (Table 3.2). Exceedance of the critical loads for nutrient nitrogen and acidity is high across the Natura network. Over 75% of the network exceeds the lower empirical critical load for nitrogen, 56% for the upper empirical critical load, and 51% of the sites exceed acidity critical loads. In terms of area, this represents 74% and 66% of the total network in hectares (assuming the sensitive habitat is found across the whole site).

Table 3.2. Exceedance statistics for the UK Natura 2000 SAC network of nutrient nitrogen and acidity critical loads.

SACs sites/area	*Exceeds minimum CL(N)	*Exceeds maximum CL(N)	*Exceeds CL for Acidity
n=632 sites	n=478 (76%)	n=352 (56%)	n=325 (51%)
2,893,985 ha	2,132,900 ha (74%)	1,896,303 (66%)	1,731,819 ha (60%)

*Exceedance is based on the most sensitive Annex 1 habitat feature at any site. For area statistics it is assumed that the habitat is present across the whole site. Not all sites have sensitive features.

Figure 3.23 and Figure 3.24 shows locations of site exceedance which includes the amount of nitrogen (in kg N/ha/yr) above the critical load. Many sites in the upland areas of the UK, notably the Pennine hills and Lake District (red) are 50 kg over the critical load. This is due to a combination of high deposition and sensitive habitats in these areas. Most sites fall into the 15-25 kg over the critical load (orange). Sites that are not exceeded (green) are found in northern Scotland where deposition is relatively low and coastal sites which have less sensitive habitats (higher critical load values). Sites marked grey either have no sensitive habitats to nitrogen deposition or no comparable critical load could be assigned.

Comparing the maximum critical load (Figure 3.24) shows significant change in the amount of exceedance, and some sites, particularly in Scotland, are no longer non-exceeded.

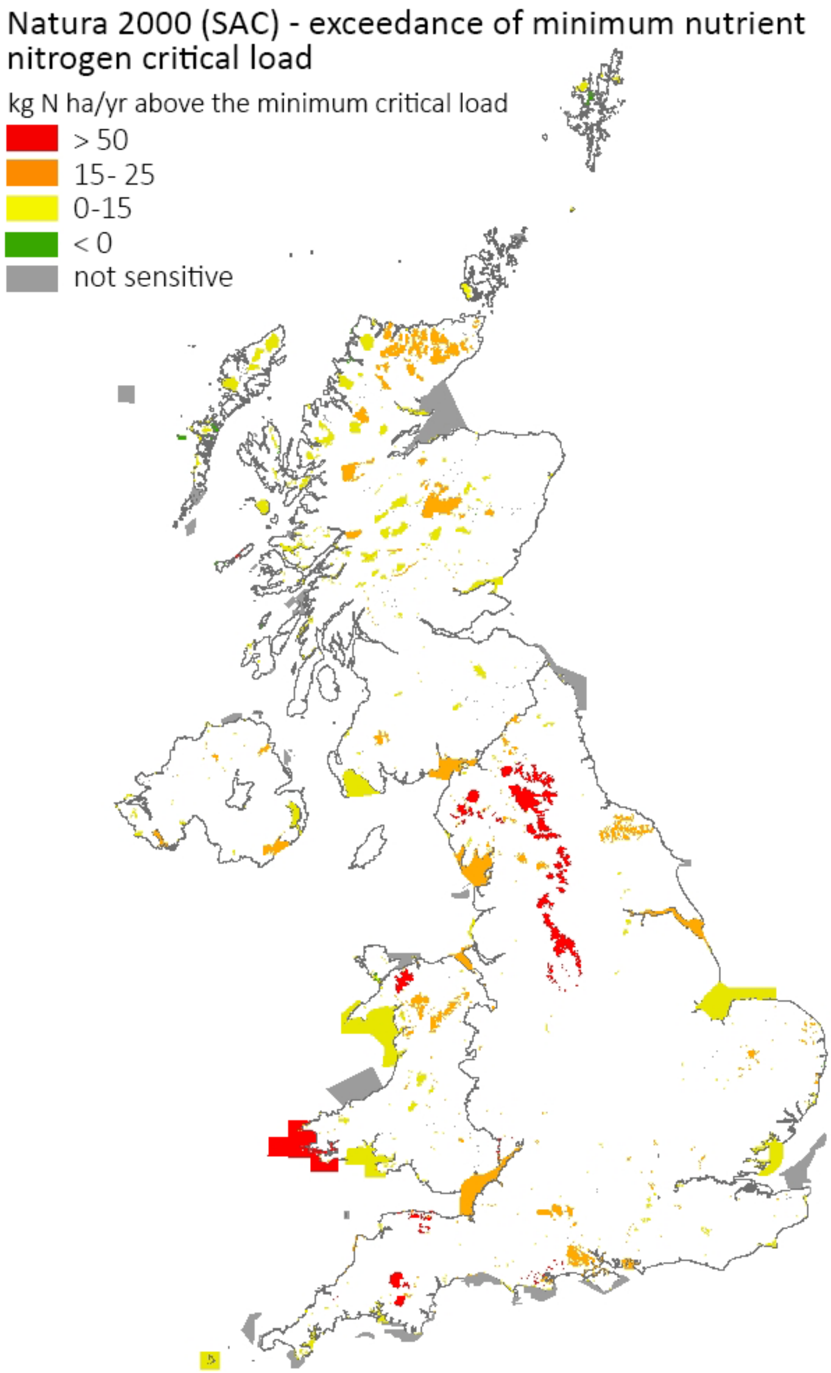


Figure 3.23. SAC site exceedance using the minimum empirical nutrient nitrogen critical load based on the most sensitive Annex 1 habitat. The legend shows how large the exceedances is above the critical load at each site.

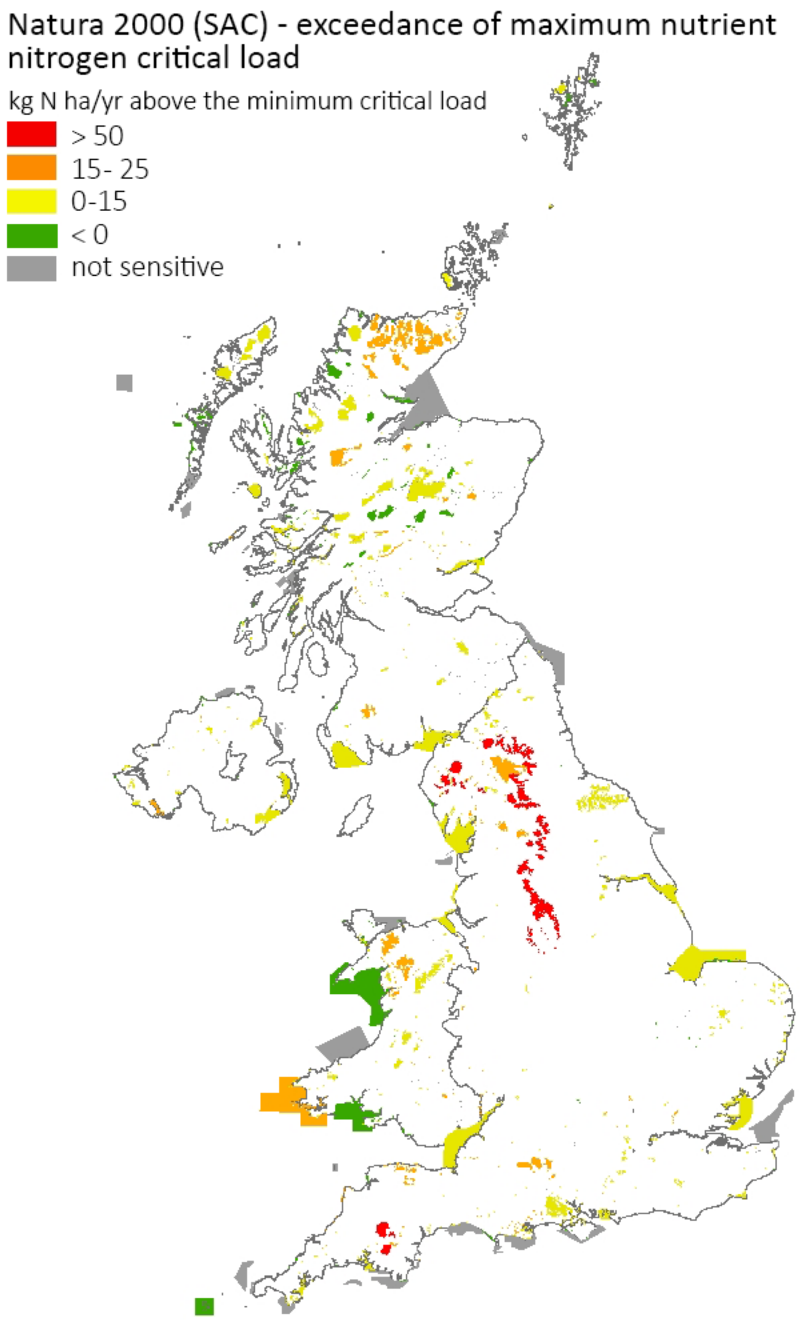


Figure 3.24. SAC site exceedance using the maximum empirical nutrient nitrogen critical load based on the most sensitive Annex 1 habitat. The legend shows how large the exceedances is above the critical load at each site.

Figure 3.25 show the exceedance of acidity critical loads. Parts of the far north of Scotland have sites which are not exceeded together with sites on calcareous soils in southern England.

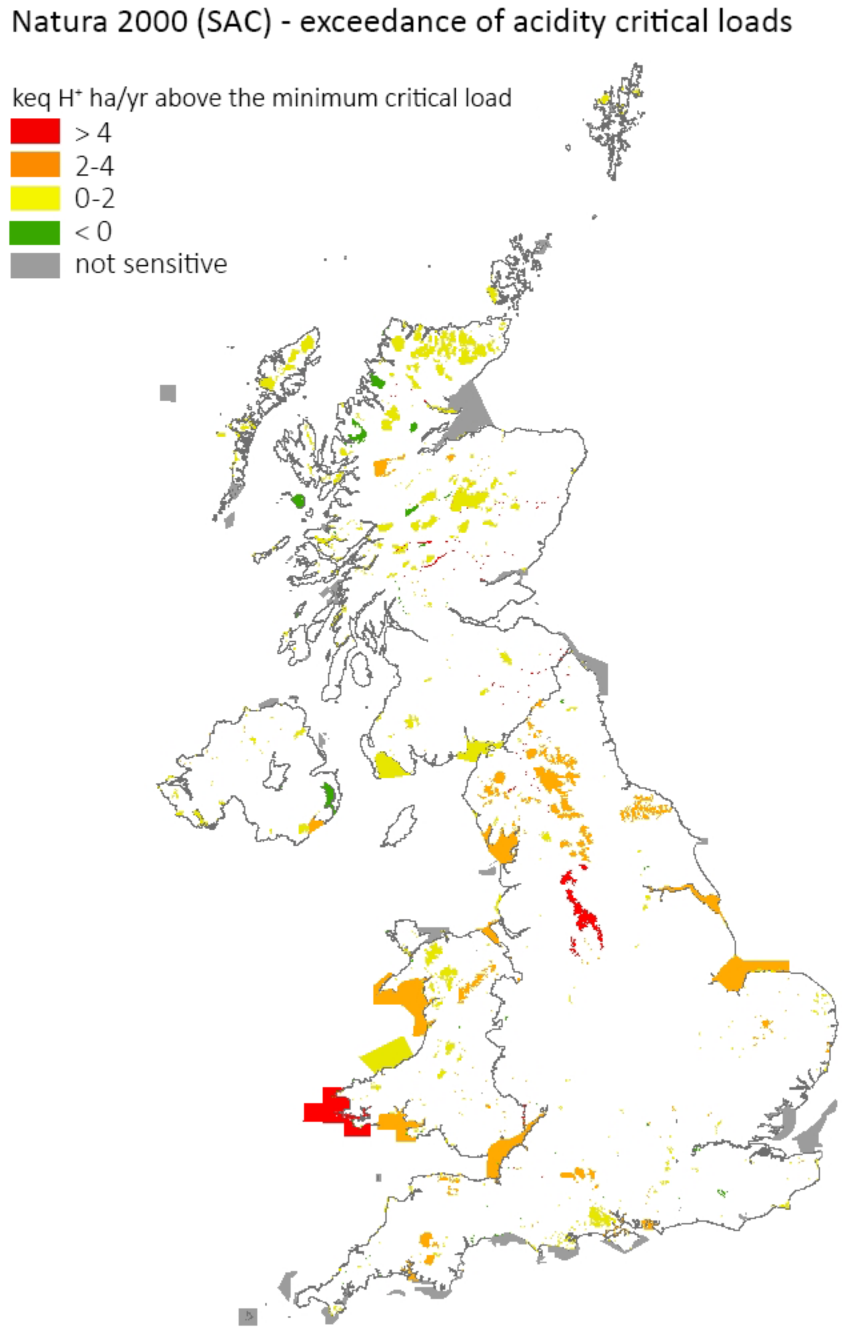


Figure 3.25. SAC site exceedance of acidity critical load based on the most sensitive Annex 1 habitat. The legend shows how large the exceedances is above the critical load at each site.

3.3.4 Source Attribution Case Study Sites

This section presents a selection of sites with various source compositions highlighting the difficult task decision makers (regulators and policy makers) have in reducing exceedance over the Natura 2000 network. Four sites are appraised for their nitrogen deposition sources and exceedances. Two sites are appraised for sulphur deposition representing short and long range effects.

Keen of Hamar SAC

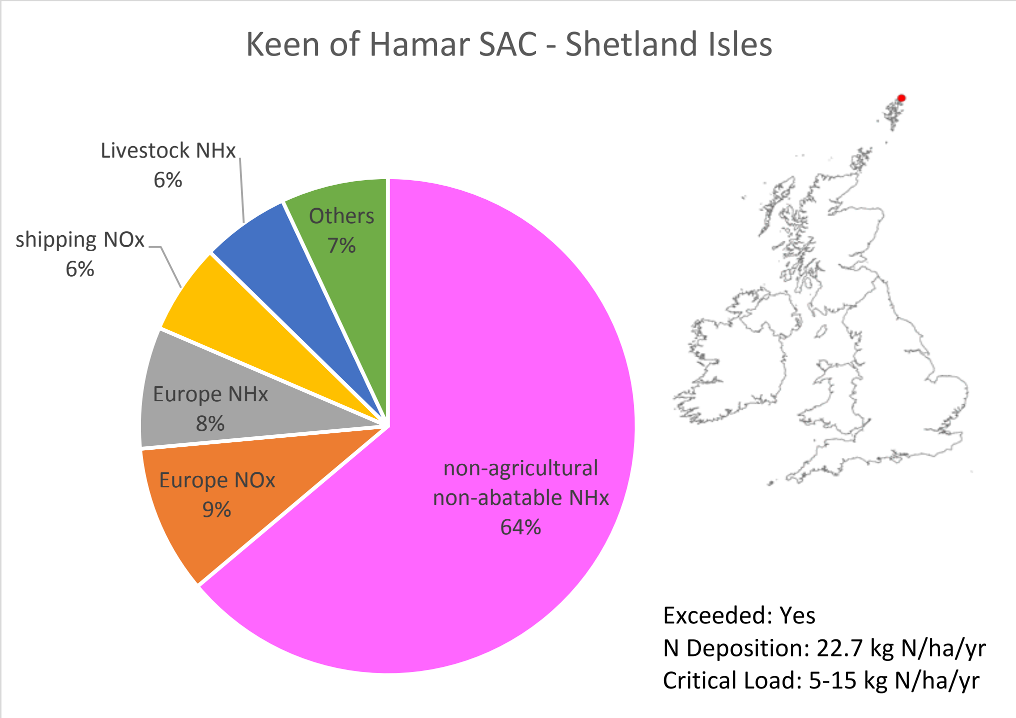


Figure 3.26. Keen of Hamar SAC showing non-agricultural non abatable ammonia as the largest source coming from seabird colonies.

Keen of Hamar in the far north of the UK is made up of shingle, sea cliffs, heath, dry grassland and screes habitats. It has the largest surviving area in the UK of near-natural Calaminarian grasslands on serpentine and is listed for Annex I habitats - *6130 Calaminarian grasslands of the Violetalia calaminariae*, and *8120 Calcareous and calcshist screes of the montane to alpine levels (Thlaspietea rotundifolii)*. The critical loads for nitrogen for both these habitats are exceeded with the calcareous and calcshist scree habitat being the most sensitive (5- 15 kg N/ha/yr). The dominant source at the site is from short range non-agricultural non-abatable ammonia. These are most likely due to the large seabird colonies on this island. There is some input from other source including long range European import and shipping. It is almost impossible to reduce emissions of ammonia to this site as the seabird colonies represent over 60% of the deposition. It is also worth noting that the condition status of the vascular plant assemblage is cited as 'unfavourable'. No mention is made of nitrogen deposition in the site management statement, although grazing is described as a potential issue.

Peatlands Park SAC

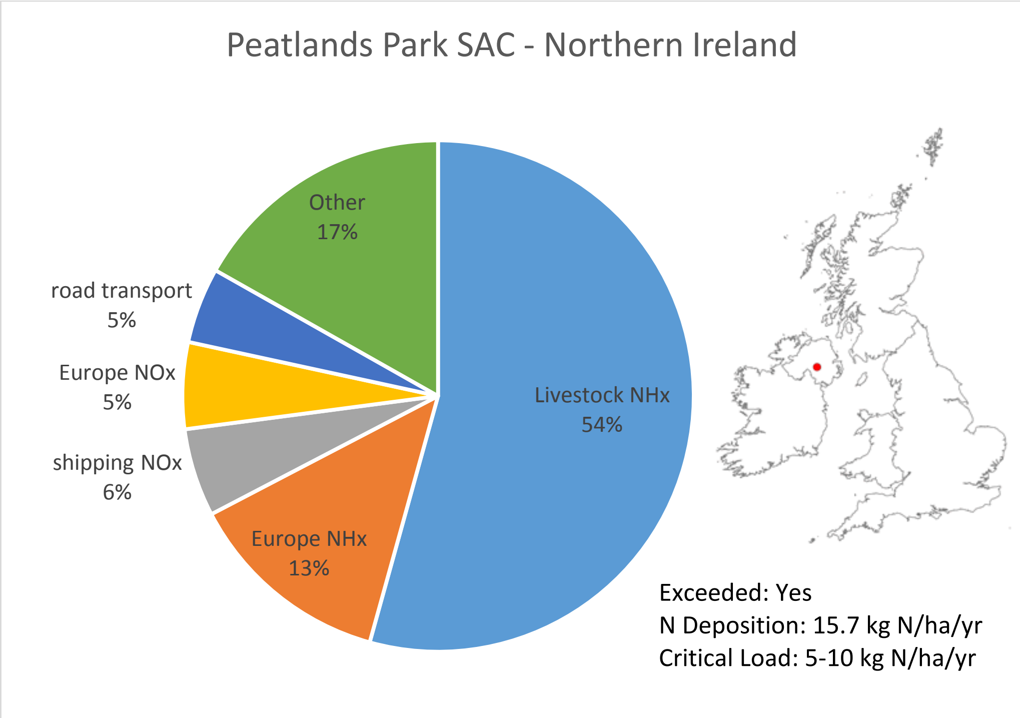


Figure 3.27. Peatlands Park SAC is dominated by ammonia emissions from livestock activities.

Peatland Park SAC is a large lowland raised bog in Northern Ireland. It is listed as having two Annex 1 habitats - *7120 Degraded raised bogs still capable of natural regeneration*, and *91D0 Bog woodland*. The dominant sources in the area are from livestock production (54%) with inputs of ammonia from long range European imports (likely from the Republic of Ireland). The critical load is exceeded for both habitats. The reduction in the critical load exceedance should be linked to reducing agricultural emissions of ammonia in the surrounding area. The degraded raised bog is currently in an unfavourable status. The conservation objectives of the site does include an action to seek to restore concentrations and deposition of air pollutants to at or below the site-relevant critical load.

Epping Forest SAC

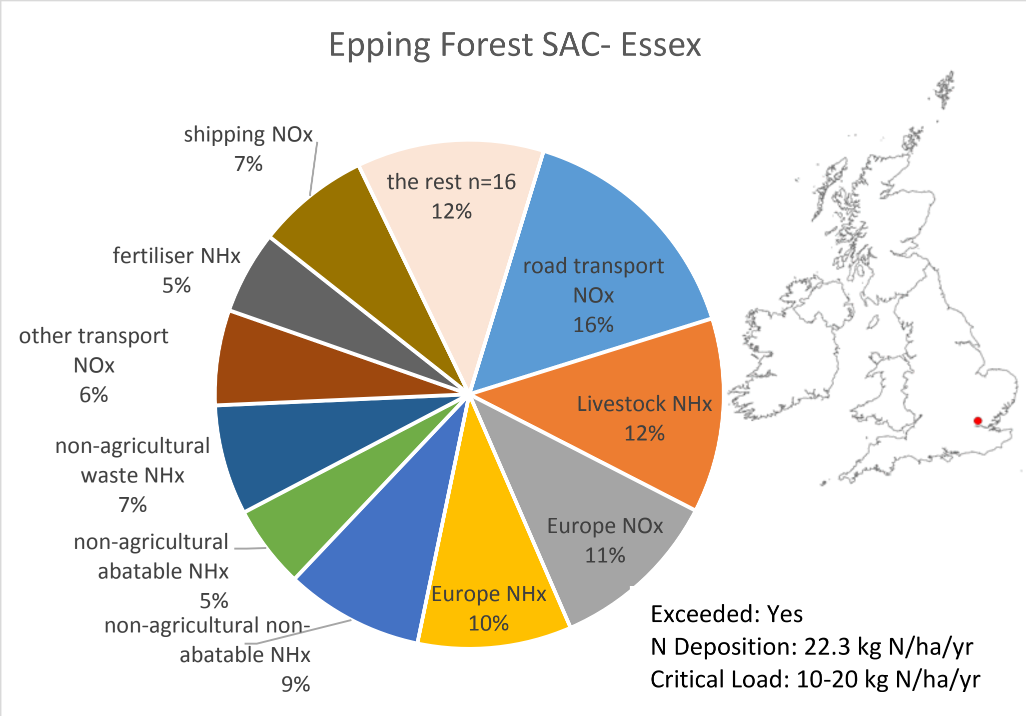


Figure 3.28. Epping Forest SAC shows a large mix of combustion and ammonia sources. Road transport is the dominant source. 21% of the deposition comes from source on continental Europe.

Epping Forest sits to the north-east of London surrounded by urban settlements and to main motorway networks. An ancient woodland it is cited for one Annex I habitat - *9120 Atlantic acidophilous beech forests with Ilex and sometimes also Taxus in the shrublayer (Quercion robori-petraeae or Ilici-Fagenion)*. Epiphytic lichens at the site have declined due to air pollution. The critical loads for nitrogen are exceeded and 64% of the site is in unfavourable status although significant areas are recovering (48%). The site represents a large heterogeneity of sources with road transport being the dominant source (16%), 22% if other transport is included. European import represents nearly a quarter (22%) of the nitrogen deposition and agriculture contributes 19%. Regulating sources to reduce critical load exceedance is a challenge, although there is opportunity to tackle some of the local sources namely agricultural and transport.

Dungeness SAC

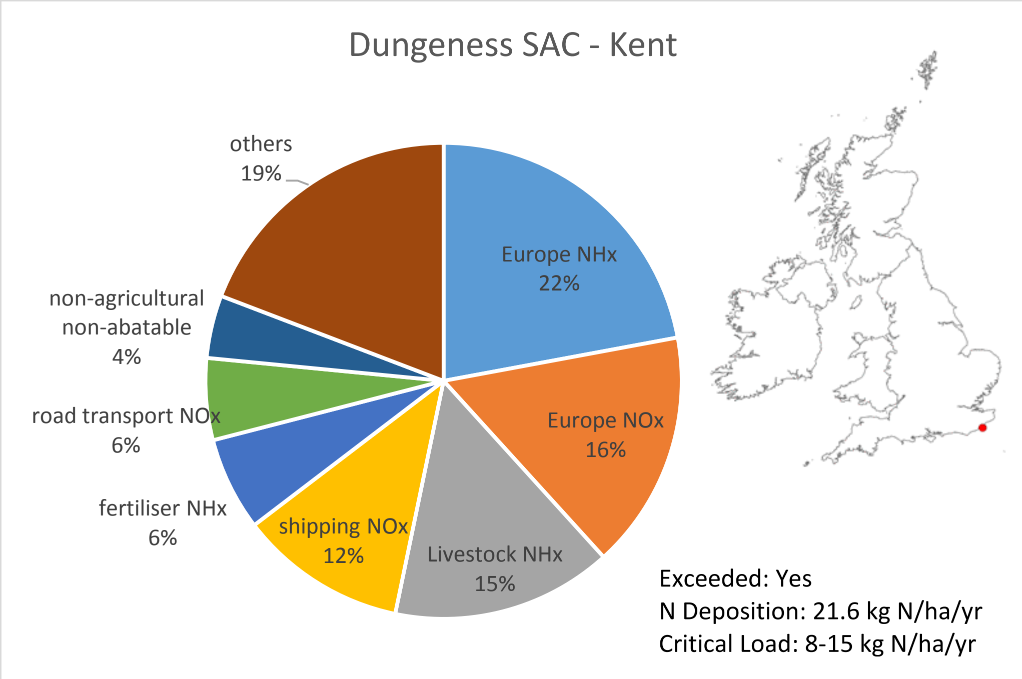


Figure 3.29. Dungeness SAC experiences large imports of deposition originating from emission sources on continental Europe (38%).

Dungeness is the UK's largest shingle structures covering over 60% of this Natura site. It is cited for two Annex 1 habitats - *1210 Annual vegetation of drift lines* and 1220 *Perennial vegetation of stony banks*. The critical loads for both these habitats is exceeded for nitrogen deposition. The site is assessed as 'unfavourable-recovering' in 38% of its area Nitrogen deposition is dominated by long range European imports of both NH_x and NO_x (38%). Air pollution is not mentioned as a threat. Agricultural emissions of ammonia make up 22% of total nitrogen deposition, while shipping, due to the sites location to the main shipping channels of the English Channel, makes up 12%.

Ben Nevis

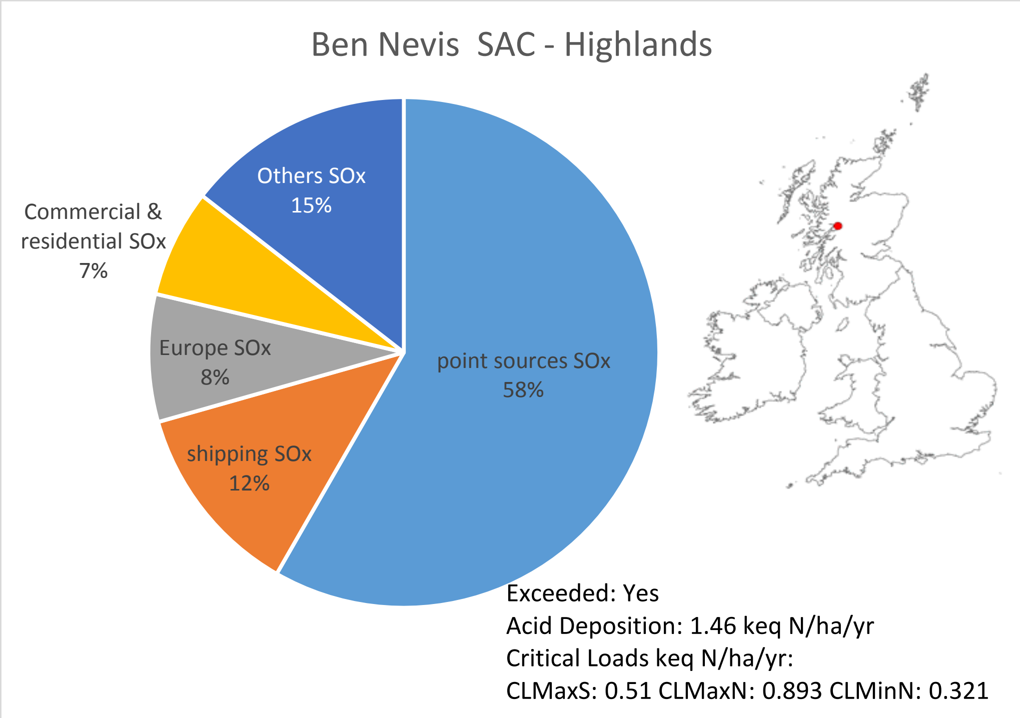


Figure 3.30. Ben Nevis SAC showing the key sulphur sources. Point sources make up 58% of the Sulphur deposition on site.

Ben Nevis is a high-altitude mountain site in the highlands of Scotland containing many sub-types of alpine and subalpine habitats. It is cited for five Annex I habitats, - *6150 Siliceous alpine and boreal grasslands, 6170 Alpine and subalpine calcareous grasslands, 8110 Siliceous scree of the montane to snow levels (Androsacetalia alpinae and Galeopsietalia ladani), 8210 Calcareous rocky slopes with chasmophytic vegetation* and *8220 Siliceous rocky slopes with chasmophytic vegetation*. Sulphur deposition is dominated by long-range point sources (58%) and other long range sources like Europe and international shipping. Regulation for this site requires Directives legislating for large point sources like Industrial Emissions Directive. The CLPTRP and NECD also plays a significant role in reducing this sites deposition.

Newlyn Downs

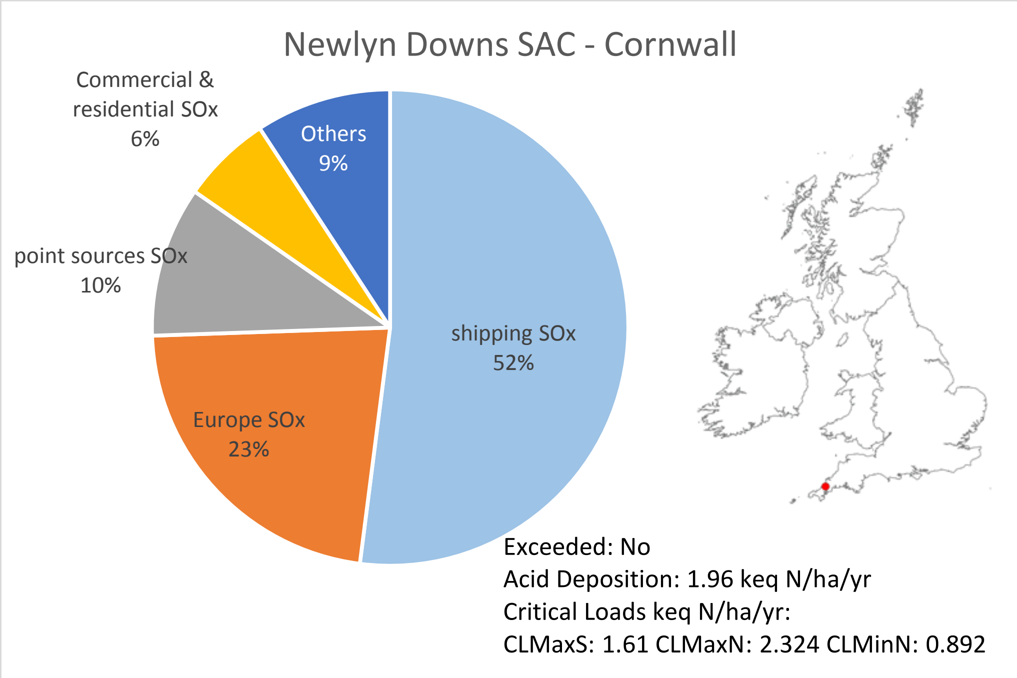


Figure 3.31. Newlyn Downs SAC is situated along the southern coast of the UK near major shipping lanes. International shipping is the most dominant source contributing to 52% of sulphur deposition

Newlyn Downs is an Atlantic wet heath cited for Annex I habitat *Temperate Atlantic wet heaths with Erica ciliaris and Erica tetralix.* The largest source contributing to sulphur deposition is from international shipping. This is due to the SAC being in south west of England and close to the coast and the English Channel shipping lanes.

3.4 Discussion

Source Attribution is a useful policy tool for assessing the spatial differences in the pollutant climate across the Natura 2000 network. The relationship between source and receptor is complex and is heavily influenced by location in a number of ways. The location of significant numbers of Natura sites invariably are in the near vicinity of agricultural activities. Short range deposition of ammonia accounts for over half the nitrogen load on Natura sites. Furthermore, the location of far-away sources that at first sight would seem to have no expected effect on sites are in fact influenced by long-range deposition of nitrogen (30%) and sulphur (67%).

By far the most predominant emission source contributing to nitrogen deposition across the Natura network is livestock emissions. Nearly 90% of all SACs (n=564) have livestock as their dominant source, and on average 32% of nitrogen deposition comes from this source too. For sulphur large point sources are the most dominant sources across the network. Over 75% of all sites (n=482) have point sources as their dominant source.

The extent of exceedance of nutrient nitrogen critical loads at SACs is daunting. Over three quarters of SAC have their most sensitive Annex I habitat exceeded, and the amount of exceedance is also high with many sites experiencing 50 kg N/ha/yr over the critical load.

The case studies presented provide a useful insight into the problems policy makers and regulators have in reducing emissions. Sites depending on their location to sources are influenced in different ways. For example sites in upland areas away from local inputs of agricultural ammonia or road transport are impacted by long-range transport of pollutants as they are often in areas of high rainfall. Similarly, sites situated right in the middle of intensive agricultural zones or next to major roads are impacted on the short range scale. Some sites are affected by natural sources (e.g. seabird colonies) so no form of reduction policy can be put forward in these cases. All sites are in some way affected by long-range transport. This phenomenon clearly has implications for regulating as the sources are often outside the UK on continental Europe. And for internal country regulators issues clearly arise where several point sources in England are contributing to critical load exceedance at sites in Scotland. Combustion sources contributing to long range transport however have been regulated for some time now under EU Directives for example IED and NECD. Tackling the short range sources within the rural landscape have until not adequately been dealt with. Results presented here clearly indicate that addressing agricultural ammonia, and in particular livestock production, is by far the main task ahead for policy makers in reducing on-site exceedances.

3.5 Conclusions

- Source Attribution provides an invaluable method of examining the spatial differences in the pollution climate and pattern of deposition across the UK.
- Deposition is driven by the location of sources in relation to a site and the meteorological variability across the UK.
- The predominant nitrogen source across the Natura SAC network is agricultural livestock which are responsible, on average, for 32% of the nitrogen deposition.
- Agricultural emissions are a big threat to favourable status of Natura sites ecosystem health and biodiversity.

Acknowledgments We gratefully acknowledge funding of this research from Scottish Environment Protection Agency, Northern Ireland Environment Agency, Natural England, Scottish Natural Heritage, the Environment Agency, Natural Resources Wales, and the Joint Nature Conservation Committee.

Chapter 4. Modelling agro-forestry scenarios for ammonia abatement in the landscape.

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4.1 Introduction

Global ammonia emissions have increased substantially over the 20th and early 21st centuries, while future trends in ammonia emission will depend mostly on agricultural practices and the measures that are introduced to decrease ammonia emissions (Van Vuuren *et al.*, 2011). The widespread use of the Haber-Bosch process since the 1950s has made it possible to produce ammonia and its derivatives in large quantities relatively inexpensively (Sutton *et al.*, 2008). Together with increased emissions from fertilizer use, ammonia emissions from intensive livestock production systems have also increased as meat consumption per capita has increased across Europe, Asia and North America (Erisman *et al.*, 2007).

Excess nitrogen can cause eutrophication and acidification effects on semi-natural ecosystems, which in turn can lead to species composition changes and other deleterious effects (Bobbink *et al.*, 2010; Krupa, 2003; Pitcairn *et al.*, 1998; Sheppard *et al.*, 2008; Van den Berg *et al.*, 2008; Wiedermann *et al.*, 2009). Species adapted to low nitrogen (N) availability are at a greater risk from this effect including many slow-growing lower plants, notably lichens and bryophytes. (Pearce & van der Wal, 2002; Bobbink *et al.*, 1998). The quantification of risk associated with air pollution effects on ecosystems was defined by the United Nations Economic Commission for Europe (UNECE) (UNECE, 1996) which describes the concept of "critical loads" and "critical levels": a critical load is the cumulated deposition under which an ecosystem/habitat is not affected by pollution while a critical level is defined as the effects above a certain threshold of concentration of a particular air pollutant. It is estimated that by 2020, 48% of sensitive habitats in the UK will still exceed the critical load for nutrient nitrogen (Hall *et al.*, 2006, Hallsworth *et al.,* 2010).

Legislative measures to reduce ammonia emissions in the UK and across Europe fall under several directives and protocols. As well as defining the concepts of ‘critical loads’ and critical levels’, the UNECE multi-pollutant, multi-effect Protocol also set out a 2010 ceiling for emissions of sulphur, NO_x, VOCs and ammonia. These were negotiated on the basis of scientific assessments of pollution effects and abatement options. The National Emission Ceilings Directive (NECD) (Council Directive 2001/81/EC) aimed to reduce emissions of pollutants that cause acidification, eutrophication and ground-level ozone in order to protect the environment and human health. These two frameworks have a long-term objective to ensure that pollutant levels remain below their critical loads and critical levels.

The EU Industrial Emissions Directive (2010/75/EU (IED)) regulates emissions from large, intensive pig (>2,000 production pigs over 30kg and 750 sows) and poultry units (>40,000 birds) through a system of permits. These ‘hot spot’ sources of ammonia emission can be readily deposited to nearby sensitive ecosystems and protected sites (Loubet *et al.*, 2009). Designated sites like Special Areas of Conservation (SAC) and Special Protected Areas (SPA) are managed under the Habitats Directive (92/43/EEC on the Conservation of natural habitats and of wild fauna and flora) and the Birds Directive (79/409/EEC). Both directives provide a high level of protection to the Natura 2000 network by taking a precautionary approach to controlling polluting activities. Agricultural industries (i.e. farmers) have to report their emissions and show that they are not posing a likely significant threat to the integrity of the protected site.

Because of their effect on turbulence, trees can be effective scavengers of both gaseous and particulate pollutants from the atmosphere (Beckett 2000; Nowak, 2000) with dry deposition rates to forest exceeding those to grassland by typically a factor of 3–20

(Gallagher *et al.*, 2002; Fowler *et al.*, 2004). This implies that the conversion of grassland and arable land to trees or targeted management of existing wooded areas, can be used to promote the removal of ammonia from the atmosphere, thereby reducing the potential impacts on nearby sensitive ecosystems and to some extent long-range transport of these pollutants. In a modelling study, Dragosits *et al.* (2006) showed that tree belts can reduce deposition to sensitive ecosystems, with trees surrounding the sensitive habitats being more effective than trees around the sources for their scenarios. The capture of ammonia by surrounding vegetation has been studied by Patterson *et al.* (2008), who observed lower NH_3 concentrations were measured when potted trees were present downwind of the poultry house fans compared with when the trees were removed (16.4 vs. 19.3 ppm). Modelling research undertaken by Asman 2008 on the entrapment of ammonia by shelterbelts showed that capture of dry deposited gaseous ammonia increased with the height of the shelterbelt and the stability of the atmosphere (favouring neutral conditions), but decreased further away from the source to the shelterbelt. At 200m away from a source the model predicted that a maximum 37% of the emission of a ground level point source of ammonia can be dry deposited before the plume reaches a shelterbelt that is located 200 m downwind. Then another 11% can be removed by a 10 m high shelterbelt.

Experimental approaches to measure ammonia recapture carried out by Theobald *et al.*, 2001, recorded a 3% recapture from throughfall measurements. While previous modelling of the MODDAS model (Theobald *et al.*, 2004) showed a recapture of ammonia emissions up to 15%. In this study we evaluated different tree planting designs near ammonia sources using the MODDAS-THETIS model to quantify optimal designs to capture ammonia thereby protecting nearby vulnerable ecosystems. The MODDAS-THETIS model allows the modification of parameters such as downwind canopy length, leaf area index (LAI) and leaf area density (LAD) to be varied, and thereby providing a tool to examine how tree configuration and structure can be optimised to maximise NH_3 capture. Potential ammonia recapture is assessed and interpreted in terms of practical farm management approaches.

4.2 Methodology

There are two important considerations (Figure 4.1) when designing tree systems for ammonia recapture:

1. To get the ammonia into the woodland and through the densest part of the canopy, a reasonably open understorey would be necessary to prevent the ammonia passing over the top of the woodland and acting as a block to the airflow.
2. Prevention of the loss of ammonia out of the downwind edge of the woodland. To stop this happening, a region of dense vegetation could be planted at the downwind edge to act as a backstop and force the ammonia up through the canopy as shown in Figure 4.1.

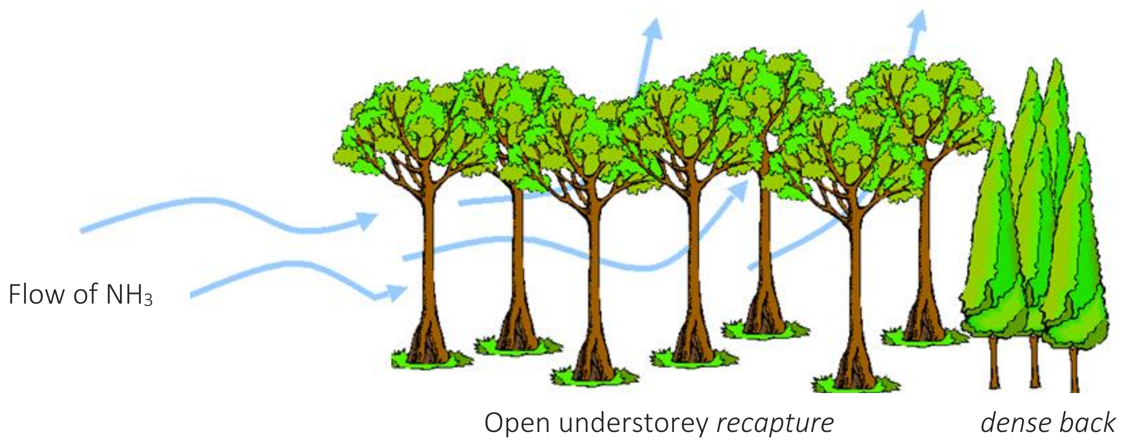


Figure 4.1. Schematic diagram of a tree belt design to maximize recapture of ammonia. From Theobald *et al.*, 2004.

MODDAS-THETIS is a flexible two-dimensional (along wind and vertical) model that can be used to examine the ammonia abatement potential of agro-forestry structures in the landscape. MODDAS is a Lagrangian stochastic model for gaseous dispersion, coupled with a multi-layer exchange model including a stomatal compensation point (Loubet *et al.*, 2006). THETIS is an Eulerian (k-ε) turbulence model designed for transfer within the planetary boundary layer as well as within a plant canopy (Foudhil, 2005). The two models are coupled together such that the output of the THETIS model serves as the turbulence input of the MODDAS model, namely the horizontal (u,v) and vertical (w) components of the wind velocity, and the dissipation rate of the turbulent kinetic energy (ε). Both models have been validated in conditions similar to those modelled here, specifically MODDAS in an ammonia release experiment over a developed maize canopy and a grassland (Loubet *et al.*, 2006), and THETIS over several canopy arrangements (Foudhil, 2005; Dupont and Brunet, 2006). The coupling of the two models requires the partitioning of the turbulent kinetic energy (*k*) into its three components (σu, σv and σw). By considering the equality of Eulerian and Lagrangian turbulent diffusivities (Raupach, 1989) and by empirically setting the horizontal partitioning (based on Loubet, 2000):

$$\alpha_{uv} = \sigma_u / \sigma_v = 1.25 \qquad (1)$$

Then the vertical partitioning is calculated as:

$$\alpha_w = \sigma_w / (\sigma_u + \sigma_v + \sigma_w) = 0.37 \quad (2)$$

The model scenario setup is based around a woodland schema as shown in Figure 4.2, where different blocks of woodland or canopy (c) are formed by varying the height of canopy (h_c), the length of canopy (x_c), the leaf area density profile (LAD(z)), the Leaf Area Index (LAI) (not shown in the figure), the source strength (Q_s) and the source length (X_s). By using the woodland schema, different heights and lengths of woodland blocks of differing LAIs and LAD structures were configured to examine the optimal combination of parameters to maximise ammonia recapture in the model run.

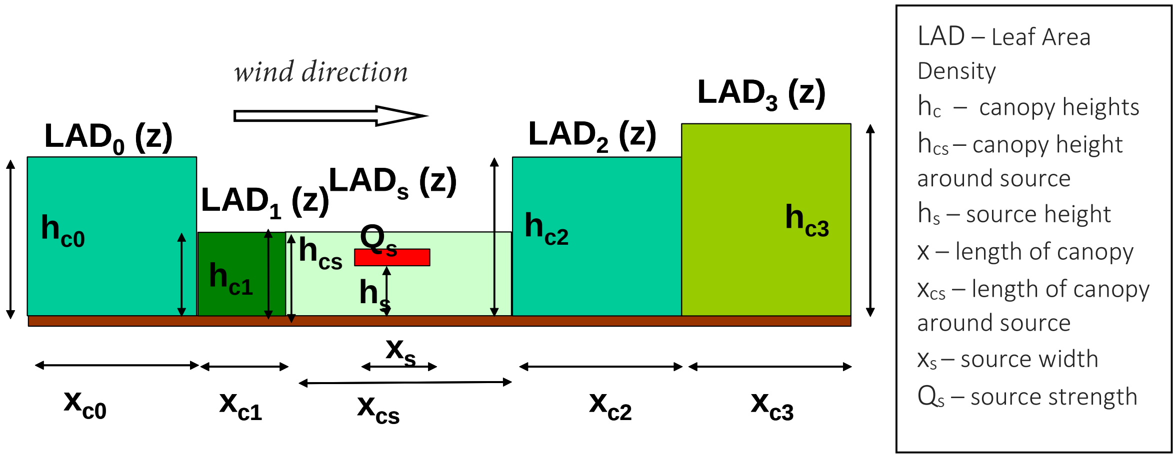


Figure 4.2. General model scheme of the woodland and source geometry that was tested in the scenarios. The shaded green boxes reflect different lengths (x_c) and heights (h_c), and LADs of canopy blocks. There is no limit to the different canopy structures that can be added to the model. The red box represents the source (Q_s) with a specified height (h_s) and downwind length (x_s). Indexes 0 to 3 to LAD, x_c and h_c correspond to canopy number, while index *s* corresponds to the source location

The vertical canopy structure of trees can be represented by the LAD which is the surface of leaves per unit volume. LAIs, the surface of leaves per unit ground surface area, are used to normalize the relative LAD profiles to produce LAD as a function of height. LAI values typically range from 0 for bare ground to ≥6 for a dense forest. Five characteristic canopy profiles are illustrated in Figure 3. LAD-0 is a flat canopy block profile from crown to base, LAD-1 is a canopy denser at the top and brashed toward the bottom, LAD-2 is a canopy with a marked crown, LAD-4 is like LAD-2 but with an additional bottom shrub layer near the ground, and LAD-10 is a coniferous profile with brashed bottom.

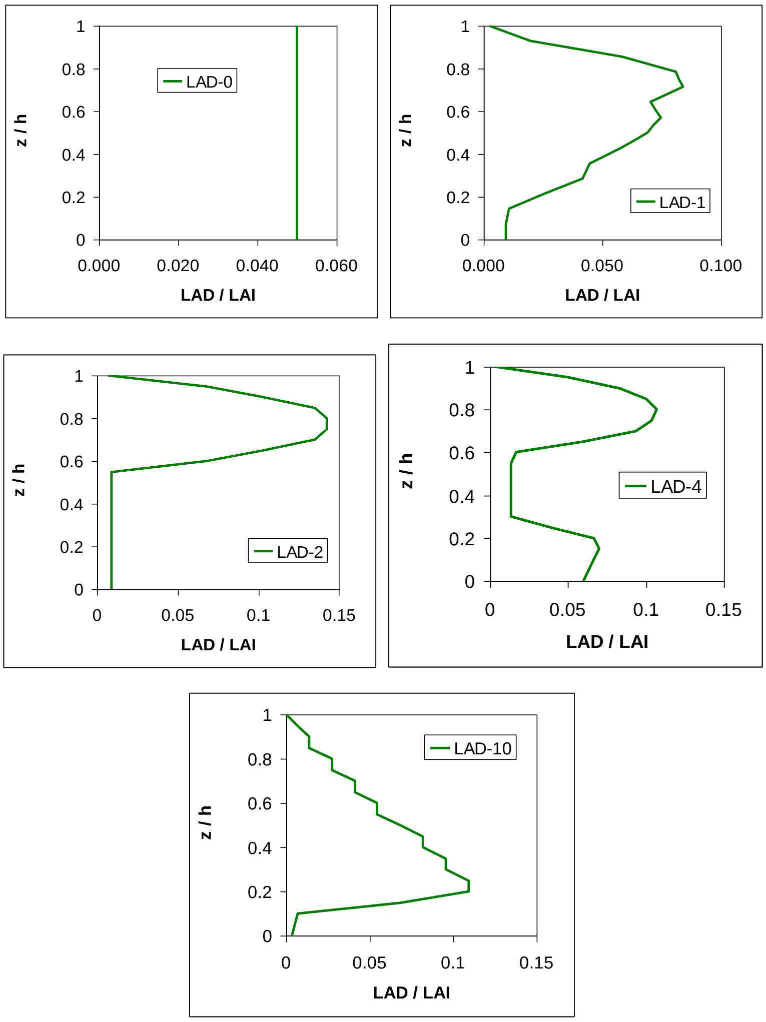


Figure 4.3. Leaf Area Density (LAD(z)) profiles of the canopies (og height h)used in the MODDAS-THETIS simulations. LAD(z) are a function of height showing the vertical canopy structure from the crown to the ground. All canopy profiles were used in these scenarios

4.2.1 Source Types

Three source types were tested representing three livestock production systems: poultry housing, a waste storage system (slurry lagoon with crust) and free-range poultry under tree cover. For each source type, the MODDAS-THETIS model was used to examine the recapture efficiency of tree planting around these sources looking at different canopy structure scenarios, lengths and differing LADs and LAIs to obtain an estimate of recapture potential.

For these three source types, the 'main canopy' was defined as the open understorey surrounding or above the source, while an optional dense 'backstop' canopy was also included. The backstop serves to capture NH_3 as it leaves the main canopy.

The source types, visualised in Figure 4.4, were:

- a **housing source** of ammonia that was emitting at a height of 2-2.5 m height, with an along wind length of 4-5 m and with a source strength of 300 kg NH_3-N yr^{-1} (Figure 3). Up to 39% of the UK's ammonia emissions comes from housing systems where hard surfaces prevent urine and manure being absorbed easily (compared with contact with the soil) (Misselbrook *et al.* 2010).
- a **slurry lagoon** which was considered to emit at a height of 0.1 to 0.2 m, with a source strength of ~400 kg NH_3-N yr^{-1} (Figure 4). Up to 6% of UK emissions of ammonia are estimated to come from slurry storage systems (Misselbrook *et al.*, 2010). Emission depends more on the surface area of slurry/manure in contact with the air rather than the total amount of slurry/manure stored.
- an "**under-storey**" source, in which the emissions (e.g. from free-range chickens) were at a height of 0.1 - 0.2 m under the canopy, with a source strength of 625 kg NH_3-N yr^{-1} (Figure 5). In 1946 nearly 98% of the UK flock of poultry layers were free-range. By 1980 95% were in cage systems (FAWC, 1998). Out of the 26 million poultry egg-layers in the UK, free-range layers currently account for around 38%.

However, although these birds have access to the outdoors they spend a significant part of their time within the barn itself (Dawkins *et al.*, 2003).

It should be noted that since the ammonia concentration is linearly related to the source in the model (see Loubet *et al.*, 2006), one can compare the three situations by normalising the concentration or the deposition by the source strength. A set of runs (on the housing source type only) were set up to examine the effect of changing the source strength by a factor of 100.

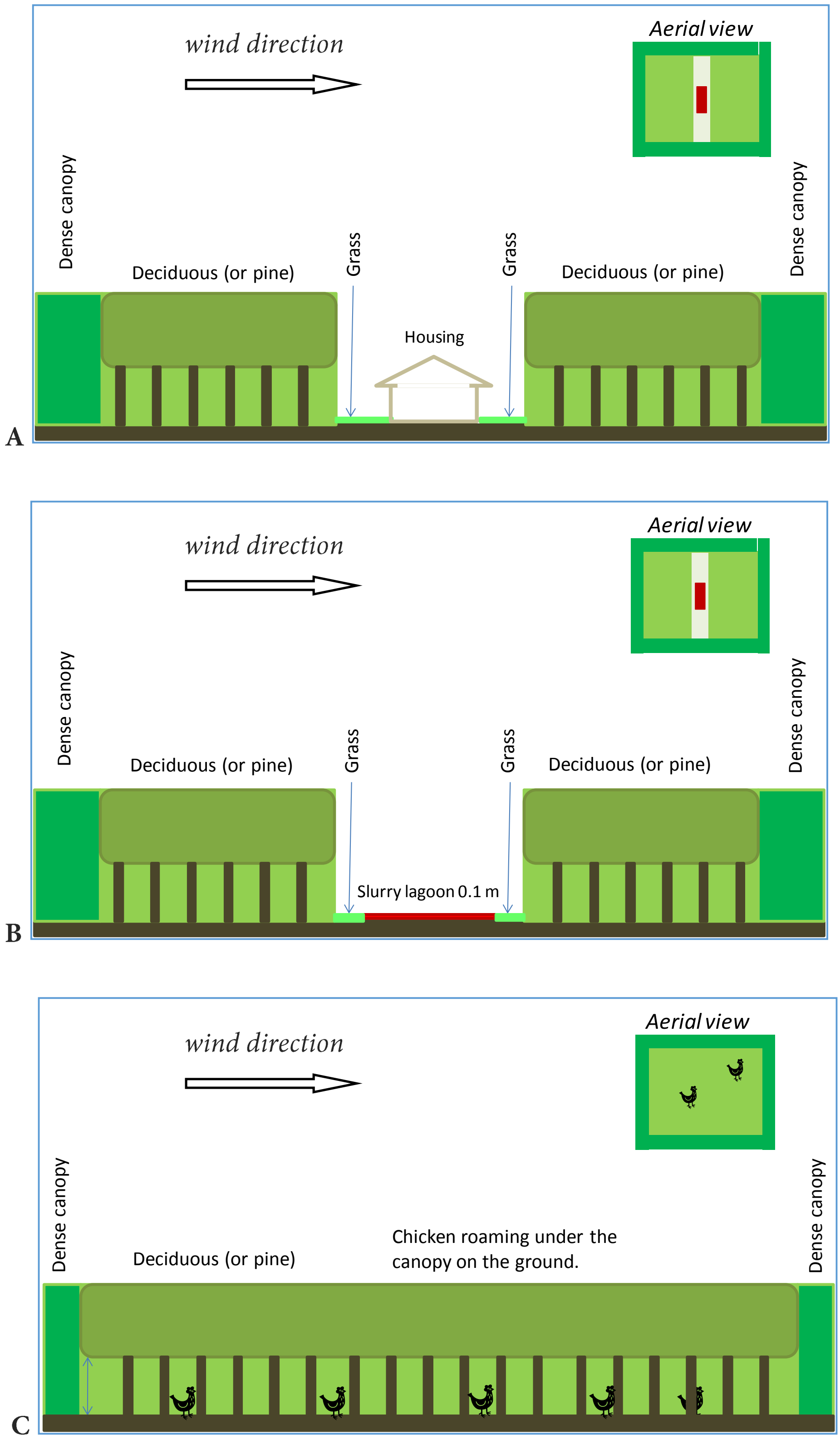


Figure 4.4. Visualisation of example source types for tree belts upwind and downwind: (A) Housing source type. (B) Lagoon source type (red line), a variant of the housing scenario and (C) Under-storey source scenario with free-ranging chickens. The 2D aerial view (top right) shows the scheme from above.

4.2.2 Scenarios

For each of the three source types, scenarios were set up by altering LAD, LAI, canopy height, source strength, and canopy length. These scenarios were run with neutral atmospheric stability, and the wind speed at 50 m upwind of the source was set to 5 m s^{-1}. For each scenario, symmetrical and non-symmetrical (i.e. only downwind) canopy structures were assessed.

Table 4.1: Model scenarios for the three source types – housing, lagoon, and understorey livestock. The green boxes shaded show the differing sets of changing parameters that are being compared. The backstop canopy was set with a LAD 10 (coniferous tree profile). Symmetrical means that the canopy profiles are identical in the upwind and downwind direction.

Model scenario	Design	main canopy length	LAI	Height (m)	LAD profile	Back-stop length (m)	LAI	Canopy height (m)
Housing 1	symmetrical	30	6	10	0	0	-	-
Housing 2	downwind	30	6	10	0	0	-	-
Housing 3	downwind	25	3	10	1	5	6	10
Housing 4	downwind	25	3	10	4	5	6	10
Housing 5	downwind	25	3	10	10	5	6	10
Housing 6	downwind	25	3	10	2	25	6	10
Housing 7	downwind	25	3	10	2	50	6	10
Housing 8	downwind	25	3	10	10	50	6	10
Housing 9	downwind	50	3	10	10	50	6	10
Housing 10	downwind	100	3	30	10	50	6	30
Housing 11 Source x 10	symmetrical	30	6	10	0	0	-	-
Housing 12 Source x 1	symmetrical	30	6	10	0	0	-	-
Housing 13 Source / 10	symmetrical	30	6	10	0	0	-	-
Lagoon 1	symmetrical	30	6	10	0	0	-	-
Lagoon 2	downwind	30	6	10	0	0	-	-
Lagoon 3	downwind	25	3	10	0	5	6	10
Lagoon 4	downwind	25	1	10	0	5	6	10
Lagoon 5	downwind	25	3	10	2	5	6	10
Lagoon 6	downwind	25	3	10	4	5	6	10

Model scenario	Design	main canopy length	LAI	Height (m)	LAD profile	Back-stop length (m)	LAI	Canopy height (m)
Lagoon 7	downwind	25	3	10	10	5	6	10
Lagoon 8	downwind	25	3	10	2	25	6	10
Lagoon 9	downwind	25	3	10	2	50	6	10
Understorey	symmetrical	100	3	10	0	0	-	-
Understorey	symmetrical	100	3	10	0	5	6	10
Understorey	symmetrical	100	3	10	0	10	6	10
Understorey	symmetrical	100	3	10	0	25	6	10
Understorey	symmetrical	100	3	10	0	50	6	10
Understorey	symmetrical	100	6	10	0	50	6	10
Understorey	symmetrical	100	6	10	1	50	6	10
Understorey	symmetrical	100	6	10	2	50	6	10

4.2.3 Model Parameterisation

The deposition parameters were selected to reproduce realistic deposition rates. The stomatal resistance was modeled with a Jarvis approach (Equation 3)

$$R_s = R_{smin}\,(1+ \beta_s/PAR) \qquad (3)$$

where PAR is the photosynthetically active radiation (W m^{-2}), R_{smin} (= 60 s m^{-1}) is the minimum stomatal resistance and β_s (= 7) is the stomatal response to light. The cuticular resistance was set with Equation 4

$$R_w = R_{wmin}\, e^{([1-RH]/\beta w)} \qquad (4)$$

where R_{wmin} = 7 s m $^{-1}$ is the minimum cuticular resistance and βw= 7 is the response to relative humidity RH (Massad *et al.*, 2010). The PAR above the canopy was set to 400 W m^{-2} and RH is the relative humidity in the canopy (set to 90% in order to study conditions favourable to NH_3 deposition). The ammonia emission potential of the canopy and soil was set to zero (Γ = 0). It should be noted that under real-life conditions there is a potential for saturation of the surfaces that are exposed to high loads of ammonia and therefore it should be stressed that the estimated deposition is an upper limit, with small cuticular and stomatal resistances and a zero compensation point in order to assess the effects of canopy structure.

4.2.4 Sensitivity analysis

To take into account the yearly variations of abiotic factors like temperature, relative humidity and radiation we have done a run for each calendar month simulating variations in key parameters (Table 4.2). We have also looked at the effect of loss of leaves in deciduous trees during winter months by varying the LAI of the main canopy for each month. The runs were based on the Housing 7 scenario (Table 4.3).

Table 4.2. Monthly variation scenarios showing changes in LAI (main canopy) to mimic leaf loss over winter, photosynthetically active radiation (PAR), temperature (Ta), Relative Humidity (RH), and Wind speed.

monthly variation in the deposition	LAI main canopy m2 m-2	LAI backstop m2 m-2	PAR W m-2	Ta °C	RH %	Wind speed m s-1
January	0.5	6.0	134	2.2	100	5.7
February	0.5	6.0	201	1.3	100	5.9
March	0.5	6.0	340	3.1	90	4.9
April	1.0	6.0	516	4.7	80	5.1
May	3.0	6.0	668	6.1	70	5.8
June	3.0	6.0	790	8.2	60	4.5
July	3.0	6.0	628	8.5	50	4.0
August	3.0	6.0	616	7.8	50	3.1
September	3.0	6.0	418	7.5	60	3.3
October	1.5	6.0	271	6.7	80	6.3
November	0.5	6.0	132	2.5	90	5.6
December	0.5	6.0	87	1.3	100	5.9

4.3 Results

A detailed array of configuration scenarios was run for each of the three source types, the results from which are summarised in Table 4.3 to Table 4.5. The key results are how much ammonia was deposited (as % of emitted NH_3) and in which part of the woodland schema the deposition occurred.

Table 4.3. Model scenarios and results for the housing source. The green shaded boxes show the sets of varied parameters that are being compared

Model scenario	Design	main canopy length	LAI	Height (m)	LAD profile	Back-stop length (m)	LAI	Canopy height (m)	% TOTAL deposited	% deposited upwind of the main canopy (x_{c0})	% deposited in main canopy (x_{c1})	% deposited in back-stop (x_{c2})
Housing 1	symmetrical	30	6	10	0	0	-	-	16%	2%	14%	0%
Housing 2	downwind	30	6	10	0	0	-	-	17%	0%	17%	0%
Housing 3	downwind	25	3	10	1	5	6	10	7%	0%	6%	1%
Housing 4	downwind	25	3	10	4	5	6	10	9%	0%	7%	2%
Housing 5	downwind	25	3	10	10	5	6	10	12%	0%	10%	2%
Housing 6	downwind	25	3	10	2	25	6	10	16%	0%	5%	11%
Housing 7	downwind	25	3	10	2	50	6	10	25%	0%	5%	20%
Housing 8	downwind	25	3	10	10	50	6	10	25%	0%	9%	16%
Housing 9	downwind	50	3	10	10	50	6	10	27%	0%	15%	12%
Housing 10	downwind	100	3	30	10	50	6	30	17%	0%	12%	5%
*Housing 11 Source * 10*	symmetrical	30	6	10	0	0	-	-	16.1%			
*Housing 12 Source * 1*	symmetrical	30	6	10	0	0	-	-	16.5%			
Housing 13 Source / 10	symmetrical	30	6	10	0	0	-	-	17.4%			

Table 4.4. Model scenarios and results for the "slurry lagoon" source. The green shaded boxes show the sets of varied parameters that are being compared

Model scenario	design	main canopy length (m)	LAI	height(m)	LAD profile	Back-stop length (m)	LAI	Canopy height (m)	% TOTAL deposited	% deposited upwind of the main canopy (x_{c0})	% deposited in main canopy (x_{c1})	% deposited in back-stop (x_{c2})
Lagoon 1	symmetrical	30	6	10	0	0	-	-	19%	2%	17%	0%
Lagoon 2	downwind	30	6	10	0	0	-	-	19%	0%	19%	0%
Lagoon 3	downwind	25	3	10	0	5	6	10	11%	0%	9%	2%
Lagoon 4	downwind	25	1	10	0	5	6	10	5%	1%	2%	2%
Lagoon 5	downwind	25	3	10	2	5	6	10	7%	0%	6%	1%
Lagoon 6	downwind	25	3	10	4	5	6	10	5%	0%	4%	1%
Lagoon 7	downwind	25	3	10	10	5	6	10	5%	0%	4%	1%
Lagoon 8	downwind	25	3	10	2	25	6	10	9%	0%	3%	6%
Lagoon 9	downwind	25	3	10	2	50	6	10	14%	0%	4%	10%

Table 4.5. Model scenarios and results the understorey source. The green shaded boxes show the sets of varied parameters that are being compared.

Model scenario	design	main canopy length (m)	LAI	height(m)	LAD profile	Back-stop length (m)	LAI	Canopy height (m)	% TOTAL deposited	% deposited upwind of the main canopy (x_{c0}	% deposited in main canopy (x_{c1})	% deposited in back-stop (x_{c2})
Understorey 1	symmetrical	100	3	10	0	0	-	-	15%	0%	15%	0%
Understorey 2	symmetrical	100	3	10	0	5	6	10	17%	0%	15%	2%
Understorey 3	symmetrical	100	3	10	0	10	6	10	20%	0%	16%	4%
Understorey 4	symmetrical	100	3	10	0	25	6	10	28%	0%	20%	8%
Understorey 5	symmetrical	100	3	10	0	50	6	10	37%	0%	24%	13%
Understorey 6	symmetrical	100	6	10	0	50	6	10	60%	0%	51%	9%
Understorey 7	symmetrical	100	6	10	1	50	6	10	49%	0%	45%	4%
Understorey 8	symmetrical	100	6	10	2	50	6	10	24%	0%	22%	2%

4.3.1 Housing Scenarios

In the Housing scenarios (Table 4.3), the maximum NH_3 deposition simulated was 27% in Housing 9 which had a 50 m downwind canopy (LAI =3 $m^2 m^{-2}$, LAD profile =10), 50 m backstop (LAI =6, LAD = 10,). The deposition in the other scenarios ranged between 7% and 25% of the emission.

Comparing Housing 1 and Housing 2, where the only difference is the presence of the symmetrical canopies, the total deposition does not differ much, with the symmetrical situation giving slightly estimated smaller deposition rates even though part of the deposition occurs in the upwind canopy due to backward diffusion. With housing runs Housing 3, 4 and 5 the effect of varying the LAD in the main canopy is observed (see Figure 4.3 for corresponding LAD profiles). NH_3 deposition increased with LAD profiles 1, 4 and 10 , with the LAD-10 profile (coniferous profile with 15-20% of the bottom free of leaves) recapturing the most NH_3. The deposition increases with the following order of LAD: LAD-1, LAD-2, LAD-4, LAD-0, LAD-10. Housing 6 and Housing 7 demonstrate that having a longer backstop increases deposition (from 16% to 25% in these cases). Most of the modelled deposition in these scenarios occurs in the backstop and the proportion deposited in the main canopy remains stable with LAD-2 but decreases with LAD-10 (when the length of the backstop increases). The deposition in the backstop is not proportional to the length of the backstop.

Increasing the main canopy length, when the backstop length is set to 50 m (HS 8 & 9), increases the proportion of NH_3 recaptured significantly in the main canopy, but at the same time decreases the deposition in the backstop. The two effects counteract each other resulting in a net increase of only 3% in recapture efficiency. Another comparison can be made between Housing 7 and 8 which compares LAD 2 (brashed trunk) with LAD 10 (coniferous profile). In both cases the deposition is estimated at 25% of the emission although the backstop plays a larger role in LAD 2 (20%) compared with LAD 10 (16%).

The increase of the canopy height from 10 to 30 m with a constant LAI leads to a decrease in the deposition rates (Housing 9 and Housing 10). This is primarily due to a decrease in LAD, hence leading to a higher wind speed within the canopy and an increase in the turbulent mixing at the source location (asymmetrical scenario).

Housing runs 11 to 13 show the effect of changing the source strength by up to 100%. The difference is small in the deposition (0.75%) when the source is multiplied by 100 with the likely differences being due to cumulated rounding errors. We can however conclude that the model is indeed linear, i.e. the concentration and deposition are both proportional to the source strength.

4.3.2 Lagoon Scenarios

In the lagoon scenarios (Table 4.4), the percentage recapture is in general smaller than in the housing scenarios. The same effects can be seen, except that the LAD profile has an inverse effect on the deposition in Lagoon 3, and in runs Lagoon 5-7 the maximum deposition is obtained with the constant LAD profile (LAD-0). The concentration profile pattern has a maximum remaining very close to the ground when compared to the housing scenarios. In the lagoon scenarios, the source is at the ground where the wind speed tends to zero and hence mixing is slow, while in the housing scenarios, the source is higher where mixing is more efficient. Hence the main differences are linked with the LAD profile characteristics near the ground. When open canopies with structures near the base (e.g. LAD-2) are used (Lagoon 8 and 9) then a long backstop is required to achieve comparable deposition rates to those with LAD-0

4.3.3 The understorey scenarios

In the understorey scenarios (Table 4.5), the capture increased from 15% to 37% for a backstop canopy length increasing from 0 to 50 m respectively (scenarios 1-5, LAI main canopy = 3, LAD main canopy=0). The percentage captured in the main canopy increased linearly with the canopy LAI (runs Understorey 4-5), but a canopy LAD denser at the top of the canopy (LAD-1), was less efficient in capturing NH_3 than a homogeneous LAD (runs Understorey 6-8). It is noted that Understorey 6 had the largest recapture percentage of all the scenarios considered.

4.3.4 Sensitivity Analysis

The sensitivity analysis shows the change in deposition in the canopy over the year with higher capture in the summer months as the main canopy is more effective at capturing ammonia (Table 4.6). However, when a varying RH is applied (Table 4.7) the opposite is true as the winter months capture more deposition mainly due to the effect of the back-stop alone. RH over the summer has a significant negative effect on both main and back stop canopies.

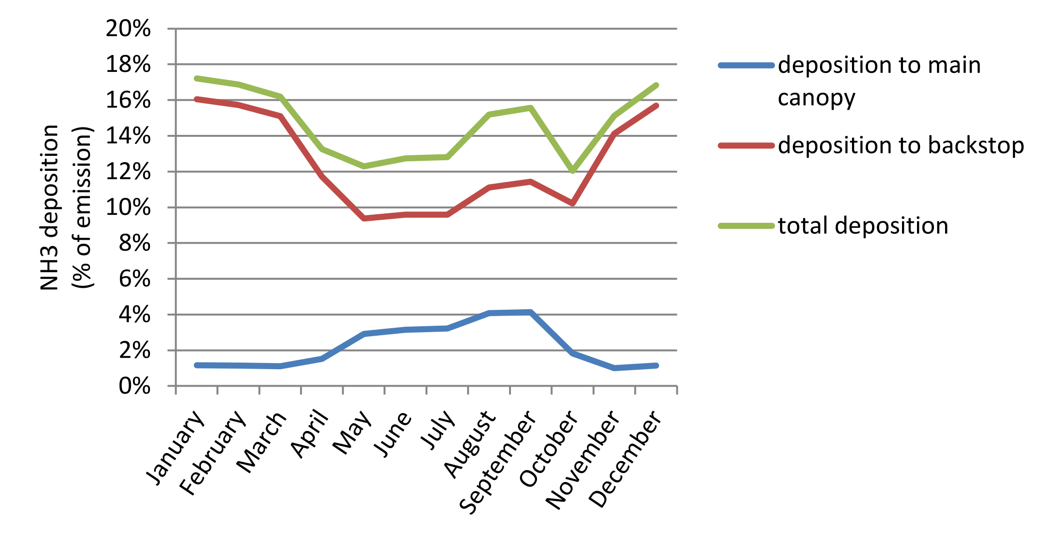
Table 4.6. Changes in deposition capture in the canopy throughout the year with RH kept constant

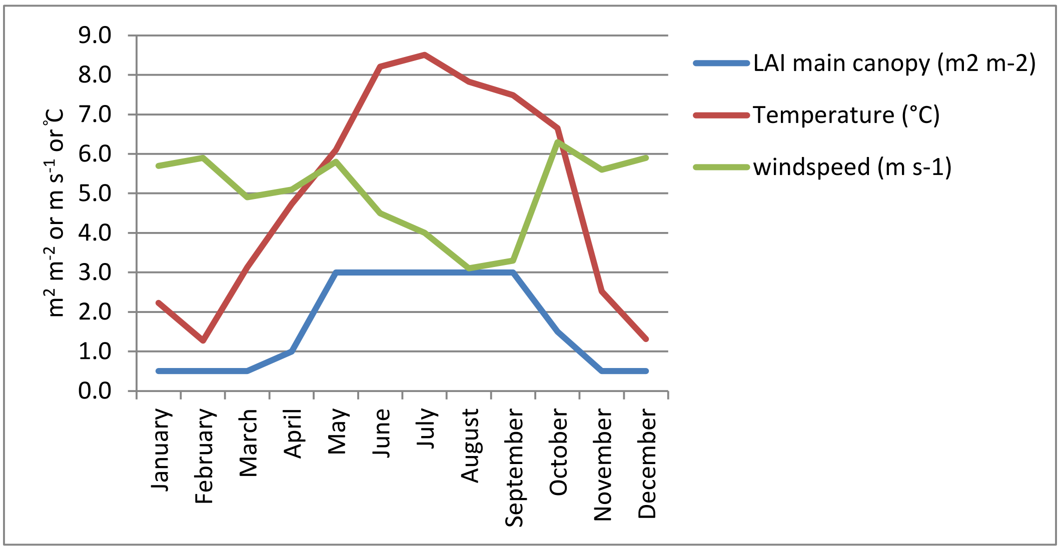
monthly variation in the deposition	deposition in the main canopy	deposition in backstop	total deposition	LAI main canopy	LAI backstop	PAR	Ta	RH	Wind speed
	%	%	%	$m^2\ m^{-2}$	$m^2\ m^{-2}$	$W\ m^{-2}$	°C	%	$m\ s^{-1}$
January	1.0%	13.9%	14.9%	0.5	6.0	134	2.2	90	5.7
February	1.0%	13.6%	14.6%	0.5	6.0	201	1.3	90	5.9
March	1.1%	15.1%	16.2%	0.5	6.0	340	3.1	90	4.9
April	1.9%	14.0%	15.9%	1.0	6.0	516	4.7	90	5.1
May	4.5%	13.4%	17.9%	3.0	6.0	668	6.1	90	5.8
June	5.4%	15.0%	20.4%	3.0	6.0	790	8.2	90	4.5
July	5.9%	15.8%	21.7%	3.0	6.0	628	8.5	90	4.0
August	7.1%	17.4%	24.5%	3.0	6.0	616	7.8	90	3.1
September	6.8%	17.0%	23.8%	3.0	6.0	418	7.5	90	3.3
October	2.3%	12.4%	14.7%	1.5	6.0	271	6.7	90	6.3
November	1.0%	14.1%	15.1%	0.5	6.0	132	2.5	90	5.6
December	1.0%	13.5%	14.5%	0.5	6.0	87	1.3	90	5.9

Table 4.7. Changes in deposition capture in the canopy throughout the year with varying RH

monthly variation in the deposition	deposition in the main canopy	deposition in backstop	total deposition	LAI main canopy	LAI backstop	PAR	Ta	RH	Wind speed
	%	%	%	$m2\ m^{-2}$	$m2\ m^{-2}$	$W\ m^{-2}$	°C	%	$m\ s^{-1}$
January	1.2%	16.0%	17.2%	0.5	6.0	134	2.2	100	5.7
February	1.1%	15.7%	16.9%	0.5	6.0	201	1.3	100	5.9
March	1.1%	15.1%	16.2%	0.5	6.0	340	3.1	90	4.9
April	1.5%	11.7%	13.3%	1.0	6.0	516	4.7	80	5.1
May	2.9%	9.4%	12.3%	3.0	6.0	668	6.1	70	5.8
June	3.1%	9.6%	12.7%	3.0	6.0	790	8.2	60	4.5
July	3.2%	9.6%	12.8%	3.0	6.0	628	8.5	50	4.0
August	4.1%	11.1%	15.2%	3.0	6.0	616	7.8	50	3.1
September	4.1%	11.4%	15.6%	3.0	6.0	418	7.5	60	3.3
October	1.8%	10.2%	12.0%	1.5	6.0	271	6.7	80	6.3
November	1.0%	14.1%	15.1%	0.5	6.0	132	2.5	90	5.6
December	1.1%	15.7%	16.8%	0.5	6.0	87	1.3	100	5.9

Figure 4.5 shows the monthly changes in LAI, wind speed, temperature and RH, as well the changing deposition captured by the canopy throughout the year. The reduction in RH is compensated by the reduction in wind speed in June to September which explains why the deposition is maintained high during this period (bottom graph).





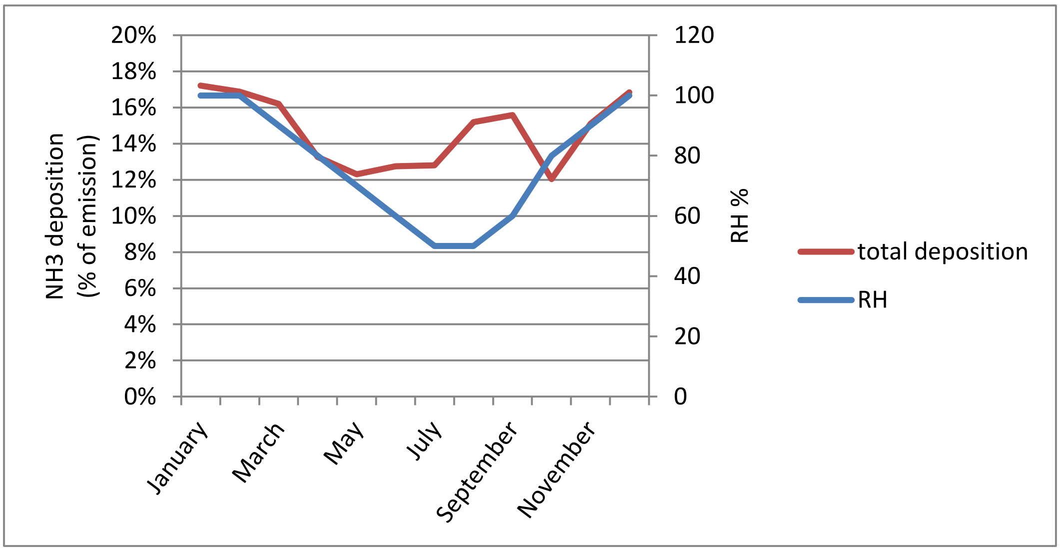


Figure 4.5. Graphs from Table 6 showing the monthly fluctuations in abiotic factors and deposition captured in the canopy.

4.4 Concentration fields

For the symmetrical scheme (Housing 1), the presence of canopy both upwind and downwind of the source increases the vertical dispersion and also the upwind dispersion due to the increased turbulent kinetic energy (Figure 4.6, Housing 1). The asymmetrical scheme, Housing 2, shows a downstream decrease in the NH_3 concentration inside the canopy, but there is a subsequent increase in downwind concentration from the canopy due to a (calm air) recirculation zone. The scheme with a longer main canopy and longer backstop (Housing 9) leads to a decrease in the concentration in the canopy which is similar to the concentration field simulated with a smaller main canopy (Housing 1). In the case of the lagoon, the same behaviour is observed for the NH_3 concentration with or without an upwind main canopy (data not shown).

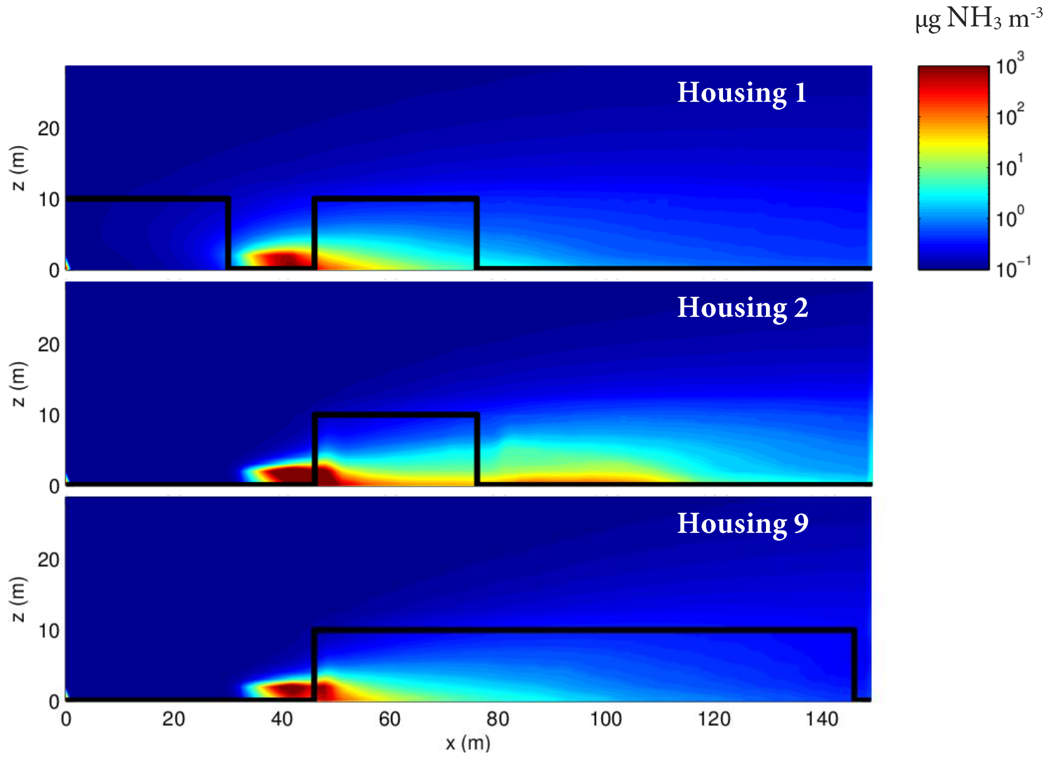


Figure 4.6. Output from MODDAS-THETIS showing the concentration field in the 'Housing' source runs from the top – scenario Housing 1, Housing 2 and Housing 9. The black line outlines the canopy structure.

In the understorey scenarios model runs, the ammonia concentration can vary significantly depending on the canopy density (LAD and LAI). Indeed, with a quite open canopy (Understorey 5, LAI=3), the maximum concentration reaches a level similar to the maximum concentration in the housing case, but when the canopy is very dense (Understorey 6, LAI=6), the concentration is much larger and reaches more than 4000 µg NH_3 m^{-3} (Figure 4.7). This can be explained by the very small level of turbulence and low wind speed in the canopy in the dense scenario, hence leading to the accumulation of high NH_3 concentrations.

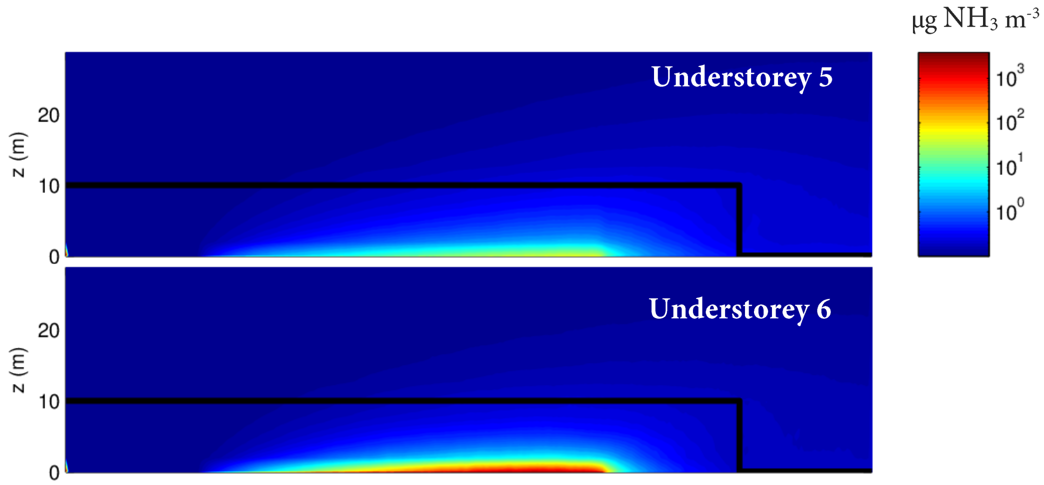


Figure 4.7. Output from MODDAS-THETIS showing the concentration field in "under-storey" model runs Understorey 5 (upper panel) and Understorey 6 (lower panel) with varying LAI 3 and 6 m2 m 2 respectively.

4.5 Deposition patterns

The NH_3 deposition patterns in the housing scenarios follow the concentration patterns but are also affected by the LAD patterns (Figure 4.3). Figure 4.8 illustrates the difference of having no back-stop (top panel) compared with a 50m back-stop (lower panel). Interestingly, deposition to main canopy structures with lower LAIs (LAI=3) is estimated to have higher deposition rates (15%) than denser back-stop canopies (LAI = 6) of a similar length (12%) as the main canopy is sufficiently long to capture most of the ammonia (Housing 9). This is also due to the concentration being much larger near the source than in the backstop.

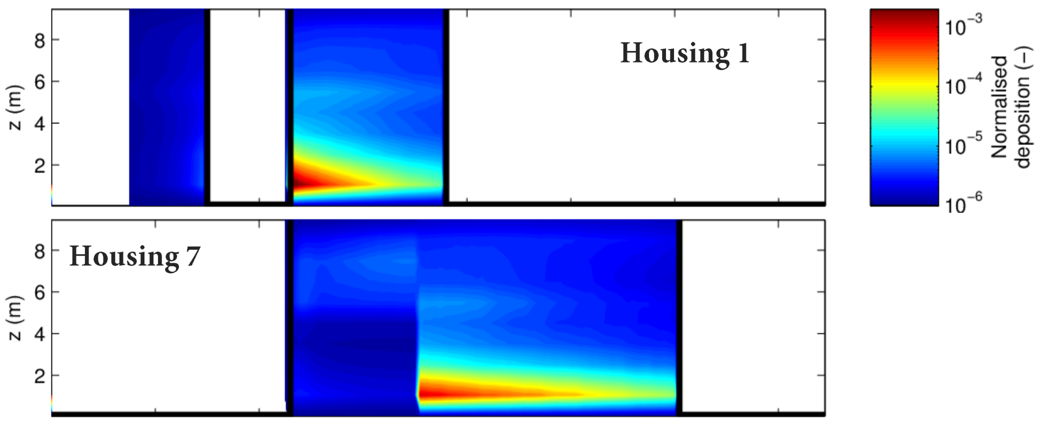


Figure 4.8. Output from MODDAS-THETIS showing the deposition patterns in Housing 1 and Housing 7. The colours show NH_3 deposition to the canopy normalised by the source strength . The lower panel shows the scenario with the backstop located at 70 m. The maximum colour-scale is 2.10^{-3}.

The deposition pattern in the understorey scenarios varied a lot depending on the concentration levels, and the LAI and LAD patterns. Figure 4.9 illustrates this when comparing a situation with a quite open canopy (LAI= 3 Understorey 5), with a situation with a dense main canopy (LAI=6, scenario 6). The deposition is only significant in the backstop for the less dense canopy (37% recapture - Understorey 5) while it is very large throughout the main canopy and the backstop in the dense canopy scheme (60% recapture – Understorey 6).

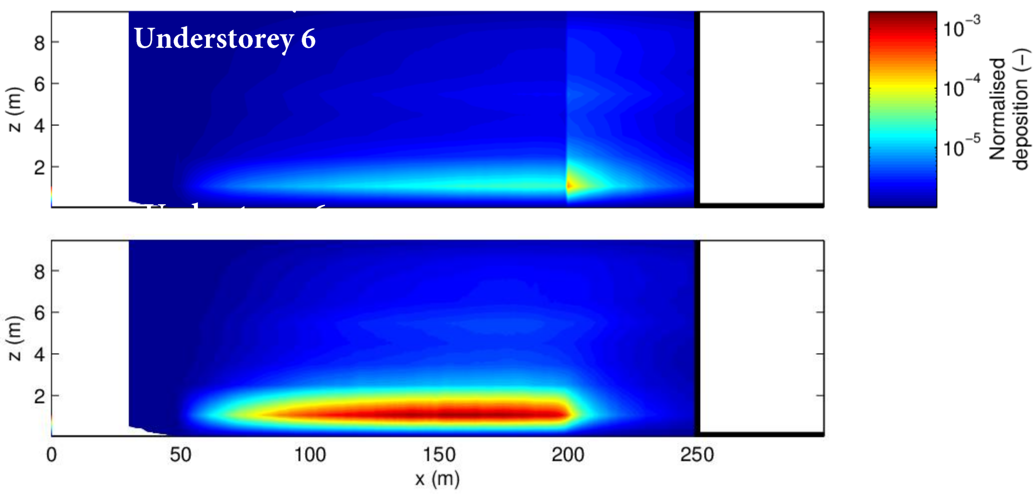


Figure 4.9. Output from MODDAS-THETIS showing the deposition patterns in the understorey model runs – Understorey 5 (upper panel) showing the effect of the backstop with an open main canopy (LAI 3), and Understorey 6 (lower panel) showing the effect of a dense main canopy (LAI 6). The deposition is normalised by dividing by the source strength. The maximum colour-scale is 2.10^{-3} as in Figure 4.8.

4.6 Discussion and conclusions

This study has investigated housing, storage lagoon and understorey emission sources of ammonia and the use of trees to mitigate emissions by planting upwind and downwind of the source.

The modelling results estimate that maximum recapture of ammonia of 27% for housing sources and, 19% for slurry lagoon sources can be attained, while 60% deposition for under-storey systems is possible (although it is noted that the dense canopy would not be suitable for free-ranging chickens). The comparison between housing systems with woodland surrounding the housing unit (symmetrical) and woodland downwind of the source only (asymmetrical) shows that there is little additional deposition upwind of the source, but local meteorological conditions (e.g. wind direction) should be assessed before only planting on one side of a source. However, it may be desirable, due to the need to reduce costs, to plant on the downwind side of a source for predominant wind directions. It would be desirable to plant any woodland structure around a housing source as reduced deposition to semi-natural areas can help to protect sensitive species and habitats from nitrogen deposition effects.

LAI and LAD together with canopy length have the most effect on deposition rates within the range of scenarios tested here. The deposition rate increased roughly in proportion to the LAI, when the LAI and the LAD are identical in the main and the backstop canopies. Optimal designs included backstop structures of high LAI (dense canopy structures) which have the ability to prevent ammonia escaping underneath the canopy and out the sides and back of the canopy. Dense backstop structures were also found to lower the wind velocity on the main canopy allowing a longer residence time and hence a better recapture efficiency. However, main canopies with high LAIs (e.g. LAI 6) also capture significant amounts of ammonia making the necessity for backstop structures less critical. The canopy with a dense and homogeneous LAD favours deposition (LAD 10), while a canopy with a dense crown and an open trunk space is less effective at recapture. However, for the under-storey scenario such dense canopies are not realistic due to the need for livestock to be able to freely roam under the canopy. Therefore the optimal canopy structure for housing and under-storey livestock systems are not the same.

The model behaves consistently with regard to changing the source strength. This means that, in the model, the percentage of ammonia that is recaptured is independent of the source strength. This makes the model adaptable for most farm scenarios making it an effective tool for calculating tree recapture.

Sensitivity analysis has shown that there is a reduction in the recapture efficiency during the winter months for deciduous trees but importantly the coniferous backstop continues to recapture ammonia throughout the year. The most sensitive parameter in the model is the RH showing reductions in recapture during the summer months for both deciduous and coniferous trees. Further analysis is required to test the effect of other relations which can lead to improvements in future versions of the model.

Specifically the optimal housing systems would have a woodland length of mixed LAI of canopy of around 75 m to achieve a deposition rate or recapture efficiency of 25%. A less dense main canopy of around 25 m and a backstop of 50 m match this objective. Slurry lagoons systems are also suited to dense canopy structures near to the ground as the source is very close to the ground. 30 m dense stands can achieve recapture efficiencies of up to 20%. For under-storey systems with free-range chickens a less dense canopy structure (LAI=3 and LAD-2) is required to allow the chickens to roam freely and use areas of dappled sunlight. Furthermore, due to the welfare targets of a maximum of 0.25 birds per metre square for free-range birds, much larger areas of woodland are required to cater for even fairly small flocks. In our scenario 2500 birds can be enclosed in a hectare of forest (100 m x 100 m). With a 100 metre main canopy for the birds to roam under (LAI=3) and a 25 m dense backstop (LAI-6), a 40% recapture efficiency should be attainable with current scenarios.

There are over 1000 IPPC permits in England for pig and poultry installations alone, some of which represent large 'hot spots' of ammonia emissions. Many sensitive ecosystems and protected sites are relatively close to these hot spots (<200 m). Hence ammonia abatement through agro-forestry systems is a relatively simple approach to mitigate some of the impacts of ammonia in the landscape. The measures would complement source mitigation options.

This work has provided the first qualitative scenario modelling, building on work by Theobald *et al.* (2004), and provides a basis for developing better tools to plan on-farm abatement measures.

4.7 Acknowledgments

I acknowledge Stéphane Glockner from the Institut de Mécanique et d'Ingénierie de Bordeaux I2M (UMR 5295), Département TREFLE, Equipe Mécanique des Fluides Numérique, for providing us with the THETIS/THETIS model (http://thetis.enscbp.fr). We also acknowledge the UK Department for Environment, Food and Rural Affairs (Defra) for funding this research.

Chapter 5. The potential for tree planting strategies to reduce local and regional ecosystem impacts of agricultural ammonia emissions

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5.1 Introduction

By 2020, it is estimated that ammonia will be the largest single contributor to the nutrient nitrogen and acid deposition, and secondary particulate matter formation in Europe (Reis *et al.*, 2015). Emissions of Ammonia (NH_3) have increased substantially during the 20th century. Globally since 1970, world population has increased by 78% and reactive nitrogen creation has increased by 120% through the intensification of agriculture including fertiliser use and livestock production (Galloway *et al.*, 2008). By 2050 the global emission of reactive nitrogen is projected to be 200 Tg N yr, while back in 1860 it was estimated at 34 Tg N yr-1 (Galloway *et al.*, 2004). Environmental impacts from nitrogen and particular ammonia are caused by the loss or leakage of reactive nitrogen as it is volatilized into the atmosphere. Bouwman *et al.* 2002 estimated that NH_3 loss from global application of synthetic N fertilizers accounts for 78 million tons N per year, and animal manure 33 million tons N per year, amounting to 14% and 23% losses respectively.

In the UK, agricultural practises currently accounts for over 80% of NH_3 emissions (Sutton, *et al.*, 2001; Misselbrook *et al.*, 2010). Five main categories of agricultural management activities can be identified as key sources of ammonia: emissions from housing, grazing, storage and manure spreading, and fertiliser use (Misselbrook *et al.*, 2010). Ammonia emissions at the local scale vary greatly within the landscape and dry deposition of ammonia occurs especially close to sources (Hellsten *et al.*, 2008; Dragosits *et al.*, 2002). As a consequence, nitrogen sensitive ecosystems close to sources are at a high risk of negative impacts. Impacts of excess nitrogen can include eutrophication and acidification effects which can lead to species composition changes (Bobbink *et al.*, 2010; Pitcairn *et al.*, 1998; Sheppard *et al.*, 2008; Van den Berg *et al.*, 2008; Wiedermann *et al.*, 2009) and other deleterious effects. Species adapted to low N availability are at a greater risk; for example, many slower-growing lower plants, notably lichens and bryophytes. (Pearce and van der Wal, 2002; Bobbink *et al.*, 2010).

A large number of abatement methods already exist for reducing ammonia emissions from agriculture (Bittman *et al.*, 2014). These include animal housing techniques like drying manure, decreasing the surface area fouled by manure and 'scrubbing' ammonia from the exhaust air of livestock houses; livestock feeding strategies where low-protein feeding is carried out; improving manure storage through covering and encouraging crusting; and using low emission manure spreading through injection or band application. Alternative options like agro-forestry have received less attention and pollution regulators and the livestock industry are increasingly interested in alternative abatement techniques that reduce the effects of nitrogen deposition on nearby protected sites.

Trees are very effective at capturing both gaseous and particulate pollutants from the atmosphere (Beckett 2000; Nowak, 2000; Novak *et al.* 2014; McDonald *et al.*, 2007; Cohen *et al.*, 2014). Deposition rates are far greater to forest than those of short vegetation e.g. grassland, by a factor of 3–20 times (Gallagher *et al*, 2002; Fowler *et al.*, 2004). However, most studies up till now have focused on gases and particulates (e.g. NO_x, $PM_{10}/_{2.5}$) in relation to improving urban air quality. There is a paucity of studies examining the capability of trees to capture ammonia from agricultural sources to protect sensitive habitats. Converting agricultural grassland or arable land to trees near emission sources can be seen as a way to increase the removal of ammonia from the atmosphere, thereby reducing the potential impacts on nearby sensitive ecosystems.

To examine this removal through scavenging of ammonia by trees across the UK, a Lagrangian national-scale atmospheric dispersion model (FRAME) was used to compare two strategies:

1. The first strategy (Strategy A) estimated the potential effectiveness of implementing local, on-farm, tree planting schemes to capture ammonia. One planting scheme was to place tree belts downwind of animal housing and storage facilities; the other planting scheme was to provide trees as shelter for livestock managed under the trees.
2. The second strategy (Strategy B) was to apply a general afforestation policy across the UK by increasing tree planting, targeting areas of high ammonia emissions.

5.2 Methodology

The first approach for reducing on-farm emissions (Strategy A) was to make use of existing estimates of percentage NH_3 recapture from trees downwind of housing and storage systems (20%), and percentage NH_3 recapture from trees with the livestock managed under the trees (45%). Using these recapture percentages a set of revised emission factors for all livestock types and management systems were developed. Finally, with these new 'on-farm' emission factors eight different scenarios (A_1 to A_8) were designed for testing with the FRAME model.

Although the reduction in Strategy A is actually associated with the trees capturing ammonia, this was implemented in the model by modifying the emission factors of each livestock type instead. In effect, the emission reduction occurs as a reduction of the whole on-farm system for a constant unit output, as ammonia is captured before being dispersed outside the 'farm boundaries'.

To assess the influence of a general afforestation strategy (Strategy B) on the re-capture of ammonia, three land cover scenarios were tested in the model. These consisted of the baseline scenario (B_0) and two planting scenarios – increasing total forest cover by 25%

(B_1) and 50% (B_2), respectively, across the UK. In addition to this, tree planting was targeted near emission sources where ammonia concentrations are highest and thus maximise re-capture potential. Only arable and grassland were converted to forests, with the other land cover categories (e.g. moorland and urban) remaining unchanged. Tree cover was increased by scaling the existing forest cover in model grid squares targeted due to high levels of ammonia emissions (or by adding new forest in grid squares with no tree covers).

To summarise, the key steps were to generate new emission factors for agro-forestry systems (Strategy A) and increased tree cover scenarios (Strategy B) for application in an atmospheric transport model, taking into account the effect of NH_3 recapture by trees.

In both scenarios it should be noted that the FRAME model does not take into account deposition to different tree species. Dry deposition is calculated to 5 land classes of which forest is one (arable, forest, moor-land, grassland and urban). For ammonia, deposition is calculated for each grid square using a canopy resistance model (Singles *et al.*, 1998). Deposition velocities are therefore generated from the sums of the aerodynamic resistance, the laminar boundary layer resistance and the surface resistance as well as the geographical and altitudinal variation of wind-speed.

The following sections describe the methodology in more detail.

5.3 Atmospheric dispersion modelling

The FRAME (Fine Resolution Atmospheric Multi-species Exchange) model (Singles *et al.*, 1998; Fournier *et al.*, 2004; Dore *et al.*, 2007; *Vieno et al.*, 2007; Dore *et al.*, 2012) was applied at a 1 km grid resolution across the British Isles to assess the influence of both abatement strategies on ammonia concentrations in air and the deposition of reduced nitrogen. FRAME is a Lagrangian atmospheric transport model developed to output annual mean deposition of reduced and oxidised nitrogen and sulphur. The model uses rainfall and wind speed inputs, (Dore *et al.*, 2006) as well as emission and land cover data and has been used to assess the environmental impact of nitrogen deposition (Matejko *et al.*, 2009). FRAME has been used to model pollutant deposition over Europe, the UK, Poland and parts of China.

FRAME at the 1km grid resolution has been used to assed critical level exceedance of ammonia over the UK's Natura 2000 sites (Special Protection Areas and Special Areas of Conservation) (Hallsworth *et al.* (2010)).

This study uses emission data from the 2008 National Atmospheric Emissions Inventory (NAEI) for SO_2, NO_x and non-agricultural NH_3. For agricultural NH_3, the Atmospheric Emissions for National Environmental Impacts Determination (AENEID; used for annual UK maps for the NAEI; Dragosits *et al.* 1998; Hellsten *et al.* 2008) was used for developing the detailed emission scenarios. The AENEID model redistributes agricultural emissions across the landscape by weighting the source strength of five broad management activities - livestock grazing, livestock housing, manure storage, land-spreading of manures and mineral fertiliser application. Emission source strength data (emission factors) are calculated annually for the UK agricultural emission inventory (Misselbrook *et al.* 2010). The spatial distribution of ammonia emissions from agricultural sources for 2008 is illustrated in Figure 5.1.

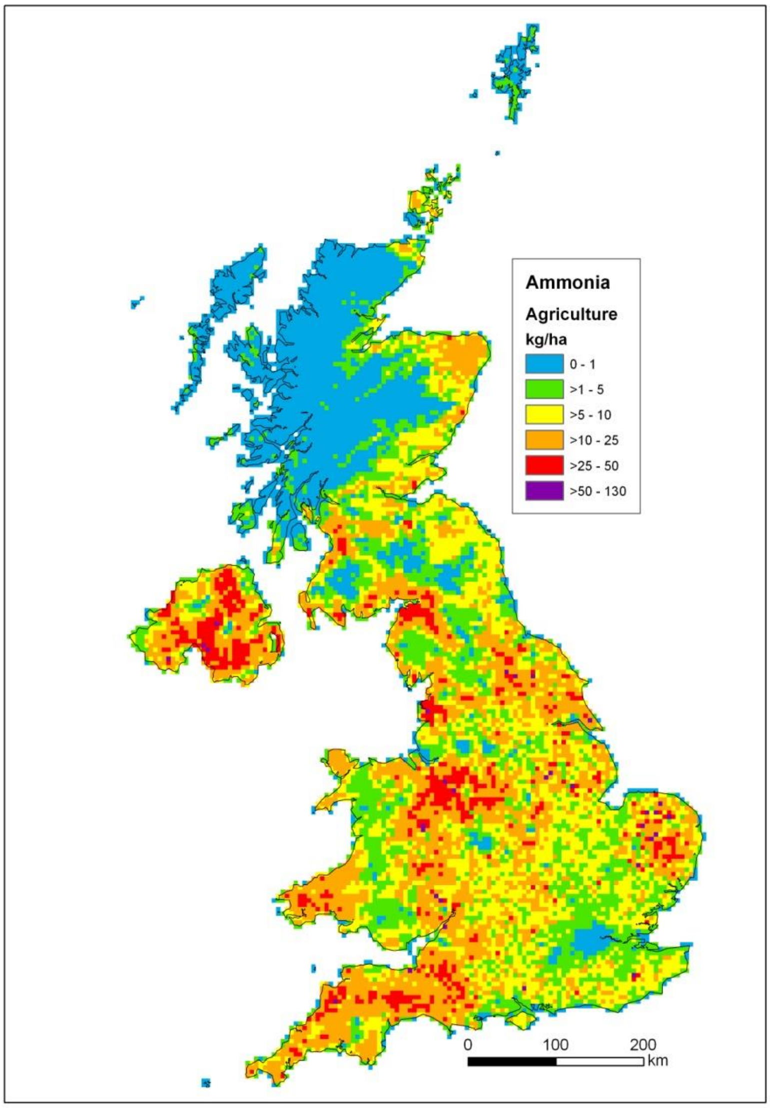

Figure 5.1. Emissions of ammonia from agricultural sources in the UK for the year 2008 (5 km grid resolution).

5.3.1 Strategy A - Revision of 'on-farm' emission factors

In a prior analysis we used the *MODDAS-THETIS* model to assess the optimum tree canopy structures for capturing ammonia from livestock farms (Bealey *et al.* 2014 in press). We assessed three farm management practices – NH_3 emissions from housing, slurry lagoons, and livestock living under the tree canopy. By changing model parameters such as width of canopy, leaf area index and leaf area density, optimal tree structure configurations for capturing ammonia were established for each management practice.

The capture efficiencies represent the extra amount of ammonia deposited in the tree canopy that would not have been deposited if the tree canopy had not been there. It is therefore an extra deposition above what would normally deposit at this distance from a farm if the land-use was not changed to trees (e.g. grassland or arable crops).

The following percentage NH_3 capture efficiencies were then used to recalculate the livestock emission factors for use in the modelling:

- 20% NH_3 capture efficiency for housing emissions which were representative of a 10m tall tree canopy, with a 25 m long main canopy (LAI 3) and a 25 m dense backstop canopy (LAI 6).
- 20% NH_3 capture efficiency for storage emissions which were representative of a 10m tall tree canopy, with a 30 m long main canopy (LAI 6).
- 45% NH_3 capture efficiency for livestock under-canopy silvo-pastoral farming systems (i.e. grazing emissions) which were representative of a 10m tall tree canopy, 100 m main canopy (LAI 3), and a 50 m dense backstop canopy (LAI 6)

In order to parameterise this effect in the FRAME model reduced emission factors were calculated for each livestock type. Table 5.1 shows the calculations of revised emission factors for the key livestock types.

For laying hens, a number of tree belt options were considered. This included a basic option to provide a tree shelter belt downwind of the housing to capture ammonia (i.e. 20% housing emission reduction). The option of having free-range laying hens under a tree canopy was also calculated (45% grazing emission reduction), with a final advanced option of having both the housing (in the form of small arks) covered with a tree belt, and free-range laying hens under the tree canopy. This system gave a reduction in the 'on-farm' emission factors of both the housing and grazing by 45% each. For other poultry types the same calculations were carried out to derive reduced emission factors. This included broilers, turkeys, pullets (young laying hens), and a summary category of 'other poultry' (which includes, ducks, geese, guinea fowl and other species less common in the UK).

For sows, around 36% of the herd are kept outdoors already. Therefore emission reduction was calculated based on doubling this to 72%, with grazing pigs also living under the tree canopy. This gave a 45% reduction in the grazing emission factor, and at the same time a reduction in the housing emission factor. This process was repeated for other pig categories.

For cattle, a similar approach was taken as for the laying hens. Reduced emission factors using trees to capture housing and slurry storage emissions were calculated. Cattle grazing under trees as a management system were not considered for an emission-factor reduction, mainly due to the requirement for very low stocking densities. However, cattle grazing under trees are used for conservation reasons and are deployed by many conservation organisations (Armstrong *et al.*, 2003).

Table 5.1. Emission factor reductions for livestock types using two tree planting scenarios, of 45% for grazing under trees, and 20% for planting trees around housing and manure storage units. The full table can be seen in Annex: Table 5.6.

Livestock Type	Management System	Housing %NH_3 Capture Efficiency	Grazing %NH_3 Capture Efficiency	Storage %NH_3 Capture Efficiency	Current (2008) total Emission Factor	Revised total Emission Factor	Total % emission reduction
Laying hens	In housing upwind of tree belt, no ranging	20%	0%	0%	0.264	0.233	**12%**
Laying hens	In housing upwind of tree belt + 25% ranging* under trees	20%	45%	0%	0.264	0.194	**27%**
Laying hens	In housing under tree canopy (arks) + 25% ranging* under trees	45%	45%	0%	0.264	0.165	**38%**
Sows	Double the number of sows outdoors (currently 36%) + ranging under trees	0%	45%	0%	5.242	2.844	**46%**
Other pigs >80-110 kg	Increase to 15% the herd outdoors (currently 0.01%) + ranging under trees	0%	45%	0%	5.310	4.857	**9%**
Other pigs >50-80 kg	Increase to 15% the herd outdoors	0%	45%	0%	4.580	4.180	**9%**

Livestock Type	Management System	Housing $\%NH_3$ Capture Efficiency	Grazing $\%NH_3$ Capture Efficiency	Storage $\%NH_3$ Capture Efficiency	Current (2008) total Emission Factor	Revised total Emission Factor	Total % emission reduction
	(currently 0.01%) + ranging under trees						
Other pigs >20-50 kg	Increase to 15% the herd outdoors (currently 0.01%) + ranging under trees	0%	45%	0%	3.060	2.815	**8%**
Dairy cows & heifers	In housing upwind of tree belt, no ranging + slurry store with trees downwind	20%	0%	20%	26.173	22.688	**13%**

* The 25% ranging value was calculated based on personal communication from poultry farmers

5.3.2 Scenario modelling

For Strategy A the revised emission factors were applied to eight scenarios covering all livestock types with the aim of showing the benefit of using trees for reducing emission source strength. The scenarios were:

- A_1: applied to 50% of the UK poultry flock trees downwind of their housing.
- A_2: 37% of the laying flock (currently the number which is free-range in the UK) were put under trees and at the same time their housing was sheltered with tree belts.
- A_3: of the 37% free-range poultry, 30% had their housing placed under the tree canopy (in arks).

- A_4: 50% of the entire UK poultry flock (around 110 million birds) had their housing sheltered with tree belts while a further 10% were allowed to range under the tree canopy.
- A_5: a combination of Scenarios A_1-A_4,
- A_6: made 20% of all cattle housing and their associated manure storage to be sheltered by trees.
- A_7: doubling (to 72%) the proportion of sows living outdoors and providing trees as shelter, 15% of the pigs were put outdoors under trees.
- A_8: a combination of Scenarios A_5, A_6 and A_7 to model the effect of a large scale implementation of grazing livestock under trees and sheltering their housing and manure storage with tree belts.

For Strategy B - national scale afforestation scenarios - a summary of the scenarios is given below:

- B_0 – baseline scenario
- B_1 – increasing total forest cover by 25%
- B_2 – increasing total forest cover by 50%

5.4 Results and Discussion

5.4.1 Strategy A: 'On-farm' emission source strength reductions

Table 5.2 summarises the percentage change in emissions based on the scenario descriptions above for the three main livestock types. The total change in NH_3 emissions across the whole livestock sector and as a percentage change across the UK was calculated. A full list of emission changes for each scenario can be found in the Annex: Table 5.7.

Table 5.2. Summary table showing the percentage change in NH_3 emissions across individual livestock types, total livestock as a whole, and the overall change (kt NH_3) in UK NH_3 emissions from all sources.

Scenario	Cattle % (kt NH_3)	Pigs % (kt NH_3)	poultry % (kt NH_3)	total livestock %	total national NH_3 emission %
A_1	-	-	-4.2% (1.3)	-0.7%	-0.5%
A_2	-	-	-2.5% (0.8)	-0.4%	-0.3%
A_3	-	-	-2.9% (0.9)	-0.4%	-0.3%
A_4	-	-	-1.9% (0.6)	-0.3%	-0.2%
A_5	-	-	-8.3% (2.6)	-1.3%	-0.9%
A_6	-2.6% (3.4)	-	--	-1.7%	-1.2%
A_7	-	-12.6% (2.5)	-	-1.3%	-0.9%
A_8	-2.6% (3.4)	-12.6% (2.5)	-8.3% (2.6)	-4.3%	-3.0%

Emission changes from carrying out partial, but fairly wide scale abatements e.g. A_1 putting trees downwind of half the poultry sheds in the UK results in only a small national reduction in ammonia emissions of 1,293 tonnes of NH_3. This is largely due to the small emission factor for poultry. The emissions are doubled with scenario A_5 where all scenarios A_1-A_4 were applied. The A_5 scenarios resulted in a 8.3% reduction in poultry emissions (2.6 kt). Applying tree planting around 20% or cattle sheds and storage resulted in a 2.6% reduction to total cattle emissions representing 3.4 kt of ammonia captured nationally. Doubling the pig population to outdoors gave 12.6% reduction representing 2.5 kt recaptured by the trees. The final scenario combined all livestock scenarios (A_1-A_7) and provided the highest emission reductions (8.4 kt, 4.3% of the total livestock population). Nationally the percentage reductions are small (0.5% to 3%) with respect to the total emissions. One might conclude that quite a lot of tree planting is required for small gains, but that tackling the largest emitters (e.g. cattle and pigs) should be the main target for reducing emissions. However, applying a combination of scenarios as set out in A_8 can significantly reduce emissions below future 2020 threshold limits set by UNECE (UNECE, 2012) or in Europe by the National Emissions Ceiling Directive (NECD) (Council Directive 2001/81/EC). Figure 5.2 shows the emission scenarios (A_1-A_8) including the current temporal trend (blue line) and the resulting emissions each scenario could achieve by 2030. 2030 was chosen as a suitable future year to achieve realistic growth and size of tree assuming trees were planted by 2020. The UNECE 2020 target for NH_3 in the UK is 283 kt of NH_3 and Figure 2 shows that by applying scenario A_8 this target can be achieved even by 2020.

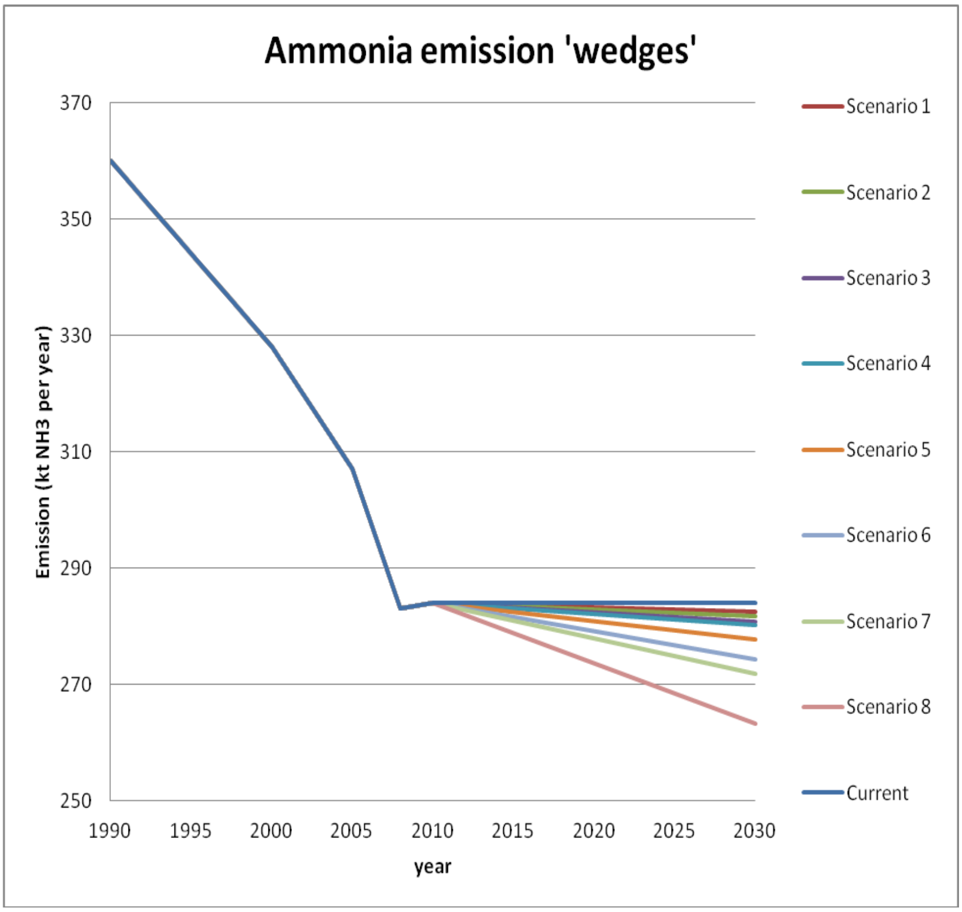

Figure 5.2. Emission scenarios (A1-A8) including the current trend (blue line) and the resulting emissions each scenario could achieve by 2030 for the UK. The emissions are cumulative.

5.4.2 Strategy B: national scale afforestation scenarios

A summary of the changes to land cover is illustrated in Table 5.3. Arable and grassland land cover was reduced for scenarios B1 and B2 to accommodate introduction of new tree plantings in targeted areas of high ammonia emissions.

Table 5.3. Percentage of land cover types for the baseline and 25% and 50% afforestation scenarios.

SCENARIO	% arable	% forest	% grass	% semi-natural ecosystems	% urban	water
B_0 BASELINE	23.0	11.7	22.3	33.8	6.6	2.6
B_1 + 25%	21.7	14.7	20.6	33.8	6.6	2.6
B_2 + 50%	20.4	17.6	19.0	33.8	6.6	2.6

The spatial distribution of forest cover for the baseline scenario and the change between the baseline and the +50% scenario are illustrated in Figure 5.3. 11.7% of forest in the UK represents around 2.8 million hectares.

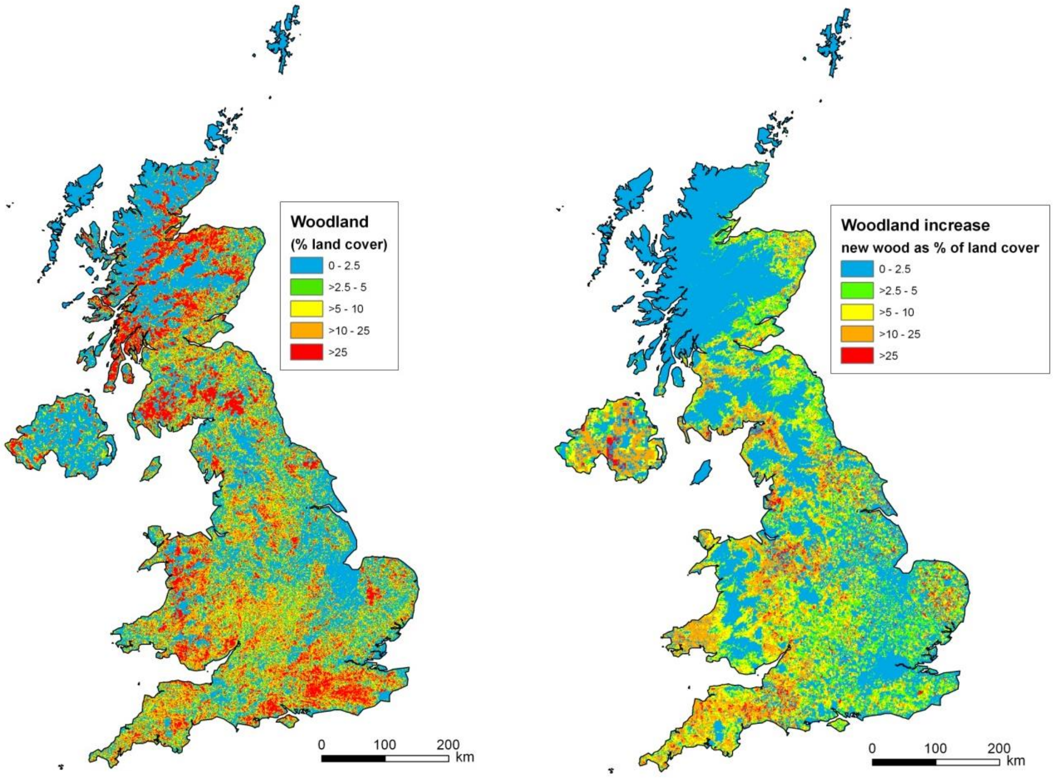

Figure 5.3. Forest distribution in the UK. Percentage of land cover which is woodland for the baseline scenario (left); Percentage of land which is new woodland for the +50% scenario (right)

5.4.3 Atmospheric dispersion modelling

Strategy A

Table 5.4 shows the percentage reduction in nitrogen deposition for each scenario across the UK. Scenarios 1-4, covering the poultry sector, show small reductions in total nitrogen deposition even though the woodland systems were applied to over half, in some cases, of the total UK flock. This is due to the low emission factor for poultry as a whole, even though there are over 160 million birds in the UK. However, Scenario A5 (all poultry scenarios 1-4 are included) has a higher reduction of 0.62%. For the cattle sector a total NH_3 emission reduction of 0.95% is achievable with placing woodland structures around 20% of the cattle housing around the UK and 20% of the slurry stores. Doubling the number of outdoors sows together with foraging under trees (36% to 72%) and putting a percentage (15%) of other pigs under trees reduces N deposition by 0.64%. The best reduction in total nitrogen deposition is achieved by the combination of all scenarios, at 2.2%.

Table 5.4. Percentage change in total nitrogen deposition from each emission reduction scenario

SCENARIOS	% (kt N-NH_x) reduction in total N (grid average)
SCENARIO A1 POULTRY - 50% of all poultry houses sheltered	0.3% (0.45)
SCENARIO A2 POULTRY - housing sheltered and foraging under trees	0.2% (0.28)
SCENARIO A3 POULTRY - Birds ranging under trees, 70% houses sheltered, 30% in arks under trees	0.2% (0.3)
SCENARIO A4 POULTRY - broilers (60% houses sheltered, 10% forage under trees)	0.14% (0.2)
SCENARIO A5 POULTRY (combination of Runs 1-4)	0.62% (0.9)
SCENARIO A6 Dairy+ Beef (20% of cattle houses and slurry stores sheltered)	0.95% (1.35)
SCENARIO A7 PIGS (72% of sows and 15% of other pigs foraging under trees)	0.64% (0.91)
SCENARIO A8 COMBO (SC5 Poultry, SC6 Cattle and SC7 Pigs)	2.2% (3.15)

For all scenarios both wet deposition and export of nitrogen deposition from the UK are reduced since more ammonia is captured in the tree canopy by dry deposition processes. The A_8 scenario resulted in a 2% reduction in both wet deposition (1.5 kt N-NH_x) and export (3.3 kt N-NH_x) compared to the base run.

Strategy B

The national reduced nitrogen (NH_x) budget for the three scenarios is illustrated in Table 5.5. The two tree planting scenarios (25%, 50%) result in significant changes to the fate of emitted ammonia, resulting not only in significant increases in dry deposited NH_x (to forest) and decreases in wet deposited NH_x, but also in decreased export of NH_x in air leaving the UK (which contributes to the long range transport of air pollution in Europe). Changes in NH_x deposition and export for tree planting scenarios B1 and B2 are expressed as percentages relative to the baseline scenario. It can be seen that the influence of a 50 % national scale increase in forest cover in the UK targeted at high ammonia emissions areas would result in a 19.5% increase in total dry N deposition, a decrease of 4.6% in total wet N deposition and a 6.8% decrease in the export of reduced nitrogen from the UK.

Table 5.5. The UK mass deposition and export budgets for simulations B_0, B_1 (+25%) and B_2 (+50%) showing reductions in dry, wet and total nitrogen deposition.

Gg N-NH_x	**B_0 BASELINE**	**B_1 + 25% forest**	**B_1 reduction (%) + 25% forest**	**B_2 + 50% forest**	**B_2 reduction (%) + 50% forest**
Dry Deposition	61.5	68.0	6.4 (10.4%)	73.5	12 (19.5%)
Wet Deposition	81.1	79.1	-2 (-2.4%)	77.4	-3.7 (-4.6%)
Total Deposition	142.6	147.1	4.5 (3.2%)	151.0	8.4 (5.9%)
Export	121.4	116.9	-4.5 (-3.7%)	113.1	-8.3 (6.8%)

The results from FRAME for the baseline scenario for ammonia concentration in air as well as deposition of reduced nitrogen are illustrated in Figure 5.4.

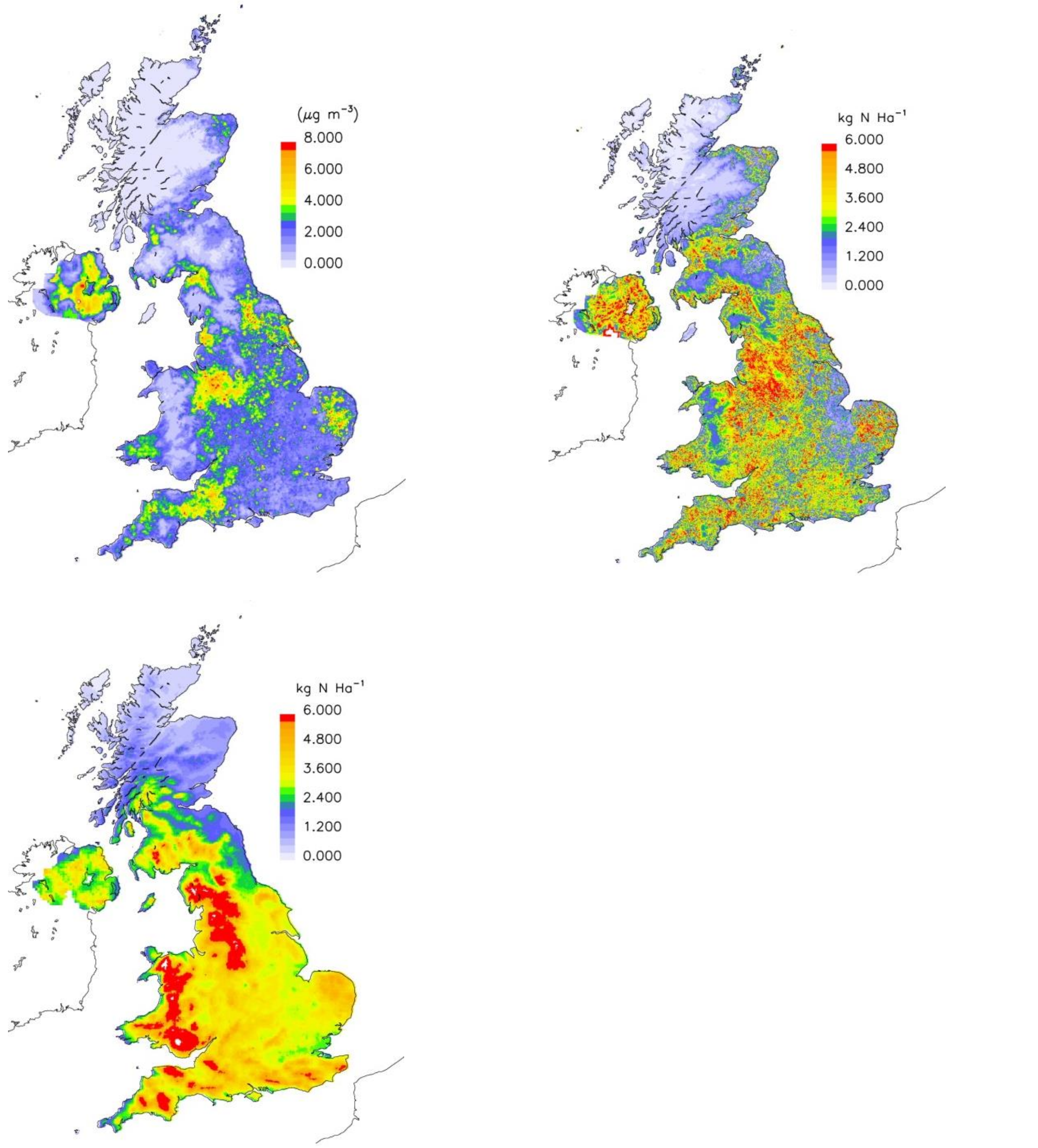

Figure 5.4 Baseline Scenario: Modelled concentration of NH_3 in air (top left); Dry deposition of NH_x (top right); Wet deposition of NH_x (bottom left)

Agricultural ammonia concentrations in the UK are highest across areas of cattle farming in the western parts of the country (in particular NW, W and SW England, SW Wales and Northern Ireland), as well as in localised hot spots around intensive pig and poultry farms (mainly NE and E England). This distribution is closely reflected in the patterns of dry deposition of NH_x, which is primarily due to the deposition of locally emitted ammonia gas. A different pattern is evident for wet deposition of NH_x, due to the chemical transformation of ammonia gas to ammonium aerosol and resulting long range transport. Wet deposition is highest in the high precipitation upland areas of Wales and the Northern England.

The modelled scenarios with increased woodland led to an increase in NH_x dry deposition near the emission sources (12 kt N-NH_x (19.5%)) due to the lower canopy resistance of forest compared to the land cover types which it replaced (grassland and arable). The reduced availability of ammonia gas in the atmosphere away from emission sources therefore resulted in decreases in NH_x wet deposition and in NH_x dry deposition to sensitive ecosystems.

Figure 5.5 illustrates the decrease in NH_x deposition resulting from implementation of scenario B (50% national increase in forest cover). Significant reductions in nitrogen deposition were achieved with this scenario. In areas of high wet deposition (the NW England and Wales), the reduction in wet deposition was up to 0.5 kg N ha^{-1} yr^{-1}. Higher decreases of up to 2 kg N ha^{-1} yr^{-1} for dry deposition were achieved for large areas of semi-natural land and forest. While the deposition per unit area of forest decreased, it is important to note that total mass of NH_x deposited to forest increased due to the national increase in forest area. This is generally considered to be beneficial, as deposition would be directed to the new plantation forests in agricultural areas, consequently reducing the impact on established semi-natural forest ecosystems.

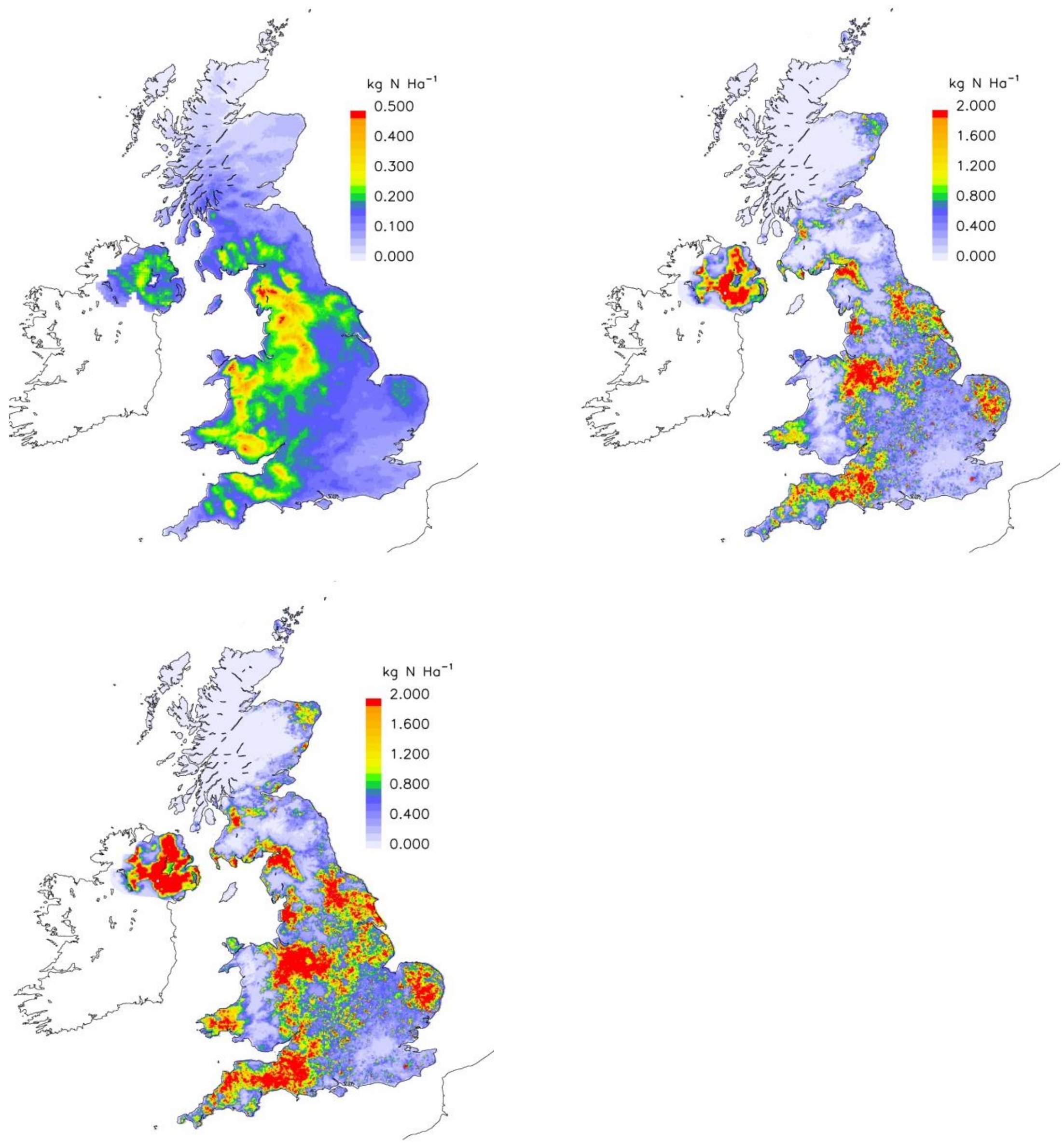

Figure 5.5. Areas and amounts of total nitrogen deposition that is reduced from a 50% increase in forest cover (B2): Wet deposition (top left); deposition to semi-natural non-forest land (top right); deposition to semi-natural forest (bottom left)

The two strategies described in this paper are in some ways quite similar – they both use the concept of planting trees to re-capture ammonia thereby protecting nearby semi-natural areas. Since both strategies have the ability to pinpoint where trees are planted they can be used to control where the ammonia deposits as a means to reduce inputs to semi-natural areas (e.g. downwind of animal housing units, storage facilities and spreading areas). However, while the two approaches are similar in their aims and N deposition reduction, they are quite different in their approach, application and amount of trees planted. Strategy A uses discreet blocks of woodland to capture ammonia around targeted ammonia hot-spots including livestock housing and manure storage, as well as directly placing livestock under the trees. Strategy B, while also targeting hot-spots, uses more of a blanket approach to distributing the trees in the landscape. Strategy A can be seen as a farming management switch to grazing livestock under trees and a sheltering of housing units with tree-belts. By contrast, Strategy B is more of a farm-forestry management technique that will not only capture ammonia to protect semi-natural areas, but also has the potential to provide timber products (e.g. for use as renewable fuels) and/or to improve carbon sequestration (increasing national carbon sinks) on a much greater scale than Strategy A. Both strategies augment the afforestation targets for the UK. Strategy B amounts to planting around 0.7 million hectares of trees for a 25% increase in forest, to 1.4 million hectares for a 50% increase in forest. Conversely for Strategy A much smaller areas of land are converted to trees. For example if the 26 million laying hens in the UK were converted to silvo-pastoral systems this would create around 10,000 ha of reforested land (stocking rate of 2500 birds/ha). 27,500 ha of new woodland could support the broiler population (110 million birds) in this way too (stocking rate of 4000 birds/ha).

One key point to be made is that both strategies are not actually reducing total emissions, but they are reducing on-farm emissions, and in both cases trees can be used as sacrificial land-use with the aim to buffer sensitive habitat areas in the landscape near agricultural areas.

5.5 Conclusions

Both strategies reduce nitrogen deposition to semi-natural areas, both target areas of high ammonia emissions, and both strategies lead to the reduction in wet deposition and the export of nitrogen out of the UK as more is captured at source by the trees. Scenario A_8 of Strategy A (the combination scenario) achieves around a 3.1 kt N-NH_x (2.2%) reduction in total nitrogen deposition across the UK, about the same as Strategy B of planting 25%

more trees in the vicinity of ammonia hotspots. For Scenario A_8 wet deposition was reduced by 1.5 kt N-NH_x (2%) and export reduced by 3.2 kt N-NH_x (2%).Planting 50% more forest in Strategy B resulted in a 12 kt of N-NH_x (19.5% increase) being deposited to the planted areas. By increasing dry deposition to the planted areas it also gave an added value effect of reducing wet deposition by 3.7 kt N-NH_x (4.6% reduction) and reducing export from the UK of 8.3 kt (6.8% reduction).

In both strategies the higher cost of transferring arable land and grassland to forest land cannot be understated in terms of income, animal feed production, and crop harvests forgone as more trees are planted. Strategy A is certainly more suitable for the livestock industry to implement as it is more targeted and involves planting smaller discreet blocks of trees around sources. Strategy B has a more blanket approach to planting around the farm which could give far reaching implications for current food production as prime agricultural land is replaced by forestry. Managing nitrogen losses on the farm and improving the efficient use of nitrogen are the key components for overall reduction in NH_3 emissions. Planting trees around hot-spots of ammonia can reduce the potential impacts on nearby sensitive ecosystems and have added benefits of reducing long-range transport.

5.6 Acknowledgments

I also acknowledge the UK Department for Environment, Food and Rural Affairs (Defra) for funding this research.

5.7 Appendix

Table 5.6 Emission factor reduction for livestock types using two tree planting scenarios - livestock grazing under trees (45% reduction in NH_3), and sheltering housing units and manure stores with trees (20% reduction in NH_3).

		Housing			Grazing			Storage and Spreading					
Livestock Type	**Management System**	**% Housing NH_3 Capture Efficiency**	**% time indoors**	**Housing Emission Factor**	**% Grazing NH_3 Capture Efficiency**	**% time outdoors**	**Grazing Emission Factor**	**% Storage NH_3 Capture Efficiency**	**% housing manure required for storage and spreading**	**Storage & spreading Emission Factor**	**Revised Total Emission Factor**	**Current 2008 total Emission Factor**	**% emission reduction**
Laying hens	*Control: full-time in housing, no free-range, no trees	0%	100%	0.155	0%	0%	0.000	0%	100%	0.109	0.264	0.264	**0%**
Laying hens	In housing upwind of tree belt, no ranging	20%	100%	0.124	0%	0%	0.000	0%	100%	0.109	0.233	0.264	**12%**
Laying hens	In housing upwind of tree belt + 25%	20%	75%	0.093	45%	25%**	0.019	0%	75%	0.081	0.194	0.264	**27%**

	ranging under trees											
Laying hens	In housing under tree canopy (arks) + 25% ranging under trees	45%	75%	0.064	45%	25%	0.019	0%	75%	0.081	0.165	0.264	**38%**
Sows	Double the number of sows outdoors (currently 36%) + ranging under trees	0%	28%	0.750	45%	72%	1.072	0%	28%	1.022	2.844	5.242	**46%**
Other pigs >80-110 kg	Increase to 15% the herd outdoors (currently 0.01%) + ranging under trees	0%	85%	2.473	45%	15%	0.203	0%	85%	2.182	4.857	5.310	**9%**
Other pigs >50-80 kg	Increase to 15% the herd outdoors	0%	85%	2.131	45%	15%	0.165	0%	85%	1.884	4.180	4.580	**9%**

	(currently 0.01%) + ranging under trees												
Other pigs >20-50 kg	Increase to 15% the herd outdoors (currently 0.01%) + ranging under trees	0%	85%	1.425	45%	15%	0.135	0%	85%	1.255	2.815	3.060	**8%**
Dairy cows & heifers	In housing upwind of tree belt, no ranging + slurry store with trees downwind	20%	46%	10.628	0%	54%	1.615	20%	46%	10.445	22.688	26.173	**13%**

Table 5.7. Full list of the 8 scenarios used for the FRAME model runs based on three woodland systems

SCENARIOS	Livestock Category	EF reduction	Applicable % of UK flock/herd
Scenario A1 POULTRY **System 1: Housing with trees downwind, no free-range (↓20%)**			
	Laying hens (Sys 1)	12%	50%
	Breeding birds (Sys 1)	8%	50%
	Broilers (Sys 1)	6%	50%
	Pullets (Sys 1)	8%	50%
	Turkeys (Sys 1)	11%	50%
	Other poultry (Sys 1)	8%	50%
Scenario A2 POULTRY **System 2: Housing with trees downwind (↓20%) + free-range under trees (↓45%)**			
	Laying hens (Sys 2)	27%	37%*
Scenario A3 POULTRY – free ranging birds under trees, 70% houses sheltered, 30% in arks under trees. **System2: Housing with trees downwind (↓20%) + free-range under trees (↓45%)** **System3: Housing under trees (↓45%) + free-range under trees (↓45%)**			
	Laying hens (Sys 2)	27%	26%
	Laying hens (Sys 3)	38%	11%

SCENARIOS	Livestock Category	EF reduction	Applicable % of UK flock/herd
	Laying hens (no reduction)	0%	63%
Scenario A4 POULTRY - broilers (50% houses sheltered, 10% forage under trees) **System 1: 50% of broilers' houses sheltered with trees, no free-range (↓20%)** **System 2: Housing with trees downwind (↓20%), + free-range under trees (↓45%)**			
	Broilers (Sys 2)	23%	10%
	Broilers (Sys 1)	6%	50%
Scenario A5 POULTRY **System1: Housing with trees downwind, no free-range (↓20%)** **System 2: Housing with trees downwind (↓20%), + free-range under trees (↓45%)** **System3: Housing under trees (↓45%) + free-range under trees (↓45%)**			
	Laying hens (Sys 1)	12%	63%
	Laying hens (Sys 2)	27%	26%
	Laying hens (Sys 3)	38%	11%
	Breeding birds (Sys 1)	8%	50%
	Broilers (Sys 1)	6%	50%
	Broilers (Sys 2)	23%	10%

SCENARIOS	Livestock Category	EF reduction	Applicable % of UK flock/herd
	Pullets (Sys 1)	8%	50%
	Turkeys (Sys 1)	11%	50%
	Other poultry (Sys 1)	8%	50%
Scenario A6 Dairy+ Beef (20% of cattle houses and slurry stores sheltered) **System 4: Housing with trees downwind (↓20%), + slurry store with trees downwind (↓20%)**			
	Dairy cows & heifers (Sys 4)	13%	20%
	Dairy heifers in calf, 2 years and over (Sys 4)	12%	20%
	Dairy heifers in calf, less than 2 years (Sys 4)	12%	20%
	Beef cows & heifers (Sys 4)	13%	20%
	Beef heifers in calf, 2 years and over (Sys 4)	13%	20%
	Beef heifers in calf, less than 2 years (Sys 4)	13%	20%
	Bulls >2 years (Sys 4)	13%	20%
	Bulls 1-2 years (Sys 4)	13%	20%

SCENARIOS	Livestock Category	EF reduction	Applicable % of UK flock/herd
	Other cattle, over 2 years (Sys 4)	12%	20%
	Other cattle, 1-2 years (Sys 4)	13%	20%
	Other cattle, under 1year (Sys 4)	10%	20%
Scenario A7 PIGS (Double sows outdoor; 15% the rest both with foraging under trees) System 5: Free-range under trees (↓45%)			
	Sows in pig & other sows (sows) (Sys 5)	46%	100%
	Other pigs, >80-110 kg (Sys 5)	9%	100%
	Other pigs, >50-80 kg (Sys 5)	9%	100%
	Other pigs, >20-50 kg (Sys 5)	8%	100%
Scenario A8 COMBINATION (combination of SC5 Poultry, SC6 Cattle and SC7 Pigs)			
	Dairy cows & heifers (Sys 4)	13%	20%

SCENARIOS	Livestock Category	EF reduction	Applicable % of UK flock/herd
	Dairy heifers in calf, 2 years and over (Sys 4)	12%	20%
	Dairy heifers in calf, less than 2 years (Sys 4)	12%	20%
	Beef cows & heifers (Sys 4)	13%	20%
	Beef heifers in calf, 2 years and over (Sys 4)	13%	20%
	Beef heifers in calf, less than 2 years (Sys 4)	13%	20%
	Bulls >2 years (Sys 4)	13%	20%
	Bulls 1-2 years (Sys 4)	13%	20%
	Other cattle, over 2 years (Sys 4)	12%	20%
	Other cattle, 1-2 years (Sys 4)	13%	20%
	Other cattle, under 1year (Sys 4)	10%	20%
	Sows in pig & other sows (sows) (Sys 5)	46%	100%

SCENARIOS	Livestock Category	EF reduction	Applicable % of UK flock/herd
	Other pigs, >80-110 kg (Sys 5)	9%	100%
	Other pigs, >50-80 kg (Sys 5)	9%	100%
	Other pigs, >20-50 kg (Sys 5)	8%	100%
	Laying hens (Sys 1)	12%	63%
	Laying hens (Sys 2)	27%	26%
	Laying hens (Sys 3)	38%	11%
	Breeding birds (Sys 1)	8%	50%
	Broilers (Sys 1)	6%	50%
	Broilers (Sys 2)	23%	10%
	Broilers (remainder, no trees)	0%	40%
	Pullets (Sys 1)	8%	50%
	Turkeys (Sys 1)	11%	50%
	Other poultry (Sys 1)	8%	50%

**37% is the current proportion of free range laying hens in the UK*

Chapter 6. Cost and benefits of agroforestry systems for ammonia abatement

6.1 Introduction

The impacts of nitrogen (N) pollution from agricultural practises are a global problem. Emissions of reactive N have increased throughout the 20th century as the world's population grew and needed to be fed (Erisman *et al.*, 2008). Ammonia (NH_3) is a key form of reactive nitrogen (N_r) pollutant associated with agricultural practices. Air emissions of ammonia across Europe are dominated by the agricultural sector (Sutton *et al.* 1995 ; Misselbrook *et al.* 2013). The main issue arising from this sector is an excess of nitrogen originating from animal manures and fertilizers. Nitrogen contributes to a cascade of environmental effects, including eutrophication of terrestrial and freshwater ecosystems, contamination of groundwater, and acidification of soils and lakes (Galloway *et al.*, 1998; Erisman *et al.*, 2011).

The current levels of nitrogen management of nitrogen is wasteful as 80% of the input of N_r is lost to the environment, resulting in economic costs due to impacts on human health, ecosystems and climate (van Grinsven *et al.*, 2013). At the same time improving the management of nitrogen through reducing emissions and subsequent abatement of effects has an associated cost. Mitigation options currently focus on technical and engineering measures like decreasing nitrogen content in animal feed or improved methods for storage and spreading of animal manures, and urease inhibitors for mineral fertiliser application. The current state-of-the-art for such measures have been set out in two keys documents: the Reference Document on Best Available Techniques for Intensive Rearing of Poultry and Pigs (BREF 07.2003) and the UNECE guidance document (ECE/EB.AIR/120) on preventing and abating ammonia emissions from agricultural sources (Bittman *et al.*, 2014). Both guidance documents focus on measures such as nutritional feeding strategies, animal housing practices, and collection, storage and spreading of manure. These measures have also been categorised by the UNECE into three core groups: Category 1, 2 and 3.

- Category 1 strategies are well researched, considered to be practical or potentially practical, and there are quantitative data on their abatement efficiency;
- Category 2 strategies are promising, but research on them is at present inadequate, or it will always be difficult to quantify their abatement efficiency generally;
- Category 3 strategies have not yet been shown to be effective or are likely to be excluded on practical grounds.

Alternative pollution mitigation options like agroforestry for ammonia abatement have, until now, received less attention. Moreover, pollution regulators and the livestock industry are increasingly interested in alternative abatement techniques that, for example, reduce the effects of nitrogen deposition on nearby protected sites. However, their potential use is becoming more relevant to both the air pollution regulators and the Forestry Commission. Under the Countryside Stewardship scheme (*Countryside Stewardship: get paid for environmental land management, 2015*), a rural development scheme in England and Wales, land managers can receive grants for woodland creation and importantly their application is scored higher where trees are used for ammonia abatement (based on ammonia targeted areas set for certain clusters of protected sites e.g. Special Areas of Conservation (SAC)). Furthermore, planting trees also supports the long terms afforestation strategy for the all UK countries (England, Wales, Scotland and Northern Ireland) where an extra 5-10k hectares of land is targeted to be planted each year by the middle of the century (Defra, 2014; Forestry Commission Scotland, 2009).

National policy options to reduce ammonia are linked to the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP), which includes new emission ceilings as part of the Gothenburg Protocol as well a list of measures to mitigate ammonia under Annex IX. In parallel, the National Emission Ceilings Directive (NECD) (Council Directive 2001/81/EC) of the EU aims to reduce emissions of pollutants that cause acidification, eutrophication and ground-level ozone in order to protect the environment and human health. In addition, large pig (>2,000 production pigs over 30kg and 750 sows) and poultry (>40,000 birds) farms are now regulated through a system of permits under the EU Industrial Emissions Directive (2010/75/EU (IED)). In order to meet these agreements substantial effort has to be placed to assist farmers in adopting new technologies and practices that can reduce NH_3 emissions. The options must therefore be shown to be seen as affordable to the farmer, but also provide significant financial and environmental benefits that outweigh the costs.

While many studies have also shown how trees can capture air pollution, research has often concentrated on particulates and NO_x in the urban environment with the aim of protecting human health (Nowak *et al.,* 2014; Tallis *et al.*, 2011; McDonald *et al.*, 2007; Beckett *et al.*, 1998). By applying similar concepts to the rural landscape, it can be shown that using trees to capture ammonia from intensive livestock practises can be an important added abatement technique for farmers, regulators and the livestock industry (Sutton et al, 2004).

Trees are effective scavengers of both gaseous and particulate pollutants from the atmosphere (Beckett 2000; Nowak, 2000). Due to trees having a higher surface roughness, than, for example crops, they create more turbulence which in turn increases dry deposition to surfaces like leaves and branches. Dry deposition rates to forest canopies can exceed those to grassland by typically a factor of 3–20 (Gallagher *et al.*, 2002, Fowler *et al.*, 2004). Modelling research undertaken by Asman (2008) on the 'entrapment' of ammonia by tree shelterbelts showed that capture of dry deposited gaseous ammonia increased with the height of the shelterbelt and the stability of the atmosphere (favouring neutral conditions), but decreased further away from the source to the shelterbelt.

Within the farm landscape the conversion of grassland and arable land to trees to capture ammonia can be seen as part of an on-farm emission reduction strategy particularly for ammonia sources like livestock housing or manure storage. Further additional benefits can be gained from grazing livestock under the trees themselves. These emission reductions can reduce the potential impacts on nearby sensitive ecosystems and to some extent reduce long-range transport of ammonia.

This chapter examines the potential for trees to capture ammonia from agricultural emissions assessing the cost and benefits of ammonia abatement to the farmer and society as a whole. The study examines the benefits for two approaches - planting tree belts downwind of animal housing and planting trees for livestock managed under the trees (silvo-pastoral systems). It should be noted that although both approaches are compared from a cost effectiveness perspective, the study does not attempt to state whether one system is better than the other or whether livestock housed in barn systems should be moved to silvo-pastoral systems. Although the approaches are compared together they are essentially assessed as separate entities.

Bealey *et al.*, 2014 have quantified the emission abatement of agricultural ammonia that is achievable with a range of different farm woodland tree systems. These range from a 20% reduction in on-farm ammonia emissions by planting trees downwind of a housing installation, to 45% reduction for placing livestock under the trees themselves. By using these reduction factors and by estimating the cost of creating and maintaining these two options, we here quantify the costs involved, the carbon storage and ammonia abatement of these scenarios over 40 years. Using this data we can estimate the cost-effectiveness of the two options in terms the co-benefits of reducing greenhouse gas and ammonia emissions. Finally, we calculate average costs per kg of ammonia saved and compare these estimates with other mitigation options.

6.2 Methodology

Evaluation of the costs and benefits of applying agroforestry options for farmers to mitigate NH_3 emissions is based on two approaches in this study.

1. The costs and benefits that apply to the individual farmer, which account for revenue and private costs only. Accordingly to this approach the costs can then be compared with other ammonia mitigation techniques.
2. The costs and benefits from the perspective of society. This extends to approach 1 to include analysis of the benefits to society through reducing air pollution damage costs and by carbon sequestration in the trees.

Individual decision-makers like farmers are influenced by market forces and prices including costs of production, labour and equipment costs etc. At the same time famers acting as individuals may cause externalities – in this case the release of ammonia from their agricultural activities leading to wide ranging impacts from air quality issues for human health to biodiversity and ecosystem services loss. These externalities can be expressed as social costs.

It is important to point out that the farmer will not receive damage mitigation costs in monetary terms directly. They are a measure of the benefits (in monetary terms) as experienced by society as a whole. The farmer has unfortunately to bear the full brunt of carrying out agroforestry options although financial incentives are present in the form of woodland grant schemes.

The study estimates the cost and benefits of planting two agroforestry systems over a 40 year period. 40 years was seen as a sensible timeframe to allow for tree shelter belt establishment and maximum ammonia recapture efficiency. Trees used in the modelling study were beech and Sitka spruce and these are estimated to have top heights of around 14.5 and 17 metres at year 40 (Forestry Commission Yield Class tables).

6.2.1 Tree Planting Options and example livestock type

Two tree planting design options were compared for two livestock scenarios based on the poultry sector (see Figure 6.1 and Figure 6.2):

- Option 1: planting design to place tree belts downwind of animal housing; Based on 15,000 laying barn birds per unit housing (50 m long), with perchery and deep litter layer. Emission factor per bird of 0.29 kg NH_3/animal place/year

(Environment Agency, 2010). Total emission – 4350 kg NH_3. Area of tree-belt: 0.5 ha

- Option 2: planting design to provide trees as shelter for livestock managed under the trees (silvo-pastoral systems). Based on 2500 birds per hectare[2] free ranging under the canopy all day. Emission factor per bird of 0.212 kg NH_3/animal place/year were based on Option 1 but with less housing emissions and extended grazing. Total emission – 530 kg NH_3. Area of tree belt: 1.875 ha.

Figure 6.1. Schematic of Option 1: a tree planting design of 25 m depth of broadleaves trees with a Leaf Area Index (LAI) of 3, and a 25 m deep dense backstop with a LAI of 6. The aerial representation shows the dense 25 m backstop 'wrapping' around the main canopy. For a 50 metre long housing installation (shed) the overall tree design is 0.5 ha in size. The emissions come from the housing/manure store location.

[2] Article 4(1)(3)(b)(ii) of Directive 1999/74/EC - laying down minimum standards for the protection of laying hens, states that the maximum stocking density should not be greater than 2,500 hens per hectare of ground available to the hens or one hen per $4m^2$ at all times.

Figure 6.2. Schematic of Option 2: a tree planting design 100 m deep with broadleaf trees with a Leaf Area Index (LAI) of 3, and a 25 m deep dense backstop with a LAI of 6. The aerial representation shows the dense 25 m backstop 'wrapping' around the main canopy. For a 1 ha area of free ranging animals the overall tree design is 1.875 ha in size. The emissions come from the livestock under the canopy.

The schematic diagrams above come from Chapter 4 and are based on the MODDAS-THETIS model. Outputs from that chapter showed that ammonia recapture efficiency for options 1 and 2 are estimated at 20% and 40% respectively.

6.2.2 Cost and benefit analysis approach

Cost effectiveness

To be able to judge whether a project is expected to meet its objectives the cost effectiveness of that project needs to be determined. This can simply be the net present value (NPV) which is the sum of all monetised costs and benefits, discounted to a base year (often the present year). A positive sum indicates that the project is beneficial. Alternatively a negative sum indicates the project would provide an overall cost to the farmer or society as a whole. For projects that have an effect on the environment and society (positive or negative) then cost-effectiveness can be expressed as the cost of saving each tonne of pollutant. The calculation for cost-effectiveness is given in equation 1 (DECC, 2014):

$$CE_s = -\frac{NPV - PVC_s}{P_s} \qquad (1)$$

CE_s = Cost effectiveness in sector s

NPV = Net present value of project (£)

PVC_s = present value of the reduction in emissions of pollutant 's' in the sector (£)

P_s = level of reduced emissions of pollutant 's' (carbon or ammonia) in this sector (tCO_2e or kg NH_3)

To assess whether a project is cost-effective it must be compared against a benchmark. For comparing greenhouse gas emissions (GHG) the benchmark is known as the weighted average cost comparator. The weighted average cost comparator represents the maximum amount that is desirable to spend to abate the average tonne of emissions. In this study it is calculated for GHG emissions based on the weighted average price of non-traded carbon (discounted) over the 40 years of the study. For ammonia emissions we can compare the damage cost from ammonia over the same time frame:

$$CC = \sum_{t=1}^{t=Y} \beta_t \, PC_t \frac{E_{A,t}}{E_A} \quad (2)$$

where:

CC = weighted average cost comparator (over Y years)

β_t = discount factor in year t

PC_t = Pollution damage cost (or Non-traded price of carbon) in year t (£/kg NH_3 or £/tCO_2e)

$E_{A,t}$ = abated emissions in year t (kg NH_3 or tCO_2e)

E_A = total abated emissions over lifetime of project (40 years) (kg NH_3 or tCO_2e)

If the cost-effectiveness value is lower than the cost comparator then the emissions, on average, are being abated in a cost-effective way.

Discounting

Due to the costs and benefits occurring over a time period of 40 years the analysis is made comparable in time by using discounted cash flow (DCF) as a way to weight measures over time. Discounting is used to reflect the fact that benefits have greater value in the present than at a distant time in the future. A recommended 3.5% discount rate was applied in this study in line with UK Treasury Green Book (HM Treasury, 2003). The discount rate applied here follows Warren, 1982:

$$A' = A * \frac{1}{(1+r)}^{(i-1)} \qquad (3)$$

Where *A'* is the discounted amount, *A* is the un-discounted amout , *r* is the discout factor as a proportion, and *i* is the time period (years).

Adjusting for relative price changes

Many of the costs in the study are based on 2011 prices. For this reason prices have been inflated at a rate of 2.5% per year up to 2014 prices. This percentage is based on the UK Government's inflation target as a general deflator for future cash flows (HM Treasury, 2003)

The resulting analysis can be calculated as a net present value (NPV) for 2014 where the discounted costs are subtracted from the discounted benefits. By dividing these two values we can also obtain a Benefit/Cost ratio.

6.2.3 Woodland creation costs and maintenance

The costs for creating and maintaining a woodland tree-belt for both Option 1 & 2 are shown in Table 1. These values are converted to values per hectare of project area per year.

Table 6.1. Costs of measures for creating and maintaining woodland structures (£ per ha per year at 2014 prices)

Costs	**Option 1 (housing/lagoon shelterbelt)**	**Option 2 (e.g. livestock under trees)**
Agricultural Opportunity Cost (p.a.)	£ 655	£ 655
Establishment (year 0)	£ 9,076	£6,801

Costs	Option 1 (housing/lagoon shelterbelt)	Option 2 (e.g. livestock under trees)
Management (year 1 onwards)	£24	£24
Fertiliser and spraying (years 1-4)	£102	£102
Fencing (year 4 onwards)	£92	£92
Thinning (year 25, 30, 35 & 40)	£716	£1526
Backstop Maintenance (year 5 onwards)	£11	£11

The agricultural opportunity cost is the cost of losing one hectare of arable land to the trees. The Silsoe Whole Farm Model (SFARMOD) (Nix, 2009 and ABC, 2010) was used to quantify opportunity costs for a range of farm types based on soil and annual rainfall. For the purposes of this study an arable farm on medium soil with an annual rainfall of 600 mm was used as these were the conditions often occurring around intensive pig, poultry, and dairy livestock production facilities in the lowlands.

Establishment costs in the first year include ground preparation, fertilising, spraying, perimeter fencing, planting costs, and managerial oversight. For tree guards the model assumes costs £0.425 per tree and labour costs at £11/hr based on protecting 56 trees per hour. Perimeter fencing costs are taken as £7/m erected to protect against all but deer. In the first 5 years allowance is made for fertilising and spot chemical weeding control (£102). From year five an allowance is made for any renewal to the perimeter fence. From year six a cost of £11/ha/annum is allowed for the maintenance cutting of the backstop. Thinning is carried out on the main canopy only in years 25, 30, 35 and 40. Managerial oversight after year one is charged at £24/ha/ annum to recognize that some organisational effort is required. This is based on an estimate of a fulltime manger (£29,731) divided by 250 ha - a typical farm size.

6.2.4 Woodland creation and maintenance grants, and timber income

Woodland creation and maintenance grants

In the UK farmers can get grants for undertaking environmental practices and management. The Countryside Stewardship (CS) scheme provides incentives for land managers to look after their environment. Eligible practices include woodland creation as long as the woodland defined has a minimum area of 0.5 ha with a minimum width of 20 m. CS is a targeted competitive scheme which is scored. Ammonia is recognised under this scheme. Applications are scored higher when woodland creating is to support ammonia mitigation of protected sites.

The grants available for woodland creation include capital costs of establishing a woodland (with a maximum of £6,800/ha for planting and protection), plus an annual maintenance payment (£200/ha/year) for 10 years (*Countryside Stewardship: woodland-only Higher Tier application*, 2015).

Timber income

Only yields from thinnings are looked at in this study. For farm woodland shelterbelts it is here considered that the management of the trees were primarily managed for ammonia recapture and not commercially. Furthermore, the main canopy of the woodland system is seen as the only area that is thinned while the backstop is left unthinned. However Forestry Commission yield table gives estimates of yield from thinnings for many tree species. Thinning generally happen in 5 yearly cycles from year 25 of the woodland. As a consequence there are 4 harvested periods for thinnings in this study (year 25, 30, 35 & year 40) which have a total volume of 53 m^3 of wood. The price of timber for thinnings/short roundwood timber vary greatly from £30-£50 per tonne, while woodfuel can command around £75/m^3 of seasoned wood and this value is used in this study.

Table 6.2. Woodland grants and timber income

Income	Option 1	Option 2
CS Planting and Protection	£6,800	£6,800
CS Maintenance £200/yr for 10 years	£2,000	£2,000
Thinnings (£75 m^3 of wood)	£994	£2,120

6.2.5 Reducing damage costs of ammonia to benefit society

Estimates of the costs to society of the likely impacts of changes in emissions are provided here from two sources – the UK's Department for Environment, Food & Rural Affairs provides damage costs for a number of pollutants including ammonia. Based on 2014 prices the damage cost for ammonia is £2.18/kg NH_3 (Defra, 2011)). However, this damage cost only includes an estimate of human health impacts of secondary particulate matter. For this reason a fuller comparison was undertaken using outputs from the European Nitrogen Assessment (ENA) report (Brink *et al.*, 2011). This report examined the costs and benefits of nitrogen in the environment including (i) loss of life years and human health, (ii) loss of biodiversity and ecosystem services and (iii) climate change. Damage costs for emissions of ammonia to air were based on data for (i) and (ii) only (excluding costs for N_r to water), for the year 2010 – climate change damage costs concerns the emissions of N_2O only. The estimated ranges for damage costs of emissions of ammonia to air ranged from 4.86 – 36.43 (17) euro/kg NH_3 or 3.65 – 27.3 (12.75) £/kg NH_3 with single values inferred from case studies used in the ENA assessment (prices based on year 2014).

Damage costs for ammonia emissions are expressed as unit damage cost per kg. For the purposes of this study we estimated the amount of NH_3 captured by the woodland systems represent a net reduction in on-farm emissions. These were then compared with the total emissions of each option over 40 years. It was judged that there are no costs to society from planting trees.

Table 6.3. Yearly ammonia emissions (kg NH_3/ha) and capture efficiency for both options

	Option 1 (housing)	Option 2 (silvopastoral)
Yearly emissions (kg NH_3/ha of woodland planted)	8700	283
Tree capture efficiency (%)	20	40

6.2.6 Carbon sequestration benefits to society

Carbon sequestration provides both social and environmental benefits by contributing to reducing global warming. Carbon accumulated in the two woodland options were estimated using the CFLOW model. CFLOW is a carbon-accounting model used to calculate the carbon fluxes associated with afforestation and harvested wood products (LULUCF, 2009). CFLOW calculates the flow of carbon through the forest ecosystem from the time of planting using afforestation rate and yield class as inputs. Carbon is partitioned between the trees, litter and soil.

To represent the two planting option in the CLFOW model beech was used for the main canopy and Sitka spruce for the back-stop canopy. The following parameters applied in the model are shown in Table 6.4.

Table 6.4. Tree species with yield classes and management options used in the CFLOW model (Central range shown)

Tree species	Yield Class	Spacing (m)	Management
Beech	6	1.2	Intermediate thinning
Sitka spruce	12	2	No-thinning

The area for each option for beech and Sitka spruce were applied to the final calculations over 100 and 40 years. The carbon savings per hectare per year over 100 year time horizon for each planting option were 275 tC ha^{-1} (1001 tCO_2e ha^{-1}) for Option 1 and 261 tC ha^{-1} (959 tCO_2e ha^{-1}) for Option 2. Over 40 years the cumulative totals were 146 tC ha^{-1} (536 tCO_2e ha^{-1}) for Option 1 and 135 tC ha^{-1} (495 tCO_2e ha^{-1}) for Option 2.

Valuing the change in carbon emissions by tree planting can be quantified in monetary terms by using the Traded Price of Carbon (TPC) or Non-Traded Price of Carbon (NTPC). For projects involving land use change or forestry the NTPC is used. The NTPC values for 2008-2100 are provided in data Table 3 of the guidance (HM Treasury and DECC, 2014) and range from £61/tCO2e in 2014 to £245/tCO2e 40 years later (year 2053) based on 2014 prices.

6.2.7 Sensitivity Analysis

Sensitivity analysis was carried out to give upper, middle and lower values (Low, Central, High). This was applied to a variety of parameters set out in Table 6.5.

Table 6.5. Sensitivity analysis parameters

	Option 1 - tree belts downwind of animal housing and/or storage facilities			Option 2 - livestock managed under the trees		
	Low	Central	High	Low	Central	High
Agricultural Opportunity Costs (£)	776	655	534	776	655	534
Ammonia recapture factor (%)	10	20	30	30	45	60
Social value of ammonia (£/kg)	3.65	12.75	27.3	3.65	12.75	27.3

	Option 1 - tree belts downwind of animal housing and/or storage facilities			**Option 2 - livestock managed under the trees**		
	Low	Central	High	Low	Central	High
Non traded price of carbon based on 2014 year (£/tCO2e)	30	61	91	30	61	91
Carbon sequestration and yield class (Beech|Sitka spruce)	Y4|Y8	Y6|Y12	Y8|Y20	Y4|Y8	Y6|Y12	Y8|Y20

The 'Low' estimates are based on the highest agricultural opportunity cost, but with the lowest recapture factor, emissions, damage costs, NTPC and carbon sequestration.

6.3 Results

6.3.1 Costs and Benefits

A summary of the costs and benefits for both woodland options are outlined in Table 6.6. There are net costs to the farmer for both Options 1&2 at the best estimate and in both the low and high estimates for the sensitivity analysis. The grants provide for start-up and maintenance help with around a third of the farmers costs. Most of the cost (~65%) for both options over the 40 year period is explained by the agricultural opportunity cost of the land used. In order to be beneficial to the farmer, the product commodities would therefore need to attract a significant price premium on a 'green' product.

Looking at the Net Present Values and taking into account the social benefits from damage cost both options are highly beneficial. The main benefits come from the ammonia abatement damage costs with Option 1 providing the highest benefits due having the highest emissions (15k birds and 8780 kg NH_3/year/ha of woodland planted). Option 2 is limited by the maximum stocking density for open-air runs for poultry (2,500 birds/ha resulting in 283 kg NH_3/year/ha of woodland planted).

Table 6.6. Present day costs and income of two woodland planting schemes over 40 years. Both private individual costs and social savings are calculated (£ per ha, at 2014 prices). Low values represent the most expensive option with the least abatement and lowest social damage costs. Negative numbers in red represent a cost.

	Option 1 - tree belts downwind of animal housing and/or storage facilities			Option 2 - livestock managed under the trees		
Farmer's Individual Costs & Income (40 years)	**Low**	**Central**	**High**	**Low**	**Central**	**High**
Woodland Creation & Maintenance Costs	-32,351	-29,409	-26,467	-29,163	-26,476	-23,790
Woodland Grants and Timber Income	9,794	9,794	9,794	9,306	9,306	9,306
Total	-22,557	-19,615	-16,673	-19,857	-17,170	-14,484
Social Damage Savings (40 years)	**Low**	**Central**	**High**	**Low**	**Central**	**High**
Ammonia Abatement damage savings	60,044	419,488	1,347,298	8,779	30,666	87,549
Carbon sequestration savings	12,381	24,815	37,248	11,359	22,766	34,173
Total	72,425	444,303	1,384,546	20,138	53,432	121,722
Net Present Value (NPV)	49,868	424,688	1,367,873	281	36,262	107,238

6.3.2 Cost-effectiveness

Table 6.7 shows the cost effectiveness of each option in relation to ammonia abatement and carbon sequestration. The cost-effectiveness estimate compared with comparator shows Option 1 to be highly cost effective showing that ammonia recapture can be achieved at a cost of between £-0.29 to £0.43 per kg NH_3 saved. These compare very favourably with the comparators ranging from £1,815/kg NH_3 to £40,723/kg NH_3. Option 2 is also seen as cost effective with costs ranging from £-4.30 to £1.86 per kg NH_3 saved. The cost effectiveness comparator ranges from £265/kg NH_3 to £2,646/kg NH_3 showing that the effectiveness of implementing this option is high. Negative values represent a net benefit of implementing the project irrespective of the existence of any ammonia mitigation benefits. Positive values indicate a net cost to the famer per kg of NH_3 recaptured by the trees.

Similarly the outcome for climate change mitigation is also favourable to implementing both options. The cost-effectiveness estimate compared with comparator shows Option 1 to be highly cost effective, showing that carbon savings can be achieved at a cost of between £-2,282 to £-114/tCO_2e saved. These compare very favourably with the comparators based upon the non-traded price of carbon ranging from £28/tCO_2e to £85/tCO_2e. Option 2 is also seen as cost effective with costs ranging from £-137 to £36/tCO_2e saved. The cost effectiveness comparator ranges from £29/tCO_2e to £85/tCO_2e showing that the effectiveness of implementing this option is also favourable with only the Low estimate not being cost effective.

Table 6.7. Cost effectiveness of ammonia abatement and carbon sequestration policies over 40 years. The cost-effectiveness indicators are calculated as (the negative of) the NPV excluding the value (£) of the emissions saved in the sector of interest, divided by the carbon equivalent or ammonia captured by trees (as tCO_2e or kg NH_3). (DECC, 2014).

	Option 1 - tree belts downwind of animal housing and/or storage facilities			Option 2 - livestock managed under the trees		
Ammonia	**Low**	**Central**	**High**	**Low**	**Central**	**High**
NPV excluding NH_3 benefits[a] (£/ha)	-10,176	5,200	20,575	-8,498	5,596	19,689
NH_3 saving (kg NH_3)	23,490	46,132	70,470	4,579	3,434	4,579
Cost effectiveness indicator[b] (£/kg NH_3)	0.43	-0.11	-0.29	1.86	-1.63	-4.30
Comparator (£/kg **NH_3**)	1,815	12,679	40,723	265	927	2,646
Carbon						
NPV excluding Carbon benefits (£)	37,487	399,873	1,330,625	-11,078	13,496	73,065
Carbon savings (tCO_2e)	330	537	583	310	495	532
Cost effectiveness indicator[b] (£/t CO_2e)	-114	-745	-2,282	36	-27	-137
NTPC cost comparator[b] (£/t CO_2e)	28	58	85	29	57	85

[a] negative values in red are a cost

[b] negative values are not a cost but a benefit

6.3.3 Comparing options for ammonia mitigation

To compare the agroforestry options against other ammonia mitigation techniques, the cost of abating each kg of ammonia emission per year was calculated (Table 6.8). An average cost over 40 years were calculated based on the net cost to the farmer, excluding social benefits. Carrying out Option 1 is the cheaper of the two options estimated at £0.34 (0.19-0.79) / kg NH_3 abated. Option 2 is significantly more expensive at £41.12 (2.60-7.14) / kg NH_3 abated.

Table 6.8. Cost per kg of ammonia captured by the trees for each option excluding social benefits.

	Option 1 - tree belts downwind of animal housing and/or storage facilities			**Option 2 - livestock managed under the trees**		
	Low	**Central**	**High**	**Low**	**Central**	**High**
Ammonia recapture (%)	10	20	30	30	45	60
(b) Abated emissions over 40 Year cumulative (kg **NH_3** per hectare woodland)	23,490	46,980	70,470	2290	3434	4579
(a) Net cost of option	22,557	19,615	16,673	19,857	17,170	14,484
(c) Cost (£ per kg **NH_3**-N saved) (c=a/b)	0.79	0.34	0.19	7.14	4.12	2.60

Table 6.9 shows the comparison of how these agroforestry options to abate ammonia compare with other measures. The first part of the table focusses on non-caged poultry measures including housing design and manure management. Option 1 compared with the other measures is the cheapest of the options but provides only a 20% (10-30%) reduction in emissions. Option 1 can also be compared with measures for manure storage.

Option 2, while having high emission reduction potential, has high cost of implementation when compared with the other measures. Only the chemical and biological scrubbers are higher in price (but do offer up 90% reductions).

Table 6.9. Comparison of mitigation options for ammonia. For comparison Option1 & 2 have been compared with other Non-caged housing systems for laying hens. Other mitigation options are shown for further comparison. Ordered by lowest cost. (Table adapted from Bittman *et al.*, 2014)

Type of Measure	**Reduction (%)**	**Cost (€[a]/kg NH_3-N abated) ↓**
	Non-caged housing systems for laying hens	
Option 1 - tree belts downwind of animal housing.	10-30	0.2-0.8
Aviaries, perch design, non-ventilated manure belts	70-85	1-5
Litter with forced manure drying	40-60	1-5
Aviaries, ventilated manure belts	80-95	1-7
Option 2 - livestock managed under the trees	30-60	2.6-7.3
Litter, partly slatted, manure belts	75	3-5
Scrubbing of exhaust air	70-90	6-9
	Cattle, pig and slurry spreading and storage	
Injecting slurry (closed slot)	80	-0.5-1.2

Type of Measure	Reduction (%)	Cost (€[a]/kg NH_3-N abated) ↓
Injecting slurry (open slot)	70	-0.5-1.5
Band spreading with trailing shoe	30-60	-0.5-1.5
Band spreading slurry with trailing hose	30-35	-0.5-1.5
Option 1 - tree belts downwind of slurry lagoon	10-30	0.2-0.8
"Tight" lid, roof or tent structure	80	1.0-2.5

[a] GB sterling converted to Euros at a rate of 1.2674 based on the average 2014 rate.

6.4 Discussion and Conclusions

The results have shown that the main cost of planting trees for ammonia abatement is the switch from productive arable land to agroforestry. This opportunity cost accounted for up to 65% of the total cost for creating and maintaining a shelterbelt system over 40 years. Although hidden in real terms, this cost could be a major obstacle for farmers to implement, unless they can see other advantages of putting their land under trees. Testing against the comparators for each range (Low, Medium, High) for Option 1 & 2 shows that both options are cost-effective as the 'cost-effectiveness estimate' is always lower than the comparator. For both ammonia and climate change mitigation only the 'Low' range for Option 2 results is a positive estimate and above the comparator, making in not cost-effective.

Agroforestry for ammonia abatement options compared with other NH_3 mitigation options show that they are both feasible options for famers to implement. Although

Option 2 is one of the more expensive options. One of the drawbacks for planting trees for ammonia abatement is the time involved in reaching the optimum emission recapture capacity. Newly planted trees will have little recapture effect and it takes around 10 years for a beech tree to reach around 5-8 metres which is around the height of most pig and poultry sheds. This compares unfavourably with measures that involve an initial capital investment only e.g. installing ammonia scrubbers in a poultry shed. The investment in planting trees can only be gained at its full potential around 10 years after planting. Despite this, tree planting has the added benefit of storing carbon on top of recapturing ammonia which is an added benefit above other mitigation measures.

At this point it should also be noted that the income from each scenario are very different. For the farmer operating under Option 1 the number of birds are 6 times that of a famer operating under Option 2 for each hectare of woodland planted. Income streams will be much higher for Option 1 as their operation is based on a barn system of many birds compared to a silvo-pastoral system of much fewer birds (x6). Subsequently, the cost per unit of livestock for Option 1 (based on central estimates) are £1.30 per livestock unit compared to £6.87 for Option 2. Implementing tree planting for a famer wishing to operate under Option 2 make this approach less attractive. However, income streams from the two options have not been assessed in this research and added value gained from the producing 'woodland reared' products can attract a premium. Similarly animal welfare gains are very apparent for Option 2 while the amount of emission captured is much higher for Option 1 due to the higher numbers of livestock per hectare of trees planted.

It may be asked whether agroforestry for ammonia abatement can be described as a Category 1 measure according to UNECE classification. There is a large body of papers reporting beneficial effects of trees on air pollution mitigation through capture. Moreover, research into recapture of ammonia by trees has been quantified in the field and by using wind tunnel experiments (Famulari et al, unpublished). The main question lies with whether the method is practical considering the long run-in times (~10 years) for full efficiency to be achieved as described above. Against this disadvantage should be set the multiple win-wins for the farmer in planting trees around hot-spots of ammonia, together with the silvo-pastoral practice of grazing livestock under trees (ordered by farmer preference):

i. The potential for producing a price premium for poultry produce e.g. woodland chickens or woodland pork
ii. Visibility impacts can be improved as trees can break up the geometric shape of a building or hide them completely
iii. Improved animal welfare using silvo-pastoral systems. Sheltering of livestock by trees provides protection from predators, the sun in hot weather (reducing heat stress) and from rain and wind during inclement weather. Productivity can be improved and mortality reduced.
iv. Tree planting for ammonia mitigation supports the process of Industrial Emissions Directive (IED). High impact farms can implement tree-planting measures to reduce their on-farm emissions as a satisfying contribution to the requirements of an IED permit to release ammonia
v. Biodiversity - maintains the viability of agricultural woodlands and forests, preserves them for future generations, and could acts as a pool for genetic diversity in the landscape if local species are planted
vi. Reducing nitrogen deposition to nearby semi-natural habitats will lower critical load/level exceedance to the network of protected nature sites.
vii. Carbon sequestration from the trees contributes, all be it small, a role in achieving the UK's emission reduction targets for CO_2
viii. Planting trees augments the afforestation targets. For example, in the UK there are some 26 million laying birds (for eggs) in the UK. Converting all current barn systems to silvo-pastoral systems would create over 10,000 ha of reforested land. 27,500 ha of woodland could support the broiler population (110 million birds) in this way too.

In England it is very positive to see that agroforestry for ammonia abatement is being incorporated into the Countryside Stewardship application process with regard to 'buffering' protected sites like SACs. Conversely there appears to be little incentive for landowners to take account of the social value of ammonia capture at present nor CO_2 storage.. Grants may cover part of establishment costs and on-going management in some cases, but are not generally enough to compensate for the opportunity cost to the land owner unless price differentials of woodland eggs or pork could help. However, further incentives for landowners to take social values into account may be needed if agroforestry schemes should achieve their potential for ammonia and climate change mitigation.

Chapter 7. Discussion

This discussion brings together the findings from Chapters 2-6 and addresses five key questions set out at the end of Chapter 1 (Introduction). It finally discusses the question: What is the efficacy of planting trees for ammonia abatement?

7.1 What is the level of critical load exceedance across the UK Natura network and what are the dominant sources for policy makers to focus on to reduce exceedance?

Source attribution modelling in Chapter 2 showed that critical load exceedance for nutrient nitrogen is very substantial across the Natura network (SACs). Of the 500 potentially sensitive sites only 22 sites were below the critical load for the most sensitive habitats. 76% of all SAC sites exceeded their critical load for nutrient nitrogen, representing 74% of the entire network area. The extent of exceedance is also notable with many sites experiencing 50 kg N/ha/yr over the critical load. The results for acidity critical load are less severe but 51% of sites are still exceeded.

Chapter 3 emphasises that the level of critical load exceedance in the UK mirrors what is happening across the rest of Europe. Across the EU on average 73% of the Natura 2000 area is exceeded for nutrient nitrogen (CCE, 2014), with some countries having over 90% of their Natura area exceeded (e.g. Spain). The Habitats Directive — Europe's cornerstone for nature conservation strategy — sets out for each Member State not only to maintain habitats but also to restore habitats to a 'favourable conservation status'. However, the present level of critical load exceedance certainly puts many Annex I habitat at high risk from deterioration, unfavourable status, and loss of biodiversity. Furthermore, not only are the Habitats Directive's conservation objectives at risk of failing, but the EU 2020 Biodiversity Strategy targets will not be met.

By far the most predominant source contributing to critical load exceedance across the Natura network is livestock emissions. Nearly 90% of all SACs in the UK (n=564) have livestock as their dominant source, and on average 32% of nitrogen deposition comes from this source too. The likelihood for such a high influence is based on two facts - one is the nature of highly reactive ammonia to deposit over short distances (< 1 km) with large deposition velocities for forest and natural habitats, and two, the location of significant numbers of Natura sites in the near vicinity of agricultural activities. Short range dry deposition of ammonia on average accounts for nearly half (48%) of the total nitrogen load on Natura SAC sites. Long-range transport also plays an important role in nitrogen deposition although long range deposition of is dominated by oxidised forms of nitrogen. Long-range contributions of the two nitrogen forms still account for 30% of nitrogen deposition across the SAC network.

For sulphur large point sources are the most dominant sources across the network. Over 75% of all sites (n=482) have point sources as their dominant source.

The level of exceedance and the demonstration of source influence, together with the case studies carried out in Chapter 2 provide a useful insight into the complex problems policy makers and regulators have in reducing emissions. It is very clear that tackling agricultural ammonia is the key priority for now and the immediate future. But as we have seen sites vary in the matrix of inputs and other sources like shipping and point sources must still be taken care of. For example sites in upland areas away from local inputs of agricultural ammonia or road transport are impacted by long-range transport of pollutants as they are often in areas of high rainfall. A few sites are affected by natural sources (e.g. seabird colonies), so no form of reduction policy can be put forward in these cases.

The short/long range component of pollutant deposition clearly has a knock on effect when it comes to regulation and legislation. Long range deposition originates partly from sources outside the borders of Member States. In the UK around 20% of nitrogen deposition comes from continental Europe and Ireland (0 Figure 3.9). Combustion sources contributing to long range transport however have been regulated for some time now under EU Directives for example Industrial Emissions Directive (IED), Large

Combustion Plant Directive, and NECD. Tackling the short range sources within the rural landscape have until now not adequately been dealt with. Results presented here clearly indicate that addressing agricultural ammonia, and in particular from livestock production, is by far the main task ahead for policy makers in reducing on-site exceedances on the Natura 2000 network.

7.2 What are other EU countries experiences in regulating nitrogen pollution sources and what are the policy measures to combat exceedance on Natura 2000 sites?

Chapter 3 has shown that nitrogen deposition impacts are widespread across Natura 2000 sites, and that agricultural emissions of ammonia are seen as a main pollutant threat. However, Chapter 2 described that currently nature conservation policy is not adequately addressed in combination with air pollution and agricultural policies. While the Industrial Emissions Directive will continue to regulate pig and poultry farms, most farming activities are unregulated, with dairy and beef farming being a prominent omission. The Nitrates Directive (91/676/EEC), with the designation of Nitrate Vulnerable Zones (NVZ), also plays a role in improving water quality by protecting against nitrate pollution from agricultural sources. This is achieved by promoting better management of animal manures being stored and spread onto land, and the efficient use of nitrogen fertilisers.

Recent reviews of the Common Agriculture Policy (CAP) and the UNCECE CLTRAP have started to address the necessity for ammonia emission reduction in the agricultural landscape. Tighter emissions ceilings for ammonia under Gothenburg Protocol and the National Emission Ceiling Directive (NECD) and the associated mitigation measures could reduce emissions. Furthermore, the prospect of the funding of mitigation options for agricultural ammonia through the reform of CAP and the Rural Development Programme (RDP) is helpful for farmers. For example, funding is now available for Member States to roll out mitigation options under one of the six RDP Union priorities - *5(d) reducing greenhouse gas and ammonia emissions from agriculture*. A key element of

CAP is the application of cross-compliance. Current awareness (and enforcement) is lacking between Member States with respect to farmers and potential adverse effects on Natura sites. Further guidance is required to make cross-compliance links more transparent, while suitable low cost methods of abatement should be promoted.

Integration of the implementation of nature, air and agricultural polices is essential to reverse critical load exceedance and the unfavourable status of Natura sites. Some Member States have gone far in implementing such integration. In the Netherlands the development of the PAS (Programma Aanpak Stikstof – Integrated Approach to Nitrogen) approach (Section 2.3.1) shows how IT systems like AERIUS can be designed to support the process of managing the permitting process (Dutch Ministry of Economic Affairs, 2015). The Dutch system demonstrates an excellent approach towards auditing sources and keeping track of the multi-source inputs of nitrogen loads to protected sites, while at the same time setting out mandatory mitigation measures and restoration programmes ensures integrity of the site can be preserved. The aim is to allow managed development, but keep depositions and concentration on a downward trend.

Implementing Article 6.3 of the Habitats Directive, concerning the assessment of plans and projects, is also key in maintaining favourable status of the network. Many EU countries appear to do this well with regard to screening and modelling industrial emissions and their methodologies are comparable (Section 2.3). Deriving critical loads for habitats and applying threshold values is well understood and applied method as described in Section 2.3.

From the perspective of critical load exceedance and sustainability of the aims of the Natura 2000 network, agricultural ammonia provides a challenge. Much of the regulatory legislation under IED cover only a small percentage of the total agricultural emissions. The issuing of permits is only carried out for pig and poultry over a certain size. There is clearly a two-way approach to this - to assign further agricultural activities to the IED regulatory process (e.g. cattle) and require wider mitigation measures to reduce emissions under

NECD. In fact one could argue that both approaches should be taken to achieve zero critical load exceedance. A third approach is to raise awareness amongst practitioners in the assessment of nitrogen impacts, policy makers and also farmers. As described in Chapter 2, case studies show that while a site is experiencing high critical load exceedance, air pollution is rarely seen or mentioned as a potential problem for the site. The setting up of the Natura 2000 Biogeographical Process is a welcome platform for the sharing of knowledge and experiences. For example, under this process a workshop has been organised in The Netherlands in November 2015 to showcase PAS and a number of the innovative approaches to reduce the emissions of ammonia from intensive farming.

Future policies for incorporating spatial planning at both local and regional levels can be used to pinpoint locations of hot spots of ammonia where appropriate measures can be used. Integrating buffer zones and tree belts in the landscape can further be exploited to reduce deposition to Natura sites as described in Chapter 4. Planting trees around sources can provide suitable reductions in ammonia emissions of around 20%. This strategy of enhancing the deposition of ammonia within the 'farm gate' has the added benefit of reducing wet deposition and country export as more ammonia is deposited locally.

7.3 How much 'on-farm' ammonia emissions can be captured according to different scenarios?

Chapter 4 examined the capacity of trees to capture ammonia from agricultural activities. The study investigated two options: (a) planting trees downwind of housing and storage facilities and (b) managing livestock under the trees in a silvopastoral type system. Modelling showed that by manipulating the depth and density - leaf area index (LAI) - of the main canopy, including the addition of a backstop, up to 27% of the ammonia emitted could be recaptured by the trees for housing systems and up to 60% for under-storey livestock systems.

Optimising the tree scenarios for realistic on the ground conditions led to three planting designs:

i. A 20% recapture efficiency for animal housing systems using a 25 m main canopy of less dense stands (LAI 3 or 4) and a 25 m dense backstop (LAI 6) of 25 m planted with conifers.

ii. 20% recapture efficiency for manure storage lagoons systems using 30 m wide dense stands.

iii. 40% recapture efficiency for under-storey systems with a less-dense canopy structure (LAI=3) and a dense backstop of 25 m.

Deposition rates were affected the most by varying LAI and leaf area density (LAD), although the deeper the planted tree-belt the more ammonia could be recaptured. The incorporation of a dense backstop using conifers, for example, had the advantage to reduce ammonia escaping underneath the canopy and out the sides and back of the tree-belt. The dense backstop also had the added advantage of reducing the wind velocity in the main canopy allowing for a longer residency time and better recapture efficiency. Sensitivity analysis also showed that a backstop of conifers also plays a key role in winter months if the main canopy is solely planted with broadleaved trees. Around 13% to 15% of the ammonia could still be recaptured even if the main canopy was only capturing around 1 or 2% (Section 4.3.4).

7.4 How much can agroforestry systems reduce ammonia emissions on a national level?

Two strategies were modelled in Chapter 5. One strategy (Strategy A) used the planting of discreet blocks of trees around targeted ammonia hot-spots including livestock housing and manure storage, as well as directly placing livestock under trees. The other strategy (strategy B) used a process of swapping out arable land and grassland for trees, distributing them around hot-spots of ammonia. Both strategies reduce nitrogen deposition to semi-natural areas, both target areas of high ammonia emissions, and both strategies lead to the reduction in wet deposition and the export of nitrogen out of the UK as more is captured at source by the trees.

Strategy A gave very modest reduction in deposition based on the scenarios modelled with a 2% maximum reduction (3.5 kt) over the whole of the UK. The scenarios used were in some way not unrealistic, but applying 20% recapture factor for ammonia to 20% of cattle houses around the UK, for example, only results in the application of an overall 4% reduction in the model. Strategy B gave much higher reductions in deposition based on increasing afforested areas in the UK by 50%. National reductions in total deposition ranged from 3-6% (4.5 to 8.4 kt N). Importantly in both strategies the planted trees enhanced dry deposition, up to 19% for Strategy B, reducing deposition to semi-natural areas of up to 2 kg N/ha/yr. This reduction goes some way to reducing critical load exceedance as discussed in Chapter 2. Capturing ammonia at source also decreased wet deposition and, importantly for international legislation and protocols decreased export of nitrogen deposition to other European countries.

While the two strategies are similar in their aims to reduce ammonia by planting trees, in terms of deployment there approach could be seen to be different. Strategy A can be seen as a farming management switch to grazing livestock under trees and a sheltering of housing units with discreet units of tree-belts. By contrast, Strategy B is more of a farm-forestry management system as many more trees are planted. There is a bigger potential to provide timber products (e.g. wood fuel) and store carbon. Both strategies support afforestation targets for the UK, with Strategy B increasing the area of forest by 0.7 million hectares for a 25% increase in forest, to 1.4 million hectares for a 50% increase. Conversely Strategy A leads to much smaller areas of land being converted to trees. For example, if the 26 million laying hens in the UK were converted to silvopastoral systems this would create only around 10,000 ha of reforested land (stocking rate of 2500 birds/ha). Similarly, only 27,500 ha of new woodland would be needed to support the broiler population (110 million birds) in this way (stocking rate of 4000 birds/ha).

7.5 What are the comparative costs and additional benefits of agroforestry systems?

Chapter 6 showed that implementing agroforestry from both an ammonia mitigation and climate change perspective proved to be cost effective, especially when the human health and environmental costs of ammonia and carbon are taken into account. The cost effectiveness indicators of ammonia abatement and carbon storage in implementing both the trees downwind of housing and livestock under trees were well below the cost comparators even for the least favourable scenario. However without the social benefits the cost to the farmer over 40 years ranged from £22,557/ha to £16,673/ha for planting trees downwind of livestock housing, and £19,857/ha to £14,484/ha for managing livestock under planted trees.

The costs of implementing agroforestry for ammonia mitigation compare favourably with other more established ammonia mitigation options. Costs over 40 years (excluding social benefits) for planting tree-belts downwind of housing were €0.2-0.8 per kg NH_3-N abated. Planting trees for managing livestock under trees cost between €2.6 and €7.3 per kg NH_3-N abated. Compared with other options, agroforestry as a mitigation option stands up well, and planting trees for livestock housing ammonia capture is cheaper to implement than more conventional measures like force drying manure (€1-€5/kg NH_3-N), ventilated manure belts (€1-€5/kg NH_3-N) or scrubbing of exhaust air (€6-€9/kg NH_3-N). One of the drawbacks in planting trees is the long lead in time for the trees to reach the optimum recapture potential described in Chapter 4. So while the measure can be cheaper in the long run, the farmer has to wait for at least 10 years before ammonia recapture reaches optimums of 20% described in Chapter 6. This compares unfavourably with measures that involve an initial capital investment only. For example, installing ammonia scrubbers in a poultry shed can take a matter of days to install. But of course taking factors like comparing the lifetime of investment into technical equipment versus 40 years of tree growth make agroforestry options more robust. Technical measures like ammonia scrubbers or injection equipment have estimated lifetime periods of 20 and 10 years, respectively (Reis *et al.*, 2015).

Allocating agroforestry a UNECE Category 1 abatement measure is certainly reasonable within the terms of research, as there is a large body of papers reporting positive effects of trees on air pollution mitigation through capture. Moreover, research into recapture of ammonia by trees has been quantified in the field and by using wind tunnel experiments (Famulari *et al.*, 2014). The main question lies with whether the method is practical given the long run-in times (~10 years) for full efficiency to be achieved. However, when looked at as a package of benefits, agroforestry can be seen as highly beneficial to the farmer and society as a whole. These 'added value' benefits include: i) carbon sequestration, ii) visibility screening around housing units, iii) improved animal welfare for silvopastoral systems, iv) reducing critical load exceedance on protected sites v) price advantage of 'woodland chicken' products, vi) supporting IED requirements for emission reduction, vii) supporting national afforestation policies. Considering these wider advantages, the use of agro-forestry to mitigate NH_3 becomes an alternative option.

7.6 Land Use change and Food Security

As the world's population continues to rise up until at least the mid-12st century demand for food will rise. It is anticipated that in 2050 more than 9 billion people will have to share the world's limited resources. The UK is able to produce most of its food and supplement produce from abroad.

One scenario in Chapter 5 describes a maximum option of doubling current forest cover around ammonia hot-spots. This translates to around 1.4 million hectares of new trees replacing agricultural land. Agricultural land in the UK stands at about 17 million hectares and therefore under this scenario it accounts for an 8% switch to forested land. Another scenario – targeted planting with livestock under trees – would result in a much lower switch away from agricultural land, less than 0.5%. For UK agriculture, threats to food security are more likely to occur through sudden disruption to supply chains either at the local, regional or international level. There are a number of options that national governments can explore including the reduction in the food waste in the supply chain as well as at the consumer end, and changes in dietary preferences leaning towards less intensively produced animal products and a reduction on overall consumption of meat products.

7.7 What is the efficacy of planting trees for ammonia abatement?

Work carried out in the thesis has shown that agroforestry options for ammonia could be a suitable mitigation option for implementing at the farm level. It has overwhelming win-win outcomes.

Practical?

There is little doubt about the phenomenon of trees being able to capture pollutants, but the practicability and implementation in the landscape is a key factor to success. Getting the tree design structure and location of the tree belt systems in relation to the source is paramount to achieving optimal recapture. Three key elements are important for achieving optimal recapture – i) the addition of a backstop to prevent ammonia passing straight through the canopy; ii) planting trees on the prevailing downwind side of the source (although trees can be planted upwind too) and iii) the understorey at the front of the tree belt should be open so that the plume is directed into the denser part of the canopy. The tree belt should be more than just one row of trees but much deeper. In the study 50 m deep tree belts were modelled but the deeper the tree belt the more likely for increase recapture. Of course planting a 50 m deep block of woodland needs space, and suitable on the ground conditions may not exist for planting large woodland structures. Other design considerations include the fact that the height of a tree belt should be the same or higher than the source height (usually 6-10 m for animal housing). Obstacles like roads could also impeded where the optimal location for siting a tree belt could be.

Species Suitability?

Species suitability has not been the focus of this work. However, Theobald *et al.*, (2004) set out a list of criteria in choosing suitable species for designing tree belts. The most important criterion for species choice is the suitability for the environmental conditions at any particular site. Other criteria for species suitability were as follows:

- High nitrogen requirement and/or tolerance to added N.
- Rapid growth rate.
- High leaf area index (LAI).
- Fine leaf structure (increases the rate of ammonia recapture).
- Evergreen (i.e. conifer) component desirable in a mixed species canopy.
- Vigorous response to coppicing or pruning.
- Relatively wide site tolerance (to accommodate changes in site conditions, both natural and man-induced).

The most important criterion for tree species selection is for the tree to maintain healthy growth and at the same time have a high nitrogen requirement and tolerance of nitrogen. Some mix of evergreens are also desirable for year round recapture and species that can withstand coppicing of pruning are preferred. Woodland tree species that are suitable included beech (shade tolerant), field maple (suitable for most soils and can be coppiced), birch (suitable on most soils, can be coppiced) and Scots pine (high LAI) and Sitka spruce (high LAI and fast growing).

Welfare?

Improved animal welfare using silvopastoral type tree planting systems can be achieved through the protection from predators, the sun in hot weather (reducing heat stress) and from rain and wind during inclement weather. As a results of this productivity can be improved and mortality reduced.

Cost Effective?

Chapter 6 and Section 7.5 have shown that planting trees for ammonia abatement is cost effective if social benefits are taken into account. The larger the source strength the more the amount of ammonia can be abated by trees. The main cost to the farmer is the opportunity cost of changing land use from productive arable to agroforestry. Analyses showed that opportunity cost accounted for up to 65% of the total cost for creating and maintaining a shelterbelt system over 40 years. Losing arable land could be a major obstacle for the farmers to overcome in implementing agroforestry for ammonia abatement.

Supporting Legislation?

As a potential UNECE Category 1 measure planting trees to abate ammonia emissions could be added as an additional measure to the UNECE Gothenburg Protocol list of ammonia measures. Furthermore, providing it as an option under RDP and CAP could also increase options for land managers and policy makers. There are nearly 6,000 pig and poultry installation registered on the European Pollutant Release and Transfer Register (E-PRTR) under IED. Tree planting for ammonia mitigation supports the process of IED. High impact farms can implement tree-planting measures to reduce their on-farm emissions and help satisfy the requirements of an IED permit.

Mitigation for ammonia by tree planting also provides a benefit to protected areas in and around emission hot-spots. Reducing the impacts of eutrophication from ammonia emissions supports the goals of the Habitats Directive where critical load exceedance across the EU is high (70%). The goal of achieving favourable conservation status across the network is a target that appears to be difficult to achieve as ammonia emission have only come down slowly. As we have seen in Chapter 3 livestock emissions account for a third of nitrogen deposition across the Natura 2000 network in the UK, and three quarters of SACs exceed their minimum critical load. Strategically placed tree belts around farm sources can recapture NH_3, increase dispersion, and reduce the N deposition to nearby semi-natural areas, thereby reducing critical load exceedance. The largest benefits will be experienced by protected sites closest to an emission hot-spot, and by increasing the depth of a tree belt the recapture efficiency increases.

Externalities?

Externalities in this case refer to potential negative (or positive) outcomes of planting trees or unforeseen consequences of carrying out the measure. They are presented here for further discussion and possible further research needs.

Studies have shown that increasing the nitrogen load to forest soils can lead to an increase in N_2O and NO emissions (Skiba *et al.*, 2004). Current calculations used by IPCC apply an emission factor of 1% of deposited nitrogen leading to emissions of N_2O and NO (IPCC 1997). Therefore only a small fraction of deposited nitrogen is released back to the atmosphere. However, Denier van der Gon and Bleeker (2005) suggested that different land use experience different emission factors and these could differ by a factor of 2. Deciduous forest could be as high as 6%. It might be interesting to know exactly how emissions factors differ from changing land-use e.g. arable to forest. Fast growing trees may maximise nitrogen (and carbon) and N_2O emissions may be minimal.

Potential for canopy saturation has not been analysed or modelled. As ammonias concentrations rise cuticular resistances could increase and reduce the rate of deposition, and ammonia recapture could tend to reduce. Also adding nitrogen into the system can cause nutrient imbalances where increased growth (primary production) can exhaust the availability of base cations that in turn leads to reduced growth (Butterbach-Bahl and Gundersen, 2011). These lend to the need for specific management of woodland for NH_3 recapture (e.g. addition of base cations supplements).

Effects on recapture efficiency have not been researched with reference to a changing climate. In general increased temperature results in an increase in ammonia emissions as volatilisation is very sensitive to temperature (Sutton *et al.*, 2013). Similarly increased CO_2 concentrations in the atmosphere lead to a net increase in primary production. Above-ground accumulation of carbon in forests is in the range of 15–40 kg C/kg N (de Vries *et* *al.*, 2009). Trees will benefit from both the increased nitrogen and CO_2 input unless systems become saturated under high nitrogen deposition doses.

Application?

The application of agroforestry systems as a measure to mitigate ammonia across Europe is limited. Agri-environment schemes under CAP offer the potential for developing agroforestry as a suitable option. For example, under the Countryside Stewardship scheme in England and Wales, land managers can now receive grants for woodland creation where ammonia reductions have been targeted in particular around clusters of protected sites e.g. SACs.

7.8 Additional Research

Future research priorities are split into three parts i) further field experimentation, ii) applied software tools, and iii) nature based solutions and ecosystem services approach.

7.8.1 Experimentation

Further areas of work for the future should look at better quantifying the full nitrogen budget including measuring the change in rate of N_2O emissions from changing land use from arable to forest after trees have been planted, and the ecosystem nitrogen pathways. Nitrogen flows can in farm systems where agroforestry is used for ammonia abatement could be exploited to gain an understanding of the net effect on both the reactive and greenhouse gas nitrogen budgets. Animal housing internal concentrations could be better linked to net external emissions which could help reduce the uncertainties in linking measurements to the theoretical model studies. Species selection is a further area of research.

7.8.2 Software tools

For the application of tree planting for ammonia to be realised in practice there is a requirement for simple planting designs and guidance for farmers to follow. This could include the creation of template design structures giving information of numbers of trees, spacing, conifer/broadleaf mix in the main canopy, and backstop construction. Set widths and depths of designs can be set to a predetermined recapture factor. For example, guidance might say: “you can achieve a 20% ammonia emission reduction by planting 0.5 ha tree shelter belt for housing with a 25m main canopy and 25 m backstop…” which could then describe the planting schedule, fencing requirements etc.

An alternative approach would be to have a dynamic design system where famers could build their own tree shelter belt. The MODDAS-THETIS model used to model different tree scenarios in Chapter 4 could be transformed into an online version. This would enable the users to create their own designs while the system would provide detail on percentage recapture, estimate the number of trees required (based on a chosen spacing) and fencing amounts in metres.

7.8.3 Nature-based solutions

Nature-based solutions (NBS) is an emerging concept on the EU policy landscape (European Commission, 2015). The concept centres on the idea that the use of nature can be harnessed to tackle societal challenges such as environmental impact, food security, climate change and sustainable management. The NBS concept builds on and supports other closely related approaches such as the ecosystem services approach, and ecosystem-based adaptation/mitigation. Planting trees for the abatement of ammonia emissions fits nicely into this new area of research and links should be made between this research and NBS research.

7.9 Conclusions

This study aimed to i) demonstrate the importance of agricultural ammonia as a threat to protected sites (e.g. Natura 2000) and ii) examine the efficacy of agroforestry as an option for ammonia abatement. The study concluded:

1. Agricultural livestock production is the dominant nitrogen source across the Natura 2000 network in the UK. Nearly 90% of all sites had livestock as their

dominant source with livestock contributing 32% of the total nitrogen deposition across the while network.

2. 76% of all SAC sites exceeded their critical load for nutrient nitrogen, representing 74% of the entire network area. The extent of exceedance is also notable with many sites experiencing 50 kg N/ha/yr over the critical load. The results for acidity critical load are less severe but 51% of sites are still exceeded.
3. Current nature conservation policy is not closely linked with air and agricultural policies. Greater integration of these polices is essential to reverse critical load exceedance and the unfavourable status of Natura sites. The PAS approach in the Netherlands demonstrates an excellent approach towards assessing, permitting and thereby indirectly managing critical load exceedance.
4. By manipulating the depth and LAI of the main canopy, including the addition of a backstop, up to 27% of the ammonia emitted could be recaptured by the trees for housing systems and up to 60% for under-storey livestock silvopastoral systems. Practical recapture potential was set at 20% and 40% for housing and silvopastoral systems respectively.
5. The addition of a dense backstop of trees at the back and sides of the tree shelter belt can prevent ammonia passing through the canopy, with an added advantage of reducing the wind velocity in the main canopy allowing for a longer residency time and better recapture efficiency. Sensitivity analysis also showed that a backstop of conifers also plays a key role in winter months if the main canopy is solely planted with broadleaved trees.
6. By scaling up the national picture showed that by planting trees in hot areas of ammonia can lead to reduced deposition on nearby sensitive habitats. Furthermore, wet deposition and export outside the UK were reduced.
7. Agroforestry for ammonia abatement was shown to be cost effective for both planting downwind of housing and in silvopastoral systems when costs to society were taken into account. Planting trees were also cost effective from a climate change perspective.

8. Comparing the cost per kg of NH_3 abated showed that planting trees are a comparable method of mitigation with other more established techniques. The costs for planting downwind of housing was €0.2-0.8/kg NH_3 abated while the cost of the silvopastoral system was €2.6-7.3/kg NH_3
9. Agroforestry for ammonia abatement offers multi win wins for the famer and society as whole including i) carbon sequestration, ii) visibility screening around housing units, iii) improved animal welfare for silvopastoral systems, iv) reducing critical load exceedance on protected sites v) price advantage of 'woodland chicken' products, vi) supporting IED requirements for emission reduction, vii) supporting national afforestation policies.

References

Aber, J. D., Nadelhoffer, K. J., Steudler, P., & Melillo, J. M.. (1989). Nitrogen Saturation in Northern Forest Ecosystems. Bioscience, 39(6), 378–386. http://doi.org/10.2307/1311067

Achermann, B., and Bobbink, R. (eds.) (2003) Empirical critical loads for nitrogen. Proceedings of an Expert Workshop, Berne 11-13 November 2002. Environmental Documentation No. 164. Swiss Agency for the Environment, Forests and Landscape SAEFL.

Adrizal, A. ; Patterson, P.H. ; Hulet, R.M. ; Bates, R.M. ; Myers, C.A.B. ; Martin, G.P. ; Shockey, R.L. ; Van Der Grinten, M. ; Anderson, D.A. ; Thompson, J.R. (2008). Vegetative buffers for fan emissions from poultry farms: 2. ammonia, dust and foliar nitrogen. Journal of Environmental Science and Health, Part B, 2008, Vol.43(1), p.96-103

Air Pollution Information System (APIS), 2015. CBED - Concentration Based Estimated Deposition. http://www.apis.ac.uk/popup/cbed

Alard D. (2013) Theme 2 – Knowledge sharing of the practical solutions to reduce nitrogen deposition impacts: France. In Whitfield, C. & McIntosh, N. 2014. Nitrogen Deposition and the Nature Directives Impacts and responses: Our shared Experiences. Report of the Workshop held 2–4 December 2013, JNCC Peterborough. JNCC Report No. 521

Armstrong, H. M., Poulsom, L., Connolly, T., and Peace, A. (2003) A survey of cattle-grazed woodlands in Britain. Report to the Forestry Commission. **65** pp.

Asman, W. A. H. (2008). Entrapment of ammonia, odour compounds, pesticide sprays and pathogens by shelterbelts. DJF Plant Science Report No. 135, Faculty of Agricultural Sciences, University of Aarhus, Denmark, **167** pp.

Asman, W. A. H., Drukker, B., and Janssen, A. J. (1988) Modelled historical concentrations and depositions of ammonia and ammonium in Europe. *Atmospheric Environment* **22**, 725–735.

Asman, W. A. H., Sutton, M. A., and Schjørring, J. K. (1998) Ammonia: emission, atmospheric transport and deposition. *New Phytologist* **139**, 27–48.

Barret, K. and Seland, O. (1995) European transboundary acidifying air pollution – Ten years calculated field and budgets to the end of the first sulphur protocol. EMEP, 1/95, Norwegian Meteor. Inst. Oslo, Norway.

Bealey W. J., Famulari, D., Braban, C., and Sutton, M. A. (2011) Agroforestry Systems for Ammonia Abatement (SAMBA), Presentation at Farm Woodland Forum Annual General Meeting 1 July 2011 at Wakelyns Agroforestry, Metfield, Suffolk, England. http://www.agroforestry.ac.uk/sites/www.agroforestry.ac.uk/files/downloads/2011_meeting/bealey_pp.pdf

Bealey, W. J., Loubet, B., Braban, C. F., Famulari, D., Theobald, M. R., Reis, S., Reay, D. S., and Sutton, M. A. (2014) Modelling agro-forestry scenarios for ammonia abatement in the landscape. *Environmental Research Letters* **9** (12), doi:10.1088/1748-9326/9/12/125001

Bealey, W.J., Bleeker, A., Spranger, T., Bernotat, D., and Buchwald, E. (2011) Approaches to assessing the impact of new plans and projects on Natura 2000 sites (Theme 1). Background document. In Hicks, W.K., Whitfield, C.P., Bealey, W.J. and Sutton M.A. (eds.) (2011) Nitrogen Deposition and Natura 2000: Science & practice in determining environmental impacts. COST729/ Nine/ESF/CCW/JNCC/SEI Workshop Proceedings, published by COST. Available at: http:// cost729.ceh.ac.uk/n2kworkshop

Beckett, K. P., Freer-Smith, P. H., and Taylor, G. (1998) Urban woodlands: their role in reducing the effects of particulate pollution. *Environmental Pollution* **99**, 347–360.

Beckett, K. P., Freer-Smith, P. H., and Taylor, G. (2000) Particulate pollution capture by urban trees: effect of species and windspeed. *Global Change Biology* **6** (8), 995–1003.

Bittman, S., Dedina, M., Howard C. M., Oenema, O., and Sutton, M. A., (eds). (2014) Options for Ammonia Mitigation: Guidance from the UNECE Task Force on Reactive Nitrogen, Centre for Ecology and Hydrology, Edinburgh, UK.

Bjerregaard H. (2011) Impact assessment and regulation of N-emissions from livestock farms in Denmark 37. In Hicks, W.K., Whitfield, C.P., Bealey, W.J. and Sutton M.A. (eds.) (2011) Nitrogen Deposition and Natura 2000: Science & practice in determining environmental impacts. COST729/ Nine/ESF/CCW/JNCC/SEI Workshop Proceedings, published by COST. Available at: http:// cost729.ceh.ac.uk/n2kworkshop

Blackall, T. D., Wilson, L. J., Theobald, M. R., Milford, C., Nemitz, E., Bull, J., Bacon, P. J., Hamer, K. C., Wanless, S., and Sutton M. A. (2007) Ammonia emissions from seabird colonies. *Geophysical Research Letters*, **34**, L10801.

Bobbink, R., and Roelofs, J. G. M. (1995) Nitrogen critical loads for natural and semi-natural ecosystems: the empirical approach. *Water, Air and Soil Pollution* **85**, 2413-2418.

Bobbink, R., Braun, S., Nordin, A., Power, S., Schutz, K., Strengbom, J., Weijters, M., Tomassen, H. (2011) Review and revision of empirical critical loads and dose-response relationships. (Bobbink R, Hettelingh JP). Noordwijkerhout: Coordination Centre for Effects; 2011.

Bobbink, R., Hicks, K., Galloway, J. N., Spranger, T., Alkemade, R., Ashmore, M., Bustamante, M., Cinderby, S., Davidson, E., Dentener, F., Emmett, B., Erisman, J.-W., Fenn, M., Gilliam, F., Nordin, A., Pardo, L., and de Vries, W. (2010) Global Assessment of Nitrogen Deposition Effects on Terrestrial Plant Diversity: a synthesis. *Ecological Applications* **20**, 30-59.

Bobbink, R., Hornung, M., and Roelofs, J. G. H. (1998) The effects of air-borne pollutants on species diversity in natural and semi-natural European vegetation. *Journal of Ecology* **86**, 717-738.

Bouwman, A. F., Boumans, L. J. M., and Batjes, N. H. (2002) Estimation of global NH_3 volatilization loss from synthetic fertilizers and animal manure applied to arable lands and grasslands. *Global Biogeochemical Cycles* **16**(2), doi:10.1029/2000GB001389.

Brink, C., Grinsven, H. V., Jacobsen, B. H., Rabl, A., Gren, I., Holland, M., Klimont, Z., Hicks, K., Brouwer, R., Dickens, R., Willems, J., Termansen, M., Velthof, G., Alkemade, R., Oorschot, M. V. and Webb, J. (2011) Costs and benefits of nitrogen in the environment. In: Sutton, M., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, H. and Grizzetti, B., eds. (2011) The European Nitrogen Assessment. Cambridge: Cambridge University Press

Cape, J. N., Tang, Y. S., van Dijk, N., Love, L., Sutton, M. A., and Palmer S. C. F. (2004) Concentrations of ammonia and nitrogen dioxide at roadside verges, and their contribution to nitrogen deposition. *Environmental Pollution* **132,** 469-478.

Caporn, S. J. M., Carroll, J. A., Studlholme, C., and Lee, J. A. (2006) Recovery of Ombrotrophic Sphagnum Mosses in Relation to Air Pollution in the Southern Pennines. Report to Moors For The Future, Edale.

CLRTAP (2004) Manual on methodologies and criteria for modelling and mapping critical loads and levels and air pollution effects, risks and trends, ECE Convention on Long-range Transboundary Air Pollution.

Cohen, P., Potchter, O., and Schnell, I. (2014) The impact of an urban park on air pollution and noise levels in the Mediterranean city of Tel-Aviv, Israel. *Environmental Pollution* **195**, 73-83.

Commission Directive 2003/76/EC of 11 August 2003 amending Council Directive 70/220/EEC relating to measures to be taken against air pollution by emissions from motor vehicles

Council Directive (EU) 1999/30/EC of 22 April 1999 relating to limit values for sulphur dioxide, nitrogen dioxide and oxides of nitrogen, particulate matter and lead in ambient air.

Council Directive (EU) 2001/80/EC of 23 October 2001 on the limitation of emissions of certain pollutants into the air from large combustion plants. (OJ L 309, 27.11.2001, p.13).

Council Directive (EU) 2001/81/EC of 23 October 2001 on national emission ceilings for certain atmospheric pollutants. (OJ L 309, 27.11.2001, p. 22).

Council Directive (EU) 2010/75/EU of the European Parliament and of the Council of 24 November 2010 on industrial emissions (integrated pollution prevention and control).

Council Directive (EU) 85/337/EEC of 27 June 1985 on the assessment of the effects of certain public and private projects on the environment.

Council Directive (EU) 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.

Council Directive (EU) 96/62/EC of 27 September 1996 on ambient air quality assessment and management.

Dawkins, M. S., Cook, P. A., Whittingham, M. J., Mansell, K. A., and Harper, A. E. (2003) What makes free-range broiler chickens range? In situ measurement of habitat preference. *Animal Behaviour* **66**, 151–160.

De Vries, W. (1991) Methodologies for the assessment and mapping of critical loads and of the impact of abatement strategies on forest soils. Report 46, DLO Winand Staring Centre, Wageningen, The Netherlands, 109 pp.

de Vries, W., Solberg, S., Dobbertin, M., Sterba, H., Laubhahn, D., Reinds, G.J., Nabuurs, G.-J., Gundersen, P., and Sutton, M. A. (2008) Ecologically implausible carbon response? *Nature* **451**(7180), E1-E3.

de Vries, W., Solberg, S., Dobbertin, M., Sterba, H., Laubhann, D., Van Oijen, M., Evans, C., Gundersen, P., Kros, J., Wamelink, G. W. W., Reinds, G. J., and Sutton, M. A. (2009) The impact of nitrogen deposition on carbon sequestration by European forests and heathlands. *Forest Ecology and Management* **258** (8), 1814-1823.

DECC (2014) Valuation of energy use and greenhouse gas (GHG) emissions. Supplementary guidance to the HM Treasury Green Book on Appraisal and Evaluation in Central Government. HM Treasury and Department for Energy and Climate Change, London, http://www.decc.gov.uk/en/content/cms/about/ec_social_res/iag_guidance/iag_guidance.aspx.

DEFRA (2000) The Air Quality Strategy for England, Scotland, Wales and Northern Ireland. Working Together for Clean Air. London: HMSO.

DEFRA (2003) Part IV of the Environment Act 1995: Local Air Quality Management. LAQM. TG(03). London. HMSO.

DEFRA (2006) Air Quality Expert Group: Report on Nitrogen Dioxide in the United Kingdom. London. HMSO.

DEFRA (2011) Air Quality Damage Cost Guidance. Department for Environment, Food and Rural Affairs, London, http://www.defra.gov.uk/environment/quality/air/air-quality/economic/damage/.

DEFRA (2012) RoTAP: Review of Transboundary Air Pollution: Acidification, Eutrophication, Ground Level Ozone and Heavy Metals in the UK. Centre for Ecology & Hydrology.

DEFRA (2014) Government Forestry and Woodlands Policy Statement Implementation Plan – 'One Year On'. http://www.forestry.gov.uk/pdf/Forestry_and_Woodlands_Policy_Statement.pdf/$FILE/Forestry_and_Woodlands_Policy_Statement.pdf.

Denier van der Gon, H. A., and Bleeker, A. (2005) Indirect N2O emission due to atmospheric N deposition for the Netherlands. *Atmos. Environ.* **39**, 5827–5838.

Dore, A. J., Kryza, M., Hall, J. Hallsworth, S., Keller, V., Vieno, M., and Sutton, M. A. (2012) The Influence of Model Grid Resolution on Estimation of National Scale Nitrogen Deposition and Exceedance of Critical Loads. *Biogesciences* **9**, 1597-1609.

Dore, A. J., Vieno, M., Fournier, N., Weston, K. J., and Sutton, M. A. (2006) Development of a new wind rose for the British Isles using radiosonde data and application to an atmospheric transport model. *Q.J.Roy.Met.Soc* **132**, 2769-2784.

Dore, A. J.; Vieno, M.; Tang, Y. S.; Dragosits, U.; Dosio, A.; Weston, K. J.; Sutton, M. A.. (2007) Modelling the atmospheric transport and deposition of sulphur and nitrogen over the United Kingdom and assessment of the influence of SO2 emissions from international shipping. *Atmospheric Environment*, 41 (11). 2355-2367. doi:10.1016/j.atmosenv.2006.11.013

Dore, C. J., Murrells, T. P., Passant, N. R., Hobson, M. M., Thistlethwaite, G., Wagner, A., Li ,Y., Bush, T., King, K. R., Norris, J., Coleman, P. J., Walker, C., Stewart, R. A., Tsagatakis, I., Conolly, C., Brophy, N. C. J., and Hann, M. R. (2008) UK Emissions of Air Pollutants 1970 to 2006. AEAT 2008.

Dore, A. J., Carslaw, D. C., Braban, C., Chemel, C., Conolly, C., Derwent, R. G., Griffiths, S. J., Hall, J., Hayman, G., Lawrence, S., Metcalfe, S. E., Redington, A., Simpson, D., Sutton, M. A., Sutton, P., Tang, Y. S., Vieno, M., Werner, M., and Whyatt, J. D.: Evaluation of the performance of different atmospheric chemical transport models and inter-comparison of nitrogen and sulphur deposition estimates for the UK, Atmos. Environ., in review, 1–24, 2015.

Dragosits, U., Sutton, M. A., Place, C. J., and Bayley, A. A. (1998) Modelling the Spatial Distribution of Agricultural Ammonia Emissions in the UK. *Environmental Pollution* **102** (S1), 195-203.

Dragosits, U., Theobald, M. R., Place, C. J., ApSimon, H. M., and Sutton, M. A. (2006) The potential for spatial planning at the landscape level to mitigate the effects of atmospheric ammonia deposition. *Environmental Science and Policy* **9**, 626-638.

Dragosits, U., Theobald, M. R., Place, C. J., Lord, E., Webb, J., Hill, J., ApSimon, H. M., and Sutton, M. A. (2002) Ammonia emission, deposition and impact assessment at the field scale: a case study of sub-grid spatial variability. *Environmental Pollution* **117**, 147-158.

Dupont, S. and Brunet, Y. (2006) Simulation of turbulent flow in an urban forested park damaged by a windstorm. *Boundary Layer Meteorology* **120**, 133-161.

Dutch Ministry of Economic Affairs, 2015. Integrated Approach to Nitrogen - Scope for economic development, more robust nature and reduced nitrogen.

EMEP (1997) EMEP, Transboundary Air Pollution in Europe. In: E. Berge, Editor, MSC-W Status Report 1997. Part 1: Emissions, dispersion and trends of acidifying and eutrophying agents. Co-operative programme for monitoring and evaluation of the long range transmission of air pollutants in EuropeMeteorological Synthesising Centre-West, The Norwegian Meteorological Institute, Oslo (1997).

EMEP Status Report 1/09 (2007) "Transboundary acidification, eutrophication and ground level ozone in Europe in 2007". Joint MSC-W & CCC & CEIP Report.

Environment Agency (2010) Environmental Permitting (England and Wales) Regulations 2010 Regulation 60(1). https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/296993/LIT_7807_d074d7.pdf

Erisman, J. W., Bleeker, A., Galloway, J., Sutton, M. A. (2007) Reduced nitrogen in ecology and the environment. *Environmental Pollution* **150** (1), 140-149.

Erisman, J. W., Sutton, M. A., Galloway, J. N., Klimont, Z., and Winiwarter, W. (2008) How a century of ammonia synthesis changed the world. *Nature Geoscience* **1**, 636 –639.

Erisman, J. W., van Grinsven, H., Grizzetti, B. *et. al.* (2011) The European nitrogen problem in a global perspective. In: The European Nitrogen Assessment, ed. Sutton, M. A., Howard, C. M., Erisman, J. W., *et al.* Cambridge University Press.

European Commission (2105) Towards an EU Research and Innovation policy agenda for Nature-Based Solutions & Re-Naturing Cities Final Report of the Horizon 2020 Expert Group on 'Nature-Based Solutions and Re-Naturing Cities'

European Commission (2003) Integrated pollution prevention and control. Reference document on Best Available Techniques for Intensive Rearing of Poultry and Pigs (BREF). http://eippcb.jrc.ec.europa.eu/reference/irpp.html Accessed 14/08/2015.

European Environment Agency, (2013). EMEP/EEA air pollutant emission inventory guidebook 2013

Falkengren-Grerup, U. (1986) Soil acidification and vegetation changes in deciduous forest in Southern Sweden. *Oecologia* **70**, 339-347.

Famulari, D., Twigg, M. M., Robertson, A., Grigorova, E., Braban, C. F., Theobald, M. R., Sutton, M. A., and Nemitz, E. (2014) Quantifying NH_3 depletion by a woodland: a wind tunnel experiment. In 'Agroforestry Systems for Ammonia Abatement', ed Bealey, W. J., Braban, C. F., Theobald, M. R., Famulari, D., Tang, Y. S., Wheat, A., Grigorova, E., Leeson, S. R., Twigg, M. M., Dragosits, U., Dore, A. J., Sutton, M. A., Nemitz, E., Loubet, B., Robertson, A., Quinn, A. D., Williams, A., Sandars, D. L., Valatin, G., Perks, M., and Watterson, D. Defra Report AC0201 [unpublished].

Farmer, A. M., Bates, J. W., and Bell J. N. B. (1991a) Ecophysiological effects of acid rain on bryophytes and lichens. In : Bryophytes and lichens in a changing environment. Bates, J.W. and Farmer, A.M. (Eds) Oxford Scientific.

Ferm, M., 1998, Atmospheric ammonia and ammonium transport in Europe and critical loads a review: Nutrient Cycling in Agroecosystems, v. 51, p. 5-17.

Flechard C.R., Nemitz E., Smith R.I., Fowler D., Vermuelen A.T., Bleeker A., Erisman J.W., Simpson D., Zhang, L., Tang Y.S., Sutton M.A. (2011) Dry deposition of reactive nitrogen to European ecosystems: a comparison of inferential models across the NitroEurope network. Atmospheric Chemistry and Physics, 11(6), 2703-2728.

Forestry Commission Scotland (2009). The Scottish Government's rationale for woodland expansion. http://scotland.forestry.gov.uk/images/corporate/pdf/ForestExpansion.pdf.

Foudhil, H., Brunet, Y., and Caltagirone, J. P. (2005) A k - ε model for atmospheric flow over heterogeneous landscapes. *Environ. Fluid Mech.* **5**, 247–265.

Fournier, N., Dore, A. J., Vieno, M., Weston, K. J., Dragosits, U., and Sutton, M. A. (2004) Modelling the deposition of atmospheric oxidised nitrogen and sulphur to the United Kingdom using a multi-layer long-range transport model. *Atmospheric Environment* **38** (5), 683-694.

Fowler, D., Flechard, C., Skiba, U., Coyle, M., and Cape, J. N. (1998) The Atmospheric Budget of Oxidized Nitrogen and Its Role in Ozone Formation and Deposition. *New Phytologist* **139**, 11-23.

Fowler, D., Pilegaard, K., Sutton, M. A., Ambus, P., Raivonen, M., Duyzer, J., Simpson, D., Fagerli, H., Fuzzi, S., Schjoerring, J. K., Granier, C., Neftel, A., Isaksen, I. S. A., Laj, P., Maione, M., Monks, P. S., Burkhardt, J., Daemmgen, U., Neirynck, J., Personne, E., Wichink-Kruit, R., Butterbach-Bahl, K., Flechard, C., Tuovinen, J. P., Coyle, M., Gerosa, G., Loubet, B., Altimir, N., Gruenhage, L., Ammann, C., Cieslik, S., Paoletti, E., Mikkelsen, T. N., Ro-Poulsen, H., Cellier, P., Cape, J. N., Horvath, L., Loreto, F., Niinemets, U., Palmer, P. I., Rinne, J., Misztal, P., Nemitz, E., Nilsson, D., Pryor, S., Gallagher, M. W., Vesala, T., Skiba, U., Brueggemann, N., Zechmeister-Boltenstern, S., Williams, J., O'Dowd, C., Facchini, M. C., de Leeuw, G., Flossman, A., Chaumerliac, N., and Erisman, J. W. (2009) Atmospheric composition change: Ecosystems-Atmosphere interactions. *Atmospheric Environment* **43** (33), 5193-5267.

Fowler, D., Skiba, U., Nemitz, E., Choubedar, F., Branford, D., Donovan, R., and Rowland, P. (2004) Measuring Aerosol and Heavy Metal Deposition on Urban Woodland and Grass Using Inventories of 210Pb and Metal Concentrations in Soil. *Water, Air and Soil Pollution*: Focus **4** (2-3), 483–499.

Fowler, D., Sutton, M. A., Smith, R. I., Pitcairn, C. E. R., Coyle, M., Campbell, G., and Stedman, J. (1998b) Regional mass budgets of oxidized and reduced nitrogen and their relative contribution to the nitrogen inputs of sensitive ecosystems. *Environmental Pollution* **102**, 337-342.

Gallagher, M. W., Nemitz, E., Dorsey, J. R., Fowler, D., Sutton, M. A., Flynn, M., and Duyzer, J. (2002) Measurements and parameterizations of small aerosol deposition velocities to grassland, arable crops, and forest: Influence of surface roughness length on deposition. *Journal of Geophysical Research* **107**, D12, 10, doi:10.1029/2001JD000817, issn:0148-0227.

Galloway, J. N. (1998) The global nitrogen cycle: Changes and consequences. *Environmental Pollution* **102** (S1), 15–24.

Galloway, J. N., W. H. Schlesinger, H. Levy II, A. Michaels, and J. L. Schnoor (1995), Nitrogen fixation: Anthropogenic enhancement-environmental response, Global Biogeochem. Cycles, 9(2), 235–252, doi:10.1029/95GB00158.

Galloway, J. N., Aber, J. D., Erisman, J. W., Seitzinger, S. P., Howarth, R. W., Cowling, E. B., and Cosby, B. J. (2003) The nitrogen cascade. *BioScience* **53** (4), 341-356.

Galloway, J. N., Townsend, A. R., Erisman, J. W., Bekunda, M., Cai, Z., Freney, J. R., Martinelli, L. A., Seitzinger, S. P., and Sutton, M. A. (2008) *Science* **320** (5878), 889-892. [DOI:10.1126/science.1136674]

Galloway, J. N., Dentener, F. J., Capone, D. G., Boyer, E. W., Howarth, R. W., Seitzinger, S. P., Asner, G. P., Cleveland, C. C., Green, P. A., Holland, E. A., Karl, D. M., Michaels, A. F., Porter, J. H., Townsend, A. R., and Vörösmarty, C. J. (2004) Nitrogen cycles: Past, present, and future. *Biogeochemistry* **70**,153-226.

Great Britain, Parliament (1847) Towns Improvement Clauses Act 1847 (c.34). Office of Public Sector Information. http://www.opsi.gov.uk/RevisedStatutes/.

Great Britain, Parliament (1956) Clean Air Act. London : HMSO.

Great Britain, Parliament (1968) Clean Air Act. London : HMSO.

Great Britain, Parliament (1995) Environment Act. London : HMSO.

Gunnarsson, U., and Rydin, H. (2000) Nitrogen fertilization reduces Sphagnum production in bog communities. *New Phytologist* **147**, 527-537.

Haines-Young, R., and Potschin, M. (2007) 'Environmental Limits, Thresholds and Sustainable Development'. *BES Bulletin* **38**(2), 22-24.

Hall, J., Dore, A., Heywood, E., Broughton, R., Stedman, J., Smith, R., and O'Hanlon, S. (2006) Assessment of the environmental impacts associated with the UK Air Quality Strategy. DEFRA, London.

Hall, J., Dore, A., Heywood, E., Broughton, R., Stedman, J., Smith, R., and O'Hanlon, S. (2006) Assessment of the environmental impacts associated with the UK Air Quality Strategy. DEFRA, London.

Hallsworth, S., Dore, A. J., Bealey, W. J., Dragosits, U., Vieno, M., Hellsten, S.. Tang, Y. S. and Sutton, M. A. (2010) The role of indicator choice in quantifying the threat of atmospheric ammonia to the 'Natura 2000' network. *Environmental Science and Policy* **13**, 671-687.

Hellsten, S., Dragosits, U., Place, C. J., Vieno, M., and Sutton, M. A. (2008) Modelling and assessing the spatial distribution of ammonia emissions in the UK. *Environmental Pollution* **154**, 370-379.

Hettelingh, J. P., Posch, M., and Slootweg, J.: CCE Status Report (2008) Coordination Centre for Effects (CCE), 231 pp., 2008.

Hettelingh, J. P., Posch, M., and Slootweg, J.: CCE Status Report (2008) Coordination Centre for Effects (CCE), 231 pp., 2008.

Hornung M., and Sutton M. A. (1995) Impacts of nitrogen deposition in terrestrial ecosystems. *Atmospheric Environment* **29** Issue 22, 3395-3396.

Krupa, S. V. (2003) Effects of atmospheric ammonia (NH_3) on terrestrial vegetation: a review. *Environmental Pollution* **124**, 179-221.

Kryza M., Werner M., Dore A.J., Błaś M., Sobik M. (2012) The role of annual circulation and precipitation on national scale deposition of atmospheric sulphur and nitrogen compounds, J.Env.Man., 109, 70-79.

Lee J.A. (1991) The effects of acid deposition on blanket peatland vegetation. In: Woodin S. J. and Farmer .M. (Eds): The effects of acid deposition on nature conservation in Great Britain. NCC Focus on Nature Conservation Report 26. Nature Conservancy Council, Peterborough, p12-16.

Loubet, B. (2000) Modélisation du dépôt sec d'ammoniac atmosphérique à proximité des sources. Thèse de doctorat Thesis, Université Paul Sabatier, Toulouse.

Loubet, B., Asman, W. A. H., Theobald, M. R., Hertel, O., Tang, Y. S., Robin, P., Hassouna, M., Dammgen, U., Genermont, S., Cellier, P., and Sutton, M. A. (2009) Ammonia deposition near hot spots: processes, models and monitoring methods. In: Sutton, M. A., Reis, S., Baker, S. M. H., (eds.) Atmospheric Ammonia: Detecting emission changes and environmental impacts. Results of an Expert Workshop under the Convention on Long-range Transboundary Air Pollution. Springer, 205-267.

Loubet, B., P. Cellier, C. Milford, and Sutton, M. A. (2006) A coupled dispersion and exchange model for short-range dry deposition of atmospheric ammonia. *Quarterly Journal of the Royal Meteorological Society* **132**, 1733-1763.

LULUCF (2009) Inventory and Projections of UK Emissions by Sources and Removals by Sinks due to Land Use, Land Use Change and Forestry, Annual Report, July 2009. Centre for Ecology & Hydrology, University of Aberdeen, Forest Research Alice Holt, National Soil Resources Institute, Agri-Food & Biosciences Institute, Queen's University Belfast, University of Edinburgh.

Massad, R. S., Nemitz, E., and Sutton, M. A. (2010) Review and parameterisation of bi-directional ammonia exchange between vegetation and the atmosphere. *Atmospheric Chemistry and Physics* **10**, 10359-10386.

Matejko, M. Dore, A.J.; Hall, J; Dore, C.J. ; Blas, M.; Kryza, M; Smith, R.; Fowler, D. (2009) The influence of long term trends in pollutant emissions on deposition of sulphur and nitrogen and exceedance of critical loads in the United Kingdom. Environmental Science and Policy, 12. 882-896

Matejko, M., Dore, A. J., Hall, J., Dore, C. J., Błaś, M., Kryza, M., Smith, R., and Fowler, D. (2009) The influence of long term trends in pollutant emissions on deposition of sulphur and nitrogen and exceedance of critical loads in the United Kingdom. *Environmental Science and Policy* **12**, 882 – 896.

McDonald, A. G., Bealey, W. J., Fowler, D., Dragosits, U., Skiba, U., Smith, R. I., Donovan, R. G., Brett, H. E., Hewitt, C. N., and Nemitz, E. (2007) Quantifying the effect of urban tree planting on concentrations and depositions of PM10 in two UK conurbations. *Atmospheric Environment* **41**, 8455-8467.

Misselbrook, D. R., Gilhespy, S. L., Cardenas, L. M.,Chambers, B. J., Williams, J. and Dragosits, U. (2013) Inventory of Ammonia Emissions from UK Agriculture 2013, Inventory Submission Report, January 2013, DEFRA contract SCF0102.

Misselbrook, T. H., Chadwick, D. R., Gilhespy, S. L., Chambers, B. J., Smith, K. A., Williams, J. and Dragosits, U. (2010) Inventory of Ammonia Emissions from UK Agriculture 2009, Inventory Submission Report, October 2010, DEFRA contract AC0112.

Misselbrook, T. H., Dore, A. J., Dragosits, U., Tang, Y. S., Sutton, M. A., Hall, J., Reis, S., Anthony, S. G., and Dore, C. (2009) Underpinning evidence for development of policies to abate ammonia emissions, Defra Project AQ0602.

Mister, A. (1970) Britain's Clean Air Acts. *The University of Toronto Law Journal* **20** No. 2, 268-273.

Nihlgård, B. (1985). The ammonium hypothesis — an additional explanation to the forest dieback in Europe. Ambio, 14 (1985), pp. 2–8

Nilsson J. and Grennfelt P. (Eds) (1988) Critical Loads for Sulphur and Nitrogen. Miljorapport 1988:15. Nordic Council of Ministers, Copenhagen.

Nowak, D. J. (2000) Impact of urban forest management on air pollution and greenhouse gases. In: Proceedings of the Society of American Foresters 1999 national convention; 1999 September 11–15; Portland, OR. SAF Publ. 00-1. Bethesda, MD: Society of American Foresters: pp. 143–148.

Nowak, D. J., Civerolo, K. L., Rao, S. T., Sistla, G., Luley, C. J., and Crane, D. E. (2000) A modeling study of the impact of urban trees on ozone. *Atmospheric Environment* **34**, 1601–1613.

Nowak, D. J., Hirabayashi, S., Bodine, A., and Greenfield, E. (2014) Tree and forest effects on air quality and human health in the United States. *Environmental Pollution* **193**, 119-129.

Oenema, O., Velthof, G., Klimont, Z., and Winiwarter, W. (2012) Emissions from agriculture and their control potentials. Service Contract on Monitoring and Assessment of Sectorial Implementation Actions (ENV.C.3/SER/2011/0009). ed. Amann, M. IIASA

Oxley, T., Dore, A.J., Kryza, M. & ApSimon, H (2013). Modelling future impacts of air pollution using the multi-scale UK Integrated Assessment Model (UKIAM). Environment International. 61, 17-37.

Patterson, P. H., Adrizal, A., Hulet, R. M., Bates, R. M., Despot, D. A., Wheeler, E. F., and Topper, P. A. (2008a) The Potential for Plants to Trap Emissions from Farms with Laying Hens. 1. Ammonia. *Journal of Applied Poultry Research* **17** 1, 54-63.

Patterson, P.H. ; Adrizal, A. ; Hulet, R.M. ; Bates, R.M. ; Myers, C.A.B. ; Martin, G.P. ; Shockey, R.L. ; Van Der Grinten, M. (2008b). Vegetative buffers for fan emissions from poultry farms: 1. temperature and foliar nitrogen. Journal of Environmental Science and Health, Part B, 2008, Vol.43(2), p.199-204

Pearce, I. S. K., and van der Wal, R. (2002) Effects of nitrogen deposition on growth and survival of montane Racomitrium lanuginosum heath. *Biological Conservation* **104**, 83-89.

Pitcairn, C. E. R., Leith, I. D., Sheppard, L. J., Sutton, M. A., Fowler, D., Munro, R. C., Tang, S., and Wilson, D. (1998) The relationship between nitrogen deposition, species composition and foliar nitrogen concentrations in woodland flora in the vicinity of livestock farms. *Environmental Pollution* **102**, 41-48.

Posch, M., Hettelingh, J. P., and Slootweg, J. (2003) Manual for Dynamic Modelling of Soil Response to Atmospheric Deposition. ICP M&M Coordination Center for Effects.

Posch, M., Hettelingh, J. P., de Smet, P. A. M., and Downing, R. J. (Eds.) (1997) Calculation and mapping of critical thresholds in Europe. Status Report 1997, Coordination Centre for Effects. National Institute of Public Health and the Environment, Bilthoven, Netherlands. RIVM Report No. 259101007.

Posthumus, A. C. (1988) Critical levels for effects of ammonia and ammonium. Proceedings of the Bad Harzburg Workshop, UBA, Berlin (1988) pp. 117-127.

Press, M. C., Woodin, S. J., and Lee, J. A. (1986) The potential importance of an increased atmospheric nitrogen supply to the growth of ombrotrophic Sphagnum species. *New Phytologist* **103**, 45-55.

Reis, S., Grennfelt, P., Klimont, Z., Amann, M., ApSimon, H., Hettelingh, J. P., Holland, M., LeGall, A. C., Maas, R., Posch, M., Spranger, T., Sutton, M. A., and Williams, M. (2012) From acid rain to climate change. *Science* **338**, 1153–1154.

Reis, S., Howard, C., and Sutton, M. A. (2015) Costs of ammonia abatement and the climate co-benefits. Dordrecht, Springer, v-vi. ISBN 978-94-017-9722-1

Rodhe, H., and Grandell, J. (1972) On the removal time of aerosol particles from the atmosphere by precipitation scavenging. *Tellus* **24**, 442-454.

Sheppard, L. J., Leith, I. D., Crossley, A.van Dijk, N., Fowler, D., Sutton, M. A., and Woods, C. (2008) Stress responses of Calluna vulgaris to reduced and oxidised N applied under "real world conditions". *Environmental Pollution* **154**, 404-413.

Singles, R., Sutton, M. A., and Weston, K. J. (1998) A multi-layer model to describe the atmospheric transport and deposition of ammonia in Great Britain. *Atmospheric Environment* **32**, 393-399.

Slootweg J., Posch, M., Hettelingh, J. P., Mathijssen L. (2014) CCE Status Report (2014) Coordination Centre for Effects (CCE), 160pp., 2014.

Smith F. B. (1975) Airborne transport of sulphur dioxide from the United Kingdom. *Atmospheric Environment* **9**, 643-659.

Smith, F. B. (1981) The significance of wet and dry synoptic regions on long-range transport of pollution and its deposition. *Atmospheric Environment* **15** 5, 863-873.

Smith, R. I., Fowler, D., Sutton, M. A., Flechard, C. and Coyle, M. (2000) Regional Estimation of Gas Deposition in the UK: model description, sensitivity analyses and output. *Atmospheric Environment* **34**, 3757-3777.

Stevens, C. J., Dise, N. B., Gowing, D. J. G., and Mountford, J. O. (2006) Loss of forb diversity in relations to nitrogen deposition in the UK: regional trends and potential controls. *Global Change Biology* **12**, 1823–1833.

Stevens, C. J., Thompson, K., Grime, J. P., Long, C. J., and Gowing, D. J. G. (2010) Contribution of acidification and eutrophication to declines in species richness of calcifuges grasslands along a gradient of atmospheric nitrogen deposition. *Functional Ecology* **24**, 478-484.

Sutton, M. A., Dragosits, U., Theobald M. R., McDonald, A. G., Nemitz, E., Blyth, J. F., Sneath, R., Williams A., Hall, J., Bealey, W. J., Smith, R. I., Fowler, D. (2004) The role of trees in landscape planning to reduce the impacts of atmospheric ammonia deposition. In: Landscape Ecology of Trees and Forests, ed. R. Smithers, IALE (UK)/ Woodland Trust, Grantham, UK, pp. 143–150.

Sutton, M. A., Erisman, J. W., Dentener, F., and Möller, D. (2008) Ammonia in the environment: From ancient times to the present. *Environmental Pollution* **156**, 583–604.

Sutton, M. A., Pitcairn, C. E. R., and Fowler, D. (1993) The exchange of ammonia between the atmosphere and plant communities. *Advances in Ecological Research* **24**, 301–393.

Sutton, M. A., Place, C. J., Eager, M., Fowler, D., and Smith, R. I. (1995) Assessment of the magnitude of ammonia emissions in the United Kingdom. *Atmospheric Environment* **29**, 1393-1411.

Sutton, M. A., Tang, Y. S., Dragosits, U., Fournier, N., Dore, T., Smith, R. I., Weston, K. J., and Fowler, D. (2001) A spatial analysis of atmospheric ammonia and ammonium in the UK. The Scientific World 1, 275e286.

Tallis, M., Taylor, G., Sinnett, D. and Freer-Smith, P. H. (2011) Estimating the removal of atmospheric particulate pollution by the urban tree canopy of London, under current and future environments. *Landscape and Urban Planning* **103** (2). 129-138.

The Royal Society. 2008 Ground-level ozone in the 21st century: future trends, impacts and policy implications. London, The Royal Society, 132pp. (Science Policy, 15/08).

Theobald, M. R., Milford, C., Hargreaves, K. J., Sheppard, L. J., Nemitz, E., Tang, Y. S., Dragosits, U., McDonald, A. G., Harvey, F. J., Leith, I. D., Sneath, R. W., Williams, A. G., Hoxey, R. P., Quinn, A. D., McCartney, L., Sandars, D. L., Phillips, V. R., Blyth, J., Cape, J. N., Fowler, D., Sutton, M. A. (2004) Impact of vegetation and/or other on-farm features on net ammonia emissions from livestock farms. AMBER: Ammonia Mitigation By Enhanced Recapture. (project code WA0719).

Tietema, A.; Emmett, B.A.; Gundersen, P.; Kjønaas, O.J.; Koopmans, C.J. The fate of 15N-labelled nitrogen deposition in coniferous forest ecosystems. For. Ecol. Manag. 1998, 101, 19–27.

Tomassen, H. B. M., Smolders, A. J. P., Limpens, J., Lamers, L. P. M. and Roelofs, J. G. M. (2004) Expansion of invasive species on ombrotrophic bogs: desiccation or high N deposition? *Journal of Applied Ecology* **41**, 139-150.

UBA (2004) Manual on Methodologies and Criteria for Modelling and Mapping Critical Loads and Levels and Air Pollution Effects, Risks and Trends. Umweltbundesamt Texte 52/04, Berlin.

UNECE (1979) United Nations Economic Commission for Europe Geneva Convention on Long-range Transboundary Air Pollution 13 November 1979. UN Doc. ECE/HLM.1/R.1.

UNECE (1999) The 1999 Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone, United Nations Economic Commission for Europe (UNECE), Geneva, Switzerland.

UNECE (1999) United Nations Economic Commission for Europe (UNECE), 1999. Control techniques for Preventing and Abating Emissions of Ammonia. EB.AIR/WG.5/1999/8/Rev.1. UNECE, Geneva, Switzerland, 37 pp.

UNECE (2004) Revised Manual on Methodologies and Criteria for Mapping Critical Levels/Loads and Geographical Areas Where They are Exceeded, Umweltbundesamt, Berlin, Germany (2004).

UNECE (2014) Guidance document on control techniques for preventing and abating emissions of ammonia. ECE/EB.AIR/120. United Nations Economic Commissions for Europe (UNECE), Geneva, Switzerland. http://www.unece.org/fileadmin/DAM/env/documents/2012/EB/ECE_EB.AIR_120_ENG.pdf Accessed 14/08/14

UNECE 1996. Manual on methodologies and criteria for mapping critical levels/loads and geographical areas where they are exceeded. Texte 71/96, Umweltbundesamt, Berlin, Germany.

UNECE, 2012. Amendment of the text and annexes II to IX to the Gothenburg Protocol and addition of new annexes X and XI. ECE/EB.AIR/111/Add.1 http://www.unece.org/fileadmin/DAM/env/documents/2013/air/ECE_EB.AIR_111_Add.1__ENG_DECISION_2.pdf

UNEP (2004) GeoYear Book. United Nations Environment Programme.

United States, Congress (1969). National Environmental Policy Act of 1969 (NEPA). 42 U.S.C. 4321-4347.

United States, Congress (1970) The Clean Air Act (CAA). 42 U.S.C. s/s 7401 et seq.

Van Breemen, N., van Dijk, H. F. G. (1988) Ecosystem effects of atmospheric deposition of nitrogen in The Netherlands. *Environmental Pollution* **54** 3-4, 249-274.

Van den Berg, L. J. L., Peters, C. J. H., Ashmore, M. R. and Roelofs, J. G. M. (2008) Reduced nitrogen has a greater effect than oxidised nitrogen on dry heathland vegetation. *Environmental Pollution* **154**, 359-369.

Van Grinsven, H. J. M., Holland, M., Jacobsen, B. H., Klimont, Z., Sutton, M. A., and Jaap Willems, W. (2013) Costs and benefits of nitrogen for Europe and implications for mitigation *Environmental Science and Technololgy* **47** (8), 3571–3579.

Van Vuuren, D. P., Bouwman, L. F., Smith, S. J., and Dentener, F. (2011) Global projections for anthropogenic reactive nitrogen emissions to the atmosphere: an assessment of scenarios in the scientific literature. *Current Opinion in Environmental Sustainability* **3**, 359–369.

Vestreng, V., Myhre, G., Fagerli, H., Reis, S., and Tarrasón, L. (2007) Twenty-five years of continuous sulfur dioxide emission reduction in Europe. *Atmospheric Chemistry and Physics* **7**, 3663–3681.

Viana, M., Kuhlbusch, T. A. J., Querol, X., Alastuey, A., Harrison, R. M., Hopke, P. K., Winiwarter, W., Vallius, M., Szidat, S., Prévôt, A. S. H., Hueglin, C., Bloemen, H., Wåhlin, P., Vecchi, R., Miranda, A. I., Kasper-Giebl, A., Maenhaut, W., and Hitzenberger, R. (2008) 'Source apportionment of particulate matter in Europe: a review of methods and results'. *Journal of Aerosol Science* **39**, 827–849.

Vieno, M., Dore, A. J., Bealey, W. J., Stevenson, D. S., Sutton, M. A. (2010) The importance of source configuration in quantifying footprints of regional atmospheric sulphur deposition. *Science of the Total Environment* **408**, 985–995.

Walmsley, J. (2002) Framework for measuring sustainable development in catchment systems, *Environmental Management* **29** No2.

Warren, M. F. (1982) Financial Management for Farmers. Hutchinson, London, 306pp.

Whitfield, C. & McIntosh, N. 2014. Nitrogen Deposition and the Nature Directives Impacts and responses: Our shared Experiences. Report of the Workshop held 2–4 December 2013, JNCC Peterborough. JNCC Report No. 521

Wiedermann, M. M., Gunnarsson, U., Ericson, L. and Nordin, A. (2009b) Ecophysiological adjustment of two Sphagnum species in response to anthropogenic nitrogen deposition. *New Phytologist* **181**, 208-217.

Zhang, W. L., Tian, Z. X., Zhang, N., Li, X. Q. (1996) Nitrate pollution of groundwater in northern China. *Agriculture, Ecosystems and Environment* **59** Issue 3, 223-231.