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Valuing improvements in biodiversity due to controls on atmospheric nitrogen pollution

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Abstract

Atmospheric nitrogen pollution has severe impacts on biodiversity, but approaches to value them are limited. This paper develops a spatially explicit methodology to value the benefits from improvements in biodiversity resulting from current policy initiatives to reduce nitrogen emissions. Using the UK as a case study, we quantify nitrogen impacts on plant diversity in four habitats: heathland, acid grassland, dunes and bogs, at fine spatial resolution. Focusing on non-use values for biodiversity we apply value-transfer based on household's willingness to pay to avoid changes in plant species richness, and calculate the benefit of projected emission declines of 37% for nitrogen dioxide (NO_2) and 6% for ammonia (NH_3) over the scenario period 2007 – 2020. The annualised benefit resulting from these pollutant declines is £32.7m (£4.4m to £109.7m, 95% Confidence Interval), with the greatest benefit accruing from heathland and acid grassland due to their large area. We also calculate damage costs per unit of NO_2 and NH_3 emitted, to quantify some of the environmental impacts of air pollution for comparison with damage costs for human health in policy appraisal. The benefit is £103 (£33 to £237) per tonne of NO_2 saved, and £414 (£139 to £1,022) per tonne of NH_3 saved.

Keywords

Nitrogen deposition; species richness; economic value; damage cost; ecosystem services; policy

Declarations of interest: None

1. Introduction

Air pollution is a global issue that has substantial adverse impacts on human health, but also on the environment (Galloway et al., 2008; Oenema et al., 2011). For example, plant diversity at sites receiving high atmospheric nitrogen deposition in Europe is typically 50% lower than sites receiving low levels of nitrogen (Maskell et al., 2010; Stevens et al., 2004). While decades of research have catalogued the impacts of nitrogen deposition on natural systems (e.g. Pardo et al., 2011; Phoenix et al., 2012), there is increasing interest in using an ecosystem services perspective to evaluate the wider impacts of nitrogen on flows of goods and services (Compton et al., 2011; Jones et al., 2014; Smart et al., 2011).

Nitrogen deposition has started to decline in Western Europe due to targeted policies on emissions, with emissions 25% lower than their peak in 1990 (Oenema et al., 2011). Applying an ecosystem services approach to evaluate the non-health impacts of this pollution decline has shown both negative and positive impacts (Jones et al., 2014). For example, there are some costs to society as a result of the decline in 'free' fertiliser from atmospheric deposition. These costs come in the form of lower productivity of agricultural grasslands, and reductions in tree growth and in carbon sequestration. However, there are also major benefits to society through reductions in emissions of the greenhouse gas N_2O, improvements in water quality, and there may be large benefits to biodiversity, although this is difficult to value.

For a pollutant like nitrogen, this leads to potential tensions in deriving a Total Economic Value of those impacts, because provisioning services generally increase with nitrogen, and are much easier to value than cultural services where nitrogen generally has an adverse impact. In many cases provisioning services can be linked to market values, providing the basis for a relatively straightforward economic assessment (e.g. agricultural crop productivity, livestock productivity, or timber productivity). By contrast cultural benefits, including non-use values for biodiversity conservation, are the domain of non-market valuation methods (Hanley and Barbier, 2009). Deriving a TEV which fails to account for impacts on biodiversity may lead to incomplete assessment of the net benefit arising from lower levels of nitrogen deposition. There is therefore a need to improve the robustness of valuation approaches focusing on biodiversity and the drivers which impact on it.

A key knowledge gap relates to economic valuation of changes to biodiversity. Biodiversity is important at all levels in ecosystem services, playing a role in supporting, intermediate and final services (Mace et al., 2012). Both the level and the stability of ecosystem services tend to improve with increasing biodiversity (Isbell et al., 2011), while nitrogen decreases plant diversity (Field et al. 2014). Nitrogen alters the core processes, functions and biodiversity which underpin a wide range of supporting and intermediate services. It also influences final services directly through effects on environmental attributes such as plant and animal diversity and landscape aesthetics which people care about (Clark et al., 2017; Rhodes et al., 2017). Stated preference methods are the main approach to value the effect of changes in biodiversity on cultural services and non-use values (Champ et al., 2003; Christie et al., 2006), but studies need to be robust enough to satisfy value transfer requirements (Ninan, 2014).

A number of other issues present problems for valuing biodiversity impacts. These centre on spatial context and the relationships between nitrogen and biodiversity. Robust assessment of impacts requires information on the spatial location of both pressures (nitrogen) and receptors (biodiversity). Previous approaches have only been applied at national level (Smart et al., 2011). However, omitting spatial context may lead to considerable over- or under-estimation of impact depending on whether the changes in air pollution occur in the same location as the components of the ecosystem experiencing damage. Addressing this spatial disconnect is most important where the pattern of an air pollutant such as ammonia is heterogeneous at relatively fine scales (Loubet et al., 2009), and where the receptor plant communities have an uneven spatial distribution.

This approach requires sufficient understanding of the dose-response function between nitrogen and biodiversity. This can be a challenge because the evidence for nitrogen impacts on organisms covers a relatively small number of species (Dise et al., 2011), and relatively few of those studies provide the dose response functions required to model impacts across a range of nitrogen deposition. The most promising are studies that have evaluated statistical relationships between nitrogen and diversity but which also account for the effects of confounding factors like climate and other pollutants (Field et al., 2014; van den Berg et al., 2016).

Policy makers are increasingly required to utilise economic tools to evaluate the positive and negative impacts of policy measures (HM Treasury, 2003) in order to justify and to better target those policies. Therefore, there is a need to develop more sophisticated approaches to quantifying air pollution impacts on ecosystem services, which incorporate spatial context, and which value those impacts in ways that can be incorporated into policy appraisal (Dickens et al., 2013).

In this paper, we develop and apply new approaches to address these issues, using the UK as a case study. We i) outline a spatially-explicit methodology to quantify the impacts of N on biodiversity, ii) present a value-transfer approach to translate those impacts into economic values and iii) combine these techniques to answer the policy question: What is the economic impact to biodiversity of forecast reductions in nitrogen pollution? Lastly, we calculate the damage cost per unit of nitrogen dioxide (NO_x) or ammonia (NH_3) emitted, for use in policy appraisal. These forms of nitrogen are emitted from two main sources: nitrogen dioxide primarily from combustion processes, and ammonia primarily from agricultural practices. Therefore, the effect of policies which only address emissions in particular sectors will vary spatially, eliciting different economic values.

Thus, we calculate the marginal value associated with a decline in nitrogen pollution and its subsequent impacts on the 'cultural' service 'Appreciation of biodiversity'. This service was identified in Jones et al. (2014) as requiring considerable development, in particular an improved evidence base for quantifying the nitrogen impacts and the development of spatial analysis. The approach taken focuses on one aspect of biodiversity –the non-use value component associated with conservation of species and maintaining species abundance. We use plant species richness as a proxy for the wider impacts of N deposition on biodiversity because responses of plant communities to N deposition are the best characterised of all organism groups, and because impacts on plants cascade up to higher trophic levels (Clark et al., 2017). We quantify the impact on species richness spatially in four habitats (heathland, acid grassland, dunes and bogs), and calculate the marginal economic value of declining nitrogen deposition per 5x5km grid cell of the UK, applying a value transfer procedure developed using data from Christie & Rayment (2012). Data are presented by region of the UK, including the uncertainty bounds for these estimates.

2. Materials and methods

2.1 Ecosystem services assessment: the Impact pathway for air (nitrogen) pollution.

We use the impact pathway approach (Friedrich and Bickel, 2001) for assessing the ecosystem services impacts of atmospheric nitrogen pollution (Figure 1). This shows how a policy initiative to curb air pollution results in a change in emissions of NO_x and NH_3 which leads, via changes in deposition, to an altered impact on biological receptors (plant species richness) and hence to the ecosystem service (Appreciation of biodiversity) they underpin. The steps are described in the following sections.

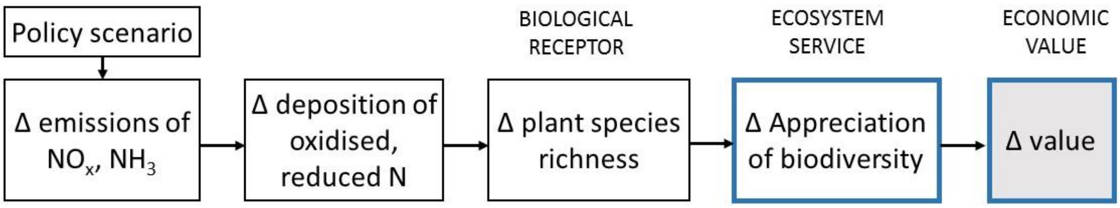

Figure 1. Impact pathway for nitrogen impacts on the ecosystem service 'Appreciation of biodiversity'. Blue outlines represent quantified impact on the ecosystem service.

2.2 Policy scenario, and nitrogen emissions and deposition

The first stage of the impact pathway is to specify alternative policy scenarios on the likely changes to N deposition. In this study, we compare a projected decline in N deposition from 2007 to 2020, against a counterfactual. Our scenarios were based on the UEP43 energy scenario 3 for 2020 (Misra et al., 2012). This scenario was seen as the most likely outcome of planned initiatives to reduce pollutant emissions across a range of sectors. The scenario estimated that policies designed to reduce air pollution emissions from combustion sources lead to a projected 37% decline in oxidised N emissions (nitrogen dioxides, NO_x), while policies to reduce emissions from agriculture lead to a projected 6% decline in the forms of reduced N from agriculture (primarily ammonia, NH_3). The counterfactual assumes emissions continue at 2007 levels. Thus, our scenarios essentially asks: "What is the expected impact on ecosystem service values under forecast reductions in nitrogen deposition"?

Nitrogen emissions data were obtained from Murrells et al. (2010) and Misra et al. (2012), while nitrogen deposition data were available at 5x5 km resolution across the United Kingdom. Deposition for 2007 used Concentration-Based Estimated Deposition (CBED) data (Centre for Ecology and Hydrology), taking a three-year average (2006-2008) to smooth inter-annual differences in deposition caused by variations in rainfall. Deposition for 2020 was calculated using the FRAME (Fine Resolution Atmospheric Multi-pollutant Exchange) model, 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 (Smith et al., 2000). FRAME model outputs were calibrated to CBED deposition in 2008.

2.3 Biological receptors: Dose response functions for nitrogen impacts on plant species richness

Four habitat types were selected that are known to be amongst the most sensitive to nitrogen deposition: acid grassland (Dupré et al., 2010; Stevens et al., 2004), upland and lowland ericoid heaths dominated by the shrub *Calluna vulgaris* (Pilkington et al., 2007; Power et al., 2006), sand dune grasslands (Jones et al., 2013; Plassmann et al., 2009; Remke et al., 2009) and bogs (Bragazza et al., 2012; Sheppard et al., 2011). Habitat area for these habitats was derived from CEH Land Cover Map 2007 (Morton et al., 2011), where acid grassland is defined as 'acid grassland' (class 8), heathland is defined as 'heather' (class 10) + 'heather grassland' (class 11), dune grassland is defined as 'supra-littoral sediment' (class 18) occurring within 2 km of the coast and where *Ammophila arenaria* was recorded in Biological Records Centre databases, and bogs were defined as 'bogs' (class 12).

The impacts of changing N deposition on biodiversity were calculated using dose response functions. These were developed from re-analysis of data from targeted gradient surveys of nitrogen impacts on plant species richness in the four selected UK habitats (Field et al., 2014). The nitrogen deposition gradients were characterised across a minimum of 20 sites for each habitat. Sites were selected to control for confounding effects of temperature and rainfall as far as possible. Total species richness of all vascular and lower plants at each site was summed over 5 quadrats, each of 2x2m, in total 20 m^2. Relationships for upland and lowland heaths were not significantly different and data were therefore combined. Dose response relationships were calculated by curve fitting in Sigmaplot v13.1, using AIC to determine the most parsimonious fit.

2.4 Ecosystem services: Valuation of change in ecosystem service provision

We utilised value transfer techniques (Johnston et al., 2015) to apply existing data on the value of biodiversity to our N deposition scenarios. The value transfer is based on Christie and Rayment (2012) who applied a discrete choice experiment (Louviere and Hensher, 1982; Louviere and Woodworth, 1983) to estimate willingness to pay (WTP) for the management of Sites of Special Scientific Interest (SSSI) for the provision of a suite of ecosystem services, under three funding scenarios. In this study we only used the ecosystem service attribute relating to species diversity for non-charismatic species[1], and for the habitats of interest in this study. WTP values were available for other services, including charismatic species, but these were excluded. We acknowledge that the parameters for non-charismatic species were not significant in the Christie study, but this remains the only study to our knowledge which quantifies and values the magnitude of change in biodiversity of non-charismatic species, allowing direct application to this study. Therefore, we decided to continue to use these values to demonstrate proof of concept for the overall methodology. Christie and Rayment (2012) specified a change in species richness for two scenarios: increase SSSI funding (25% increase in species richness), or remove SSSI funding (50% decrease in species richness), compared with the status quo of maintain SSSI funding (no change in species richness). We re-interpret the 'Increase funding' scenario as analogous to a situation where species richness increases relative to the status quo (2007 reference situation) due to a decline in N deposition, and we use the WTP estimates associated with that scenario as the basis for our value transfer, taking into account the predicted % change in species richness under our scenarios.

Christie and Rayment (2012) provide both unit WTP values per hectare for each habitat, based on habitat area within SSSI sites in England and Wales, and aggregate values for England and Wales. In

[1] Non-charismatic species include all plants, all insects apart from butterflies, in contrast to charismatic species such as birds, butterflies and animals (Christie & Rayment 2012).

this study we used the unit values per hectare, in order to scale up to the whole of the UK. The WTP per habitat is shown in Table 1.

2.5 Calculating economic impacts of N deposition on 'Appreciation of biodiversity' service

Our first economic measure relates to the impact that change in N deposition has on the value of the ecosystem service 'appreciation of biodiversity'. All ecosystem service calculations were made at the resolution of the N deposition data, i.e. on a 5 x 5 km grid. Nitrogen deposition data for each grid cell were scaled linearly between 2007 and 2020, the start and end time-points of the scenario comparison. In each 5 x 5 km grid cell and for each year of the scenario analysis, we calculated the predicted species richness under the N deposition for that year using the dose response relationships developed earlier. The percentage difference in species richness from the reference year was then calculated, as the basis for calculating economic value. The economic value was scaled according to the percentage change in species richness, relative to the percentage change in species richness used in Christie & Rayment (2012) – see Figure 2, to give a £ per ha for the change in species richness within each grid cell. This was multiplied by the area of habitat in each cell (Table 1).

	Heathland	Acid grassland	Dunes	Bogs	**Total 4 habitats**
WTP (£/ha)	£46.40	£44.45	£58.10	£57.55	**n/a**
<u>Habitat area (ha)</u>					
England	363,725	319,997	15,850	196,513	896,085
Wales	111,875	283,861	6,126	41,608	443,470
Scotland	1,567,895	1,023,537	19,505	769,461	3,380,398
Northern Ireland	73,971	21,709	1,502	92,808	189,990
UK	2,117,466	1,649,104	42,983	1,100,390	**4,909,943**

Table 1. WTP values per hectare for increase in diversity of non-charismatic species (Christie and Rayment, 2012) and area of each habitat (ha) (CEH Land Cover Map 2007) in the UK.

In each scenario year, the difference in value between the scenario and the counterfactual (reference scenario) was calculated. Values for all grid cells were aggregated to country and to national UK level. Aggregated economic values are presented in terms of an equivalent annual value (EAV) for the scenario, estimated as:

$$EAV = \frac{PV}{A_{t,r}} \qquad [1]$$

Where *PV* is the present value of the change in ecosystem service value and *A* is the relevant annuity factor for time horizon *t* with discount rate *r*. The present value of the change in ecosystem service value is estimated in the standard manner:

$$PV = \sum_{t=0}^{T} \frac{V}{1+r^{t}} \qquad [2]$$

Where *V* denotes the value of the change in ecosystem service provision. A discount rate of 3.5% was used, following UK Government guidance (HM Treasury, 2003). Calculation of the PV of the change in ecosystem service value provides an estimate of the accumulated damage to ecosystem services from air pollution over the 13 year duration of the scenario, whilst the EAV provides a measure of the annualised change in the value of the flow of ecosystem services for the scenario.

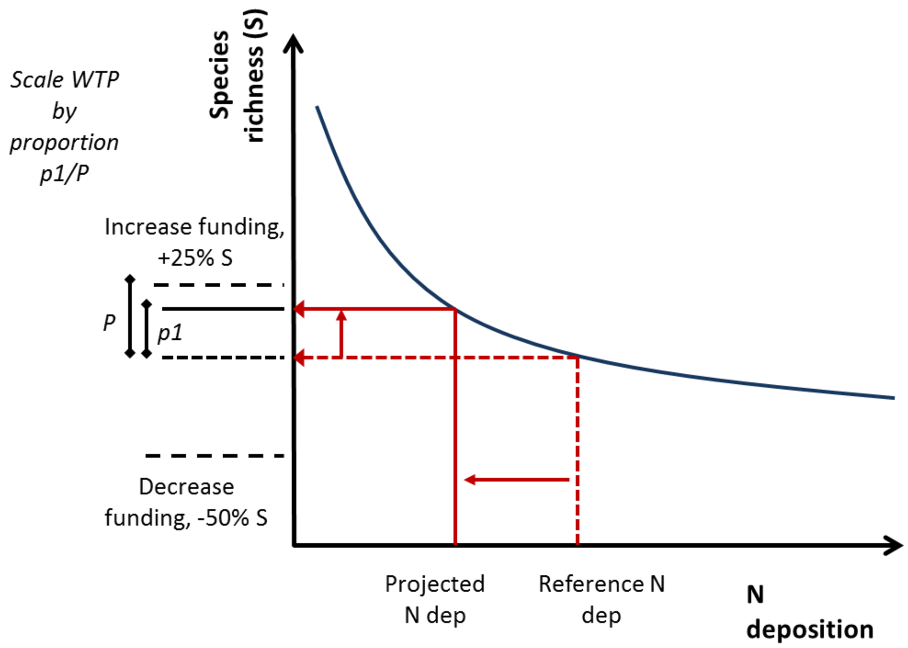

Figure 2. Scaling of changes in species richness and associated WTP relative to values in Christie & Rayment (2012). *p1* is the difference between species richness under the reference level of N deposition (counterfactual) and the projected N deposition. *P* represents the 25% increase specified in the choice experiment of Christie & Rayment. Values were scaled as the ratio of *p1/P* of the scenario WTP.

2.6 Calculating damage costs

Our second economic measure investigated related to the damage cost impacts per tonne of ammonia or tonne of nitrogen oxides emitted. This entailed separate calculation of the ecological impacts of ammonia and of nitrogen dioxide. There is currently no consensus on whether oxidised or reduced N is more damaging to plant species richness, and robust dose-response relationships do not exist separately for reduced forms of N and for oxidised forms of N (van den Berg et al., 2016). Therefore, for this study it was assumed that they have equal impact per unit of N deposited. Since the dose response functions we derived are based on total N deposition, separate oxidised or reduced N deposition cannot simply be substituted into the equation. Therefore the total impact in each year was calculated using total N deposition, and the value apportioned to oxidised or reduced N according to the proportion of change in the deposition of each N form. i.e. If total deposition declined by 2 kg N ha^{-1} yr^{-1} and 25% of this change (0.5 kg N ha^{-1} yr^{-1}) was in deposition of reduced forms of N, then 25% of the value was apportioned to reduced forms of N, and the remaining 75% to declines in oxidised N. The calculated EAV was divided by the average change in oxidised N emissions and in ammonia emissions over the scenario period (Table S1).

2.7 Uncertainty

There is uncertainty in all steps of the impact pathway, from estimates of nitrogen emission and deposition to the model parameters for the dose response functions. We used Monte Carlo simulation to propagate the uncertainty in the parameters and variables through the model, thereby calculating the uncertainty in the estimated value of impacts on biodiversity. Probability density functions were derived to describe the uncertainties in each model parameter and variable. Details are given in Tables S2 and S3 in Supplementary Material. We assumed that the uncertainties in the model parameters were at the UK scale and so for any one iteration of the Monte Carlo simulation the same values of the model parameters were applied in each grid cell. For other inputs the uncertainties were applied at the scale of a grid cell and assumed to be independent. We used @Risk software (Palisade Corporation, USA, 2010) to run the Monte Carlo simulation. We used Latin hypercube sampling and ran the simulation for 50,000 iterations. Uncertainty in the economic value of impacts is expressed as 95% Confidence Intervals. We followed the IPCC convention and assumed this interval to be defined by the 2.5th and 97.5th percentiles (Eggleston et al., 2006), while noting that this is not precisely the same as the usual meaning of a confidence interval in statistics.

3. Results

3.1 Change in N deposition

In response to the 37% decrease in emissions of nitrogen oxides and 6% decrease in ammonia emissions in our scenario, the average UK deposition projected by the FRAME model fell by 11%. This relatively small decrease is because approximately two-thirds of deposition is in the form of ammonia and other compounds of reduced N. Emissions from these compounds did not decrease as much as those of oxidised N. Figure 3 shows the spatial distribution of nitrogen deposition in 2007 and the change between 2007 and 2020. Nitrogen deposition is greatest in the uplands of north-west England and Wales, driven by high wet deposition in rainfall, and in large agricultural source areas such as Northern Ireland and in Norfolk in the east of England. By 2020, it is projected to decline in most areas, with the greatest decrease in areas which currently have high deposition, but will also decrease around large urban areas such as London. Nitrogen deposition at a few locations is projected to increase, attributed to expansion of localised point sources.

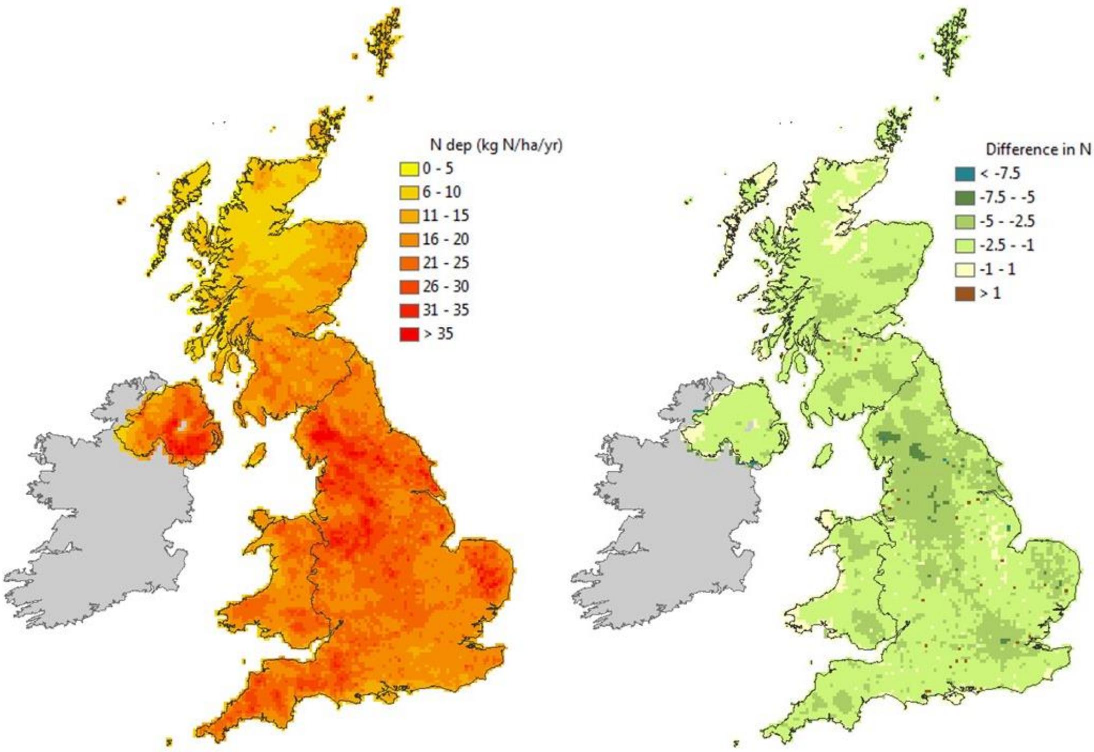

Figure 3. Nitrogen deposition in the UK (kg N ha^{-1} yr^{-1}) showing a) Spatial pattern in 2007, b) Forecast difference from 2007 to 2020.

3.2 Dose response functions for nitrogen and species richness

Log relationships provided the most parsimonious fit for all habitats except bogs, where a linear fit was the most appropriate (Figure 4). A quadratic relationship for acid grasslands gave a higher R^2, but was rejected due to the shape of the curve at high N deposition which predicted an increased species richness above 35 kg N ha^{-1} yr^{-1}, which was not supported by the data. All curves were significant. The equations for each habitat are summarised in Table 2.

3.3 Change in species richness due to nitrogen

In response to the general decline of N deposition, there is a corresponding predicted increase in species richness. The spatial pattern of increase reflects the combination of habitat location and declines in N deposition (Figure S1, Supplementary Material). Heathlands have the greatest UK coverage and show up to 20% increases in species richness with a spatial pattern reflecting that of changes in N deposition. Acid grasslands also occur widely across the UK, with greatest increases in species richness in the uplands of north-west England and Wales. Bogs have a more restricted distribution in the north and west UK, and show smaller increases, typically up to 10%, in species richness. Dune grasslands are distributed all around the UK coasts and show increases up to 20% in species richness.

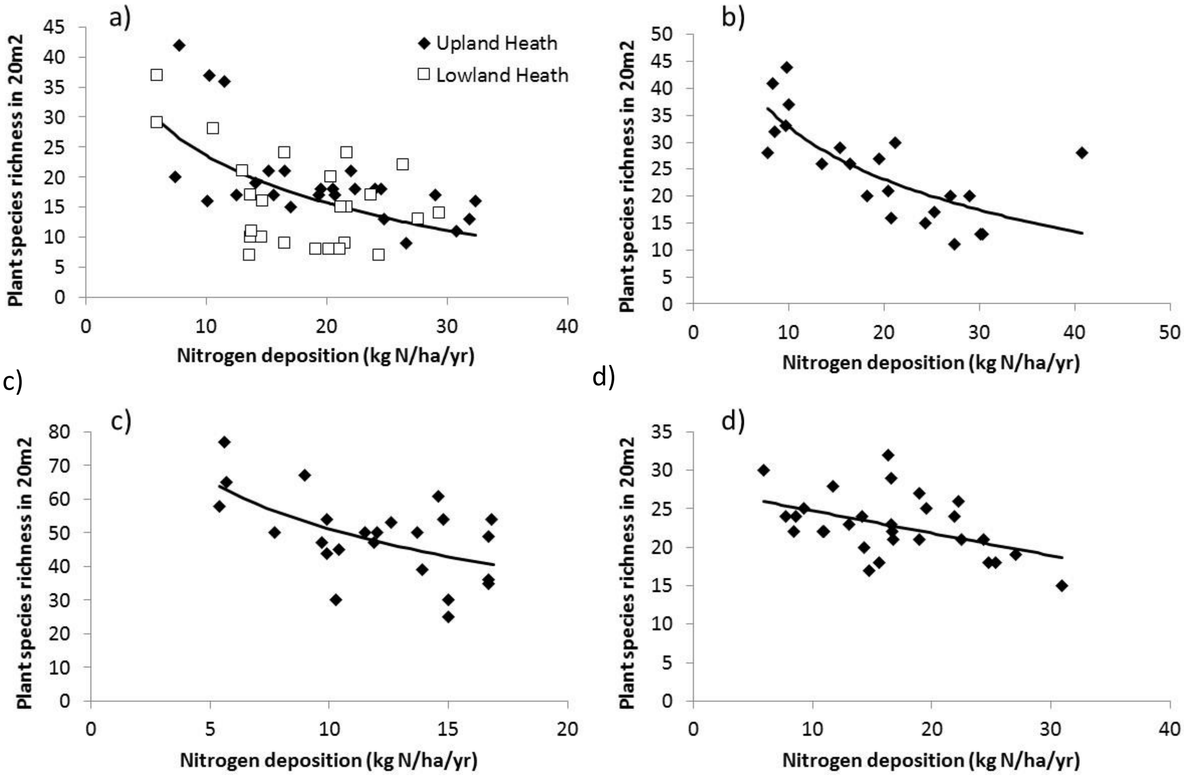

Figure 4. Dose response curves for nitrogen impacts on plant species richness for a) heathland, b) acid grassland, c) dune grassland and d) bogs, showing fitted equations (Table 2).

Habitat	Number of sites surveyed	N deposition range (kg N ha^{-1} yr^{-1})	Form of equation	Coefficients (SE)	R^2, SE, (Significance) of equation
Heaths: Upland + Lowland	25 + 27	5.9 – 32.4	f = y0 + a*ln(x)	y0 = 49.6654 (6.5632) a = -11.3114 (2.2716)	0.3315, 6.6414, (p<0.001)
Acid grassland	22	7.8 – 40.8	f = y0 + a*ln(x)	y0 = 65.1623 (7.927) a = -14.026 (2.7211)	0.5705, 6.1451, (p<0.001)
Dune	24	5.4 – 16.8	f = y0 +	y0 = 98.351	0.3346,10.2808,

grassland			a*ln(x)	(15.06) a = -20.4662 (6.1534)	(p=0.003)
Bogs	29	5.9 – 30.9	f = y0 + a*x	y0 = 27.6647 (1.9195) a = -0.2909 (0.1074)	0.2136, 3.6072, (p=0.012)

Table 2. Dose response equations linking N deposition to plant species richness. Data re-analysed from Field et al. (2014). Heath data from upland and lowland surveys were combined prior to analysis. Species richness was calculated as number of species in an area of 20 m^2 (five random quadrats of 2x2m).

3.4 Change in value of 'appreciation of biodiversity' ecosystem service

The economic value of projected declines in N deposition to 2020 on the ecosystem service 'appreciation of biodiversity' are shown in Table 3. Heathlands show the greatest benefit from declines in N deposition, with a projected benefit of £17.1 m (£2.7 – 56.0 m, 95% CI) EAV, while acid grasslands show a benefit of £12.2 m (£1.8 – 39.9 m, 95% CI) EAV. Despite their large area, the benefit to bogs is much lower £3.0 m (£0.3 – 10.7 m, 95% CI) EAV, since bogs occur primarily in lower deposition areas. Similarly, despite their high species richness, the limited area of dunes means the value to dunes is also relatively low at £0.2 m (£0.01 – 0.8 m, 95% CI) EAV. The combined annualised benefit to the whole UK is £32.6 m (£4.4 – 109.7 m, 95% CI) EAV. Figure 5 shows the spatial pattern in EAV from the four habitats combined. The combined benefit from reductions in N deposition is greatest in Scotland, and the upland areas of NW England and Wales reflecting the greater extent of the semi-natural habitats in these areas (Table 1). The economic benefit per ha (Figure 6) differs between habitats and is strongly non-linear, with the greatest economic benefit found at low levels of N deposition, with the exception of bogs which show a linear relationship.

3.5 Damage costs

The unit damage costs show the benefit to biodiversity per tonne decrease in emission of the main nitrogen compounds. For emissions of nitrogen oxides the benefit was £102.8 (£33.3 to £237.4, 95% CI) per tonne of NO_2 emission saved, and for ammonia the benefit was £413.8 (£139.1 to £1,021.5) per tonne of NH_3 not emitted.

Equivalent Annual Value	Heaths	Acid grassland	Dune grassland	Bogs	**Total 4 habitats**
England	£4.1m	£3.0m	£0.09m	£1.2m	**£8.3m**
Wales	£0.9m	£1.9m	£0.03m	£0.2m	**£3.0m**
Scotland	£11.7m	£7.3m	£0.1m	£1.4m	**£20.6m**
Northern Ireland	£0.4m	£0.1m	£0.008m	£0.2m	**£0.7m**
UK (95% CI)	**£17.2m (£2.7m to £56.0m)**	**£12.3m (£1.8m to £39.9m)**	**£0.2m (£0.01m to £0.8m)**	**£3.0m (£0.3m to £10.7m)**	**£32.7m (£4.4m to £109.7m)**

Table 3. Equivalent Annual Value of nitrogen impacts on appreciation of biodiversity for non-charismatic species, by country and by habitat, future scenario (95% Confidence Intervals).

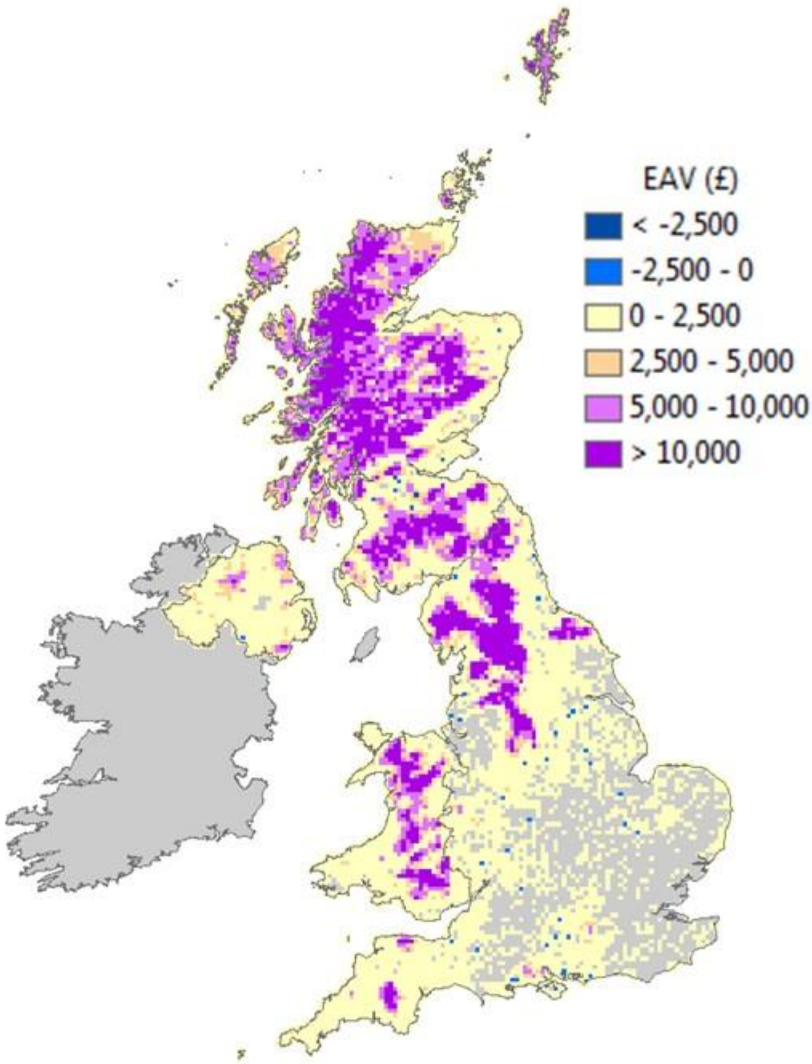

Figure 5. Spatial pattern of equivalent annual value (EAV) resulting from projected declines in N deposition impacts on biodiversity (£ per 5x5km grid cell).

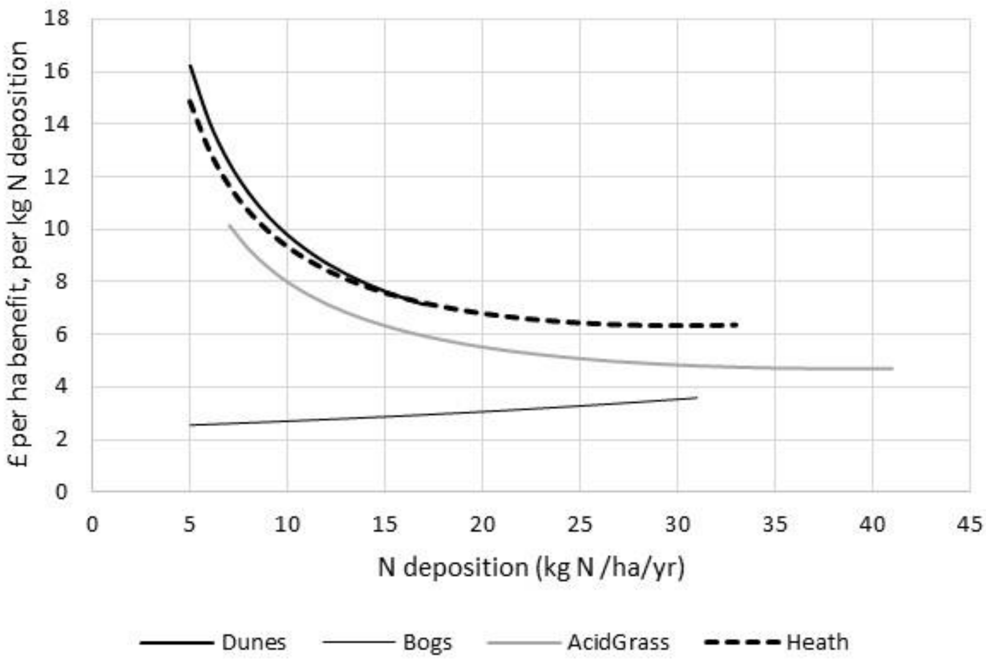

Figure 6. Marginal cost response curves showing change in value of economic benefit of a 1 kg N/ha/yr pollutant reduction, depending on initial level of N deposition (£ per ha, per unit change in N deposition).

4. Discussion

In this study we developed a spatially-explicit methodology to quantify N impacts on biodiversity, and a value transfer function to calculate the marginal value of changes in N deposition. We used this to quantify the economic value of reductions in nitrogen deposition on a cultural ecosystem service "Appreciation of biodiversity" at national scale, and to calculate the damage cost per tonne of nitrogen dioxide or ammonia emitted, for use in policy appraisal.

4.1 Economic values and damage costs

This study uses a spatially explicit approach to calculate N impacts on ecosystem services, which is more robust than previous studies using national figures only (Jones et al., 2014; Smart et al., 2011), and makes use of new data to calculate dose response functions linking N deposition and species richness (Field et al., 2014). The value transfer approach provides direct linkage between response functions for changes in species richness and the WTP values, demonstrating a clear impact pathway. Spatial context is a key component of ecosystem service assessment where location plays a part in determining the amount of benefit supplied, or where the spatial location of supply and beneficiaries differ (Eigenbrod et al., 2010). In this study, the considerable spatial variation in benefit supply arises from the congruence of the pressure affecting the ecosystem and where the benefits are provided. The importance of incorporating spatial context is illustrated by the value calculated for bogs which, despite covering an area almost half that of heathland, have annualised benefits less than one fifth that of heathland due to their spatial location in relation to the changes in N deposition.

This study also calculates primary estimates of damage costs for N impacts on biodiversity. While the values we calculate (£414 per tonne of ammonia) are somewhat lower than the value of £1,972 (2010 prices) recommended for UK policy appraisal of human health impacts related to the $PM_{2.5}$ aerosol component of ammonia (Dickens et al., 2013), they represent a previously unquantified component of air pollution impacts on the environment.

4.2 Valuation methods

Our analysis utilised WTP value data from Christie and Rayment (2012), which assessed the UK public's WTP for changes to non-charismatic species riches at different protected (SSSI) habitats. The population base for the economic values, the types of habitats valued and the percentage changes in species richness are consistent between their study and ours. Therefore, we are reasonably confident that the use of these data for value transfer is acceptable. WTP values may differ spatially either in terms of (i) the differences in the socio-economic attributes of people living in different locations or (ii) the accessibility to substitute sites. While robust data on the spatial variation of values was not available from Christie and Rayment (2012), an earlier study looking at WTP to protect UK Priority Habitats for conservation (Christie et al., 2011) showed no significant effect of regional variation in WTP values. Therefore, our analysis assumes that values are spatially homogenous. The Christie et al. studies only estimated WTP values for England and Wales. Our extension of these values to Scotland and Northern Ireland carries assumptions that WTP does not vary by country outside of the original studies. Our analysis incorporated differences in habitat area in these countries at a fine spatial scale (5x5 km), but did not adjust for potential differences in WTP, since average levels of household disposable income for Scotland and Northern Ireland are within or very close to the range of average disposable income in England and Wales.

Since the valuation focuses on the non-use component of biodiversity in the form of existence value for non-charismatic species as a final service, it does not capture the contribution of biodiversity to direct and indirect use values; i.e. the value that is embedded in production of crops, regulating climate, recreation, etc., nor the 'value' that biodiversity can have in terms of resilience and supporting continuing flows of ecosystem services (Baumgartner, 2007; Kumar and Kumar, 2008). In this way, we avoid issues of double accounting. However, we are also assuming 'constant flow' over time. This is not problematic so long as current flows are sustainable; i.e. we are assuming the resilience function of biodiversity is not impaired. If the resilience function is depleted, then potential thresholds and non-linear effects may come into play and the value could be considered an underestimate (Baumgartner, 2007).

4.3 Response functions

The non-linear response function in all habitats except bogs shows that the majority of biological impact on plant diversity occurs at relatively low levels of N deposition, but that it continues to have an impact at higher N deposition. This has consequences for valuation in that a unit change in N deposition will have a greater value at low N deposition than at high N deposition, because the ecological impact on species richness is greater.

The response functions use species richness as a metric to represent biodiversity in common with many other studies. However, this may mask more complex biological impacts. For example where species of conservation interest are replaced by other, faster growing, nitrogen-loving species (Hodgson et al., 2014), this may result in no net change in species richness, despite substantial changes in species composition. There was no evidence of such changes in the data underpinning this study (Field et al., 2014). However, other metrics such as difference from a pristine reference species composition, e.g. Mean Species Abundance (Alkemade et al., 2009) could be used instead. Using a different biodiversity metric may then require a modified value-transfer approach.

4.4 Assumptions

A number of assumptions underlie these calculations. Economic theory suggests that values of biodiversity appreciation may be non-linear: i.e. marginal value per species is likely to decline as species richness increases or there may be thresholds which result in marked changes in value (Kumar, 2010). Other non-linearity effects due to scope insensitivity in the WTP study may influence our scaling assumptions, in which we used a value per habitat based on its coverage within protected areas and scaled it up to its extent nationally on the assumption that the value would increase linearly with area. In the absence of more detailed information, we assumed a linear response in both cases. Alternative approaches to value nitrogen impacts could include restoration cost (Van Grinsven et al., 2013), the estimated cost of restoring an ecosystem from its degraded state, or a Regulatory revealed preference cost which assumes that all costs of managing protected areas, including to manage impacts of drivers such as nitrogen deposition, were built into the funding model. These techniques also carry major assumptions, for example the restoration cost approach assumes that the cost of replacing an ecosystem or its services is an estimate of the value of the ecosystem or its services (Ott et al., 2006).

From a nitrogen impacts perspective, the calculations assume that biological response to a change in N deposition occurs within a year. In reality, there are lags in the response of plant communities to changes in N deposition due to species persistence effects and continued cycling of stored N in the soil (Rowe et al., 2017). The complexity and varying timescales of these interactions make it difficult to incorporate them in this sort of economic appraisal currently.

The majority of species with clear response functions for N impacts can be classed as non-charismatic species. However, there is emerging evidence of impacts on more charismatic species such as butterflies (Wallis de Vries and Van Swaay, 2006) and on birds via impacts on prey items (Nijssen et al., 2001). WTP values for charismatic species are far greater than for non-charismatic species (Christie and Rayment, 2012; Loomis and White, 1996a, b). However, at present it is not possible to model impacts of air pollution on these species due to a lack of dose response functions. This remains an important evidence gap that requires further research.

5. Conclusions

In conclusion, we demonstrate the potential for spatially-explicit calculation of pollutant impacts, by combining dose-response functions for nitrogen impacts on plant species with a well-aligned WTP study, and that it is possible to then value pollutant impacts on biodiversity, albeit with large uncertainty bounds. This demonstrates an approach that can be applied with other services and in other contexts, particularly as new relevant WTP studies emerge in the literature.

This study provides clear potential for an economic benefit to biodiversity from policies which reduce N deposition. The spatial pattern of the supply of benefit varies considerably and accounting for this spatial variation is essential to correctly quantify those impacts. The response itself is non-linear, and the greatest benefit comes from reducing nitrogen pollution in areas which are still relatively un-impacted.

From a policy perspective there are two messages. Avoiding damage to habitats which are still relatively un-impacted will have the greatest economic value. However, there is also continued economic benefit to reducing N deposition to habitats which already receive high levels of N deposition. The study also provides an indicative estimate of the potential damage costs due to adverse effects on non-charismatic species, which can be considered in the context of existing health damage costs. Understanding the spatial context to those impacts can help design intervention measures to alleviate pollutant pressures in particular locations or regions.

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7. References

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Table S1. Change in emissions of NO_2 and NH_3 used to calculate damage costs for the future scenario. Emissions are scaled linearly between start and end years of the scenario.

	NO_x as NO_2		NH_3	
Year	NO_2 Emissions (kt)	Change from baseline	NH_3 Emissions (kt)	Change from baseline
2007	1403.0	0.0	289.6	0.0
2008	1363.1	-39.9	288.2	-1.4
2009	1323.1	-79.9	286.9	-2.7
2010	1283.2	-119.8	285.5	-4.1
2011	1243.3	-159.7	284.2	-5.5
2012	1203.3	-199.7	282.8	-6.8
2013	1163.4	-239.6	281.4	-8.2
2014	1123.5	-279.5	280.1	-9.5
2015	1083.5	-319.5	278.7	-10.9
2016	1043.6	-359.4	277.3	-12.3
2017	1003.7	-399.3	276.0	-13.6
2018	963.8	-439.2	274.6	-15.0
2019	923.8	-479.2	273.2	-16.4
2020	883.9	-519.1	271.9	-17.7
Average change (kt)[1]		-279.5		-9.5

[1] Not including Reference Year.

Table S2. Assumptions and parameterisation used in the uncertainty analysis

Variable	Assumptions and parameterisation
Spatially variable N deposition	Uncertainty for each predicted value of N deposition was distributed log-normally with a standard deviation of 25% of the mean (this approximates 95% confidence limits of ±50%) (Jones et al. 2016). We used a log-normal distribution because the standard deviation was large, thereby avoiding negative values which would result from a normal distribution. Correlation in errors between the values in 2007 and 2020 was estimated as 0.99.
Response function (slope of $y = ax + b$ relationship)	Based on examination of the data, uncertainty in the model parameters was distributed normally with means standard deviations and correlations listed in Table S3 below.
Percentage area of habitat in 5x5km square	Uncertainty in the percentage of each habitat across the UK had a triangular distribution with limits ±5% of the mean.
Maintain/Increase Funding	Based on the information in Christie et al. (2012). Willingness To Pay values for non-charismatic species were distributed log-normally with standard deviation 65% of the mean. We used a log-normal distribution because the standard deviation was large. The uncertainty in this variable does not account for the uncertainties accumulated when aggregating from the price per 1% change in unit (£/household/year) as this information was not available.

Table S3. Parameters for response functions in uncertainty analysis.

	Means		Standard deviations		Correlations
	a_m	b_m	a_s	b_s	
Heaths	-11.3	49.67	2.27	6.56	-0.99
Acid grassland	-14.0	65.15	2.72	7.93	-0.99
Dunes	-20.5	98.25	6.15	15.06	-0.99
Bogs	-0.29	27.66	0.11	1.92	-0.94

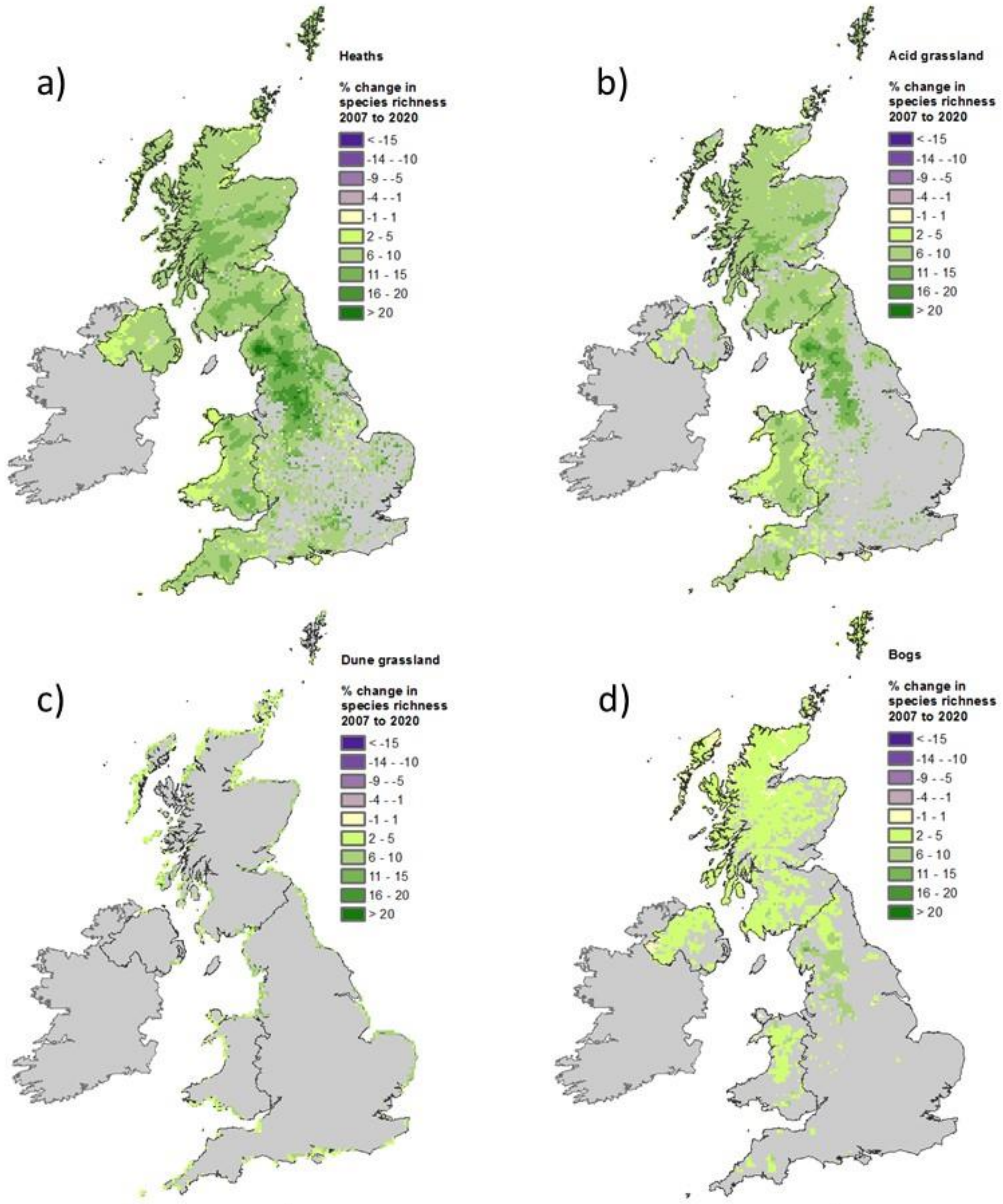

Figure S1. Projected changes in species richness due to declines in nitrogen deposition, for four habitats: a) heaths, b) acid grassland, c) dune grassland, d) bogs.