The use of National Plant Monitoring Scheme data for making inferences concerning air pollution impacts

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Executive Summary

- The National Plant Monitoring (NPMS), a partnership between the BSBI, CEH, JNCC and Plantlife, is a volunteer-based, habitat and plant monitoring scheme conceived as a representative sample of high quality semi-natural habitats across the UK.
- The habitat classification scheme used by the NPMS was developed in relation to the UK National Vegetation Classification (NVC) and the European Nature Information System (EUNIS) classifications, meaning that there are clear correspondences between habitat schemes.
- Many NPMS habitats have already accumulated more than 30 samples in the first two years of the scheme (2015-16). Although this does not directly map to the number of samples per species, it suggests that the power to detect moderate change (e.g. a 10% change over ten years) in species richness in these habitats is likely to be high in the most popular habitats (i.e. dry acid grassland; dry calcareous grassland; dry deciduous woodland; hedgerows of native species; neutral damp grassland; neutral pastures and meadows; and common wetland habitats).
- There is much existing evidence in the scientific literature for links between the previously selected NPMS Indicator species and the deposition of nitrogen.
- Comparison with a contemporary professional survey (the Welsh Glastir Monitoring and Evaluation Programme) suggests that NPMS full plot ('Inventory') and indicator data are likely to be biased. These biases should be re-evaluated periodically, and analytical adjustments for them should be investigated.
- The uptake of NPMS 1 km squares currently under-represents upland habitats. However, current N deposition gradients in NO_x and NH_y appear to be covered by allocated and surveyed squares. The detection of air pollution impacts are therefore likely to be correspondingly biased towards lowland habitats.
- Key air pollution impact metrics that could be created for NPMS data include mean Ellenberg N or R and species or indicator richness. The scientific literature suggests that these metrics are meaningful and can be directly related to the impacts of nitrogen deposition. The NPMS partnership should also consider the inclusion of assessments of grass:forb cover ratio and assessments of flowering in its field protocol. These are both relatively easy for volunteers to assess, and have been noted in the literature as being related to N deposition and ozone impacts respectively. Implementing changes to the NPMS field protocol, however, is dependent on agreement with all NPMS partners, and an assessment of the capacity of NPMS volunteers to perform additional tasks within an already demanding structured survey framework.
- An analysis of the likely power of the NPMS to detect changes in species richness suggests that the smallest simulated change in richness (a 10% increase or decline over ten years) is only likely to be detectable in a subset of habitats at the current time, depending on the typical richness in that habitat and the number of available plots. Habitats that are well covered by the current resource are listed above (bullet 3); relatively poorly represented habitats of relevance for air pollution impacts include: blanket bog; coastal sand dunes; montane calcareous and dry heathlands; and raised bog.
- The 'Indicator' species chosen by the NPMS partnership do, however, appear to capture broad UK ecological gradients, suggesting that they provide a good basis for inferring the main types of ecological change likely to be observed, strengthening the value of these plots for ecological inference.

- WP2 Task 3 reviews the available air pollution datasets currently available for correlative modelling of vegetation response variables. With the exception of O_3 (available only from EMEP4UK at 5 x 5 km resolution at the current time), the 1 x 1 km predictions of the FRAME model are likely to be provide the most analytical power for inferring relationships between nitrogen deposition and NPMS response data at the current time.
- Recently developed Bayesian modelling frameworks can be used to estimate the impacts of nitrogen deposition on species richness whilst taking account of surveyor level, habitat, and spatio-temporal correlation structures. Initial estimates of the effects of N deposition for dry acid grassland and dry calcareous grassland species richness are presented and discussed.
- In conclusion, the efforts of NPMS surveyors to collect plant community data on Britain's semi-natural habitats represents an important and growing resource. However, unevenness in square uptake, and other biases at the plot and species level, mean that care should be taken with respect to inference and model construction. Metrics representing air pollution impacts should be possible for well-represented habitats, whilst the future development of the scheme will take expansions suggested here into consideration. Communication of these issues with volunteers through the NPMS should also increase awareness of issues pertaining to air pollution, and encourage volunteers to stay with the scheme.

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Work Package 1

Introduction

The National Plant Monitoring Scheme (NPMS) is a recently developed, volunteer-based, habitat and plant monitoring scheme (Walker *et al.*, 2015). It was conceived as a national sample of high quality semi-natural habitats, but one which was straightforward enough in design to be appealing to volunteer botanical recorders. Comprehensive information on the founding motivations and design of the scheme is available in several reports and papers (e.g. Walker *et al.*, 2010, 2015; Pescott *et al.*, 2014, 2015, 2016, In review), and we do not review all of this information here. Unpublished reports cited herein are to be made available on the scheme website (http://www.npms.org.uk/content/conservation-and-research), or are being prepared for publication in the peer-reviewed literature. The current report deals with information on NPMS habitats, and the current recording of these habitats, in order to inform decisions on the potential utility of NPMS data for assessing the impacts of air pollutants, especially nitrogen, on semi-natural habitats. In general, we assume that this type of research would be framed within a correlative mode, but we do not assess all of the general limitations of this type of approach here (however see Smart *et al.*, 2012 for a concise review of some of the limitations that can arise within this framework).

Here we: (**1**) provide comprehensive information on the equivalences of the broad- and fine-scale habitats, as defined within the NPMS, with other habitat classification frameworks (EUNIS, Annex 1 (Habitats Directive) habitats, and the National Vegetation Classification) that may be of importance for reporting within larger political boundaries, or for the amalgamation or comparison of data with other monitoring programmes. Note that these equivalences were used throughout the development of the NPMS, rather than only being considered subsequent to its finalisation (Pescott *et al.*, In review); (**2**) provide summary information on the numbers of plots recorded at the fine-scale habitat level by the most experienced recorders during the first two field seasons of the NPMS, and link these habitats, and the species recorded therein, to published critical loads for nitrogen; (**3**) present a recent assessment of recorder bias performed for a subset of NPMS data within Wales. (This assessment provides information that will support the accurate interpretation of NPMS data across subsequent analyses, and provide a framework for future quality assurance.) Finally, under (**4**), we present an assessment of how current NPMS 1 km squares (as represented by the 10 x 10 km squares within which they are nested) are currently representative of large-scale environmental gradients, including a subset of important air pollutants (as represented by estimates from the FRAME model of pollutant deposition; Dore *et al.*, 2007).

Task 1. NPMS habitat equivalences (EUNIS/Annex 1/NVC)

The tables below summarise equivalences between the final NPMS broad- and fine-scale habitat classifications and EUNIS Levels 1-3 (Davies *et al.*, 2004) and Annex 1 habitats (Table 2) and National Vegetation Classification communities (Table 3; Rodwell, 1991 *et seq.*). The NVC habitat equivalences were used directly in the selection of the fine-scale habitat-specific plant indicator species (Pescott *et al.*, 2014, In review.) Although a final decision was made to define NPMS habitats in terms of the NVC, the EUNIS definitions were considered alongside development of the NVC classification throughout the project, rather than being matched as an addendum to the habitat development work; indeed, the NPMS design process began with EUNIS Level 2 as the basis for its fine-scale habitat classification. Correspondence, therefore, should be close. Our NVC definitions, and the final names of the NPMS habitats, were subject to external expert review; Table 1 lists the experts involved in this process and their areas of expertise.

The habitat correspondence tables given below are also now available as PDFs on the NPMS website at http://www.npms.org.uk/content/conservation-and-research. Excel versions are available on request from the lead author of this report. The indicator species chosen as a result of the NPMS to NVC habitat definition work are available in the NPMS volunteer support pack: see 'NPMS Species lists' at http://www.npms.org.uk/content/resources.

Table 1. The habitat experts consulted and habitats reviewed.

Expert	Organisation	Area of expertise	Habitats reviewed
Iain Diack	Natural England	Wetland habitats, bogs, mires, wet heath	Wet grassland, fens, mires, bogs, swamps
Ian Strachan	SNH/JNCC (former)	Upland habitats, standing waters, coastal	Montane, bogs, mires, standing waters, coastal
Richard Jefferson	Natural England	Grasslands	Dry grasslands
Stuart Smith	Natural Resources Wales	Grasslands, fens	Wet and dry grasslands, fens and mires

Table 2. NPMS habitat to EUNIS and Annex 1 community correspondence table.

NPMS Fine Habitat	Priority habitat	EUNIS Level 1	EUNIS habitats (Level 2-3)	Annex I habitats
Arable field margins	Arable Field Margins	I	Part of I1.1, I1.3, I1.4, I1.5	None
Raised Bog	Lowland raised bog	D	In UK more or less equivalent to D1.1 Raised bogs	Includes both H7110 Active raised bogs and H7120 Degraded raised bogs. Also an overlap with H3160 and H7150.
Blanket Bog	Blanket bog	D	Equal to D1.2 Blanket bogs	Equal to H7130 Blanket bogs. Also an overlap with H3160, H7140 and H7150.
Wet heath	Lowland Heathland	F	F4.1	H4010, H4020
Hedgerows of native species	Hedgerows	F	New definition will include FA.2, FA.3 and FA.4.	None
Wet woodland	Wet Woodland	F/G	Contains F9.2, G1.1, G1.4 and G1.5	Contains H91E0
Dry deciduous woodland	Lowland Beech and Yew Woodland	G	Beech dominated areas will be equivalent to the UK part of G1.6, whilst yew dominated areas will be equivalent to the UK part of G3.9.	Contains H9120 and H9130, as well as most of H91J0
	Lowland Mixed Deciduous Woodland	G	Not really divided into upland / lowland. Will contain lowland components of G1.8, G1.9 and G1.A	Contains H9160 and part of H9190 and possibly H9180.
	Upland Oakwood	G	Not really divided into upland / lowland. Will contain some upland components of G1.8 and G1.A	Probably equivalent to H91A0 Western acidic oak woodland
	Upland Birchwoods	G	Upland component of G1.9	None, or possibly some of H91A0
	Upland Mixed Ashwoods	G	A part of G1.A	Overlaps with H9180 and part of H91J0
Coastal saltmarsh	Coastal saltmarsh	A	Largely equivalent to A2.5 Coastal saltmarshes and saline reed beds, although this may be defined slightly more broadly.	Contains H1330 Atlantic salt meadows, H1420 Mediterranean saltmarsh scrub, H1320 Cord-grass swards and H1310 Glasswort and other annuals colonising mud and sand. Spartina is on the NPMS species list for coastal saltmarsh.
Coastal sand dunes	Coastal sand dunes	B	Included within B1 – probably B1.3, B1.4, B1.5, B1.6 and B1.8.	Contains H2250, H2190, H2170, H2160, H2150, H2140, H2130.
Machair	Machair	B	Equivalent B1.9 Machair	Equivalent H21A0 Machair
Coastal vegetated shingle	Coastal vegetated shingle	B	Contained within B2 Coastal shingle – probably B2.3, B2.4 and B2.6	Contains H1220 Coastal shingle vegetation outside the reach of waves. Overlaps with H1210 Annual vegetation of drift lines (this also includes other substrates). Annex I habitat broader than Eunis level 3.
Maritime cliff and slopes	Maritime cliff and slopes	B	Includes B3.2, B3.3 and B3.4	Equivalent to H1230 Vegetated sea cliffs
Nutrient poor lakes and ponds	Oligotrophic and dystrophic lakes; Mesotrophic lakes; Ponds	C	Includes parts of C1.1 and C1.4, although note that the littoral zone (which would be surveyed) are C3 habitats	Includes parts of H3110, H3130, H3140 and H3160. Annex 1 habitats do not distinguish between lakes and ponds
Nutrient rich lakes and ponds	Eutrophic standing waters; Ponds	C	Includes part of C1.2 and C1.3, although note that the littoral zone (which would be surveyed) are C3 habitats	Includes most of H3150. Annex I habitats do not distinguish between lakes and ponds.
	Reedbeds	A/B/C/D	May occur in a very wide range of EUNIS types – not compatible with this classification	No Annex I habitats
Rivers and Streams	Rivers	C	Will include part of C2.1, C2.2, C2.3 and C2.4 – the priority habitat is focussed on near-natural and other high status rivers	Includes H3260 which overlaps with C2.1, C2.2 and C2.3 in Eunis

Table 2 cont.

NPMS Fine Habitat	Priority habitat	EUNIS Level 1	EUNIS habitats (Level 2-3)	Annex I habitats
Dry heathland	Lowland Heathland	F	Contains the lowland component of F4.2	Contains some of H4030. The whole of H4040 is lowland in occurrence.
Montane dry heathland	Mountain Heaths and Willow Scrub	E/F	F2.2, F2.3	H6150, most of H4060
	Upland Heathland	F	Contains the montane component of F4.2	Contains some of H4030. Also contains some of H4060.
Dry calcareous grassland	Lowland Calcareous Grassland	E	Not divided into lowland/montane. Contains parts of E1.2.	Includes H6210, H6211, H5130 and some of H6230 and H6170
	Calaminarian Grasslands	E	Equivalent to E1.B	Equivalent to H6130 Calaminarian grasslands of the Violetalia calaminariae
Dry acid grassland	Lowland Dry Acid Grassland	E	Not divided into lowland/montane. Contains part of E1.1, E1.7, E1.9.	Includes H2330 Inland dunes with open Corynephorus and Agrostis grasslands
Neutral pastures and meadows	Upland Hay Meadows	E	UK component of E2.3	Equivalent to H6520 Mountain hay meadows
	Lowland Meadows	E	Overlaps with E2.1, E2.2 and E3.4. Distinctions are due to whether it is managed for hay, and whether there is seasonal inundation.	Includes H6510 Lowland hay meadows
Neutral damp grassland	Coastal and Floodplain Grazing Marsh	E	Difficult to classify. In many cases may be within E3. However, much of the plant interest can be in drainage ditches, more like E5.4.	Some of H6510 Lowland hay meadows are included.
Acid Fens, mires and springs	Lowland Fens; Upland Flushes, Fens and Swamps	D	Includes some of D2.1, D2.2, D2.3, some of D4.1	Part of H7140 and H7150
	Purple Moor Grass and Rush Pastures	E	Equivalent to E3.51	Includes H6410 Purple moor-grass meadows.
Base-rich fens, mires and springs	Lowland Fens; Upland Flushes, Fens and Swamps	D	Includes some of D2.1, most of D4.1, D4.2	H7230, H7220, H7210, H7240 part of H7140
Montane calcareous grassland	Upland Calcareous Grassland	E	E4.4 and parts of E1.2.	May contain some of H6210, and some of H6230 and H6170.
Montane acid grassland	Mountain Heaths and Willow Scrub	E/F	E4.3	
Conifer woods and juniper scrub	Native Pine Woodlands	G	Equivalent to G3.41	H91C0
Montane rocks and scree	Mountain Heaths and Willow Scrub		E4.1, E4.2, F2	H4080
	Inland Rock Outcrop and Scree Habitats	H	A very broad habitat type, including some of H2 and H3 (with the exception H3.5 Pavements) – breadth due to both geology and whether it is an outcrop or a scree.	Includes H8110 Acidic scree and H8120 Base-rich scree. On other outcrops, includes H8210 Plants in crevices in base-rich rocks and H8220 Plants in crevices on acidic rocks, as well as part of H6430 Tall herb communities which occurs on ledges.
Inland rocks and scree	Inland Rock Outcrop and Scree Habitats		A very broad habitat type, including much of H2 and H3 (with the exception H3.5 Pavements) – breadth due to both geology and whether it is an outcrop or a scree.	Includes some of H8210, H8220 and H6430.
	Limestone Pavements	H	A part of H3.5	Equivalent to H8240 Limestone pavements

Table 3. NPMS habitat to NVC community correspondence table.

NPMS Broad Habitat	**NPMS Fine Habitat**	**NVC communities included**
Arable margins	Arable field margins	OV1-11, 13-17
Bog and Wet Heath	Raised bog	M1-3, 18, 19, 20, 25
	Blanket bog	M1-3, 17-20, 25
	Wet heath	M15, 16
Broadleaved woodland, hedges and scrub	Hedgerows of native species	W21-25
	Wet Woodland	W1-7
	Dry Deciduous woodland	W8-17
Coast	Coastal saltmarsh	SM10-28
	Coastal sand dunes	SD7-18
	Machair	SD8
	Coastal vegetated shingle	SD1
	Maritime cliffs and slopes	MC5, 8-12; H7; CG1
Freshwater	Nutrient poor lakes and ponds	A7, 13-14, 22-24; S4, 8-10, 19
	Nutrient rich lakes and ponds	A1-17, 19-21; S1-8, 10-19, 22, 24, 25, 28
	Rivers and streams	A1-2, 5-21; S4-7, 11-19, 22, 28
Heathland	Dry heathland	H1-8, 11
	Dry montane heathland	H4, 8-10, 12-22
Lowland grassland	Dry calcareous grassland	CG1-10
	Dry acid grassland	U1-6, 20; SD10-11
	Neutral pastures and meadows	MG1-6
	Neutral damp grassland	MG8-13
Marsh and fen	Acid fens, mires and springs	M4-8, 21, 23, 25, 27-29, 31-35
	Base-rich fens, mires and springs	M9-14, 22-28, 37-38; S1-3, 9-13, 24-27
Upland grassland	Montane calcareous grassland	CG9-14
	Montane acid grassland	U2-14, 19
Native Pinewood and juniper scrub	Conifer woodlands and Juniper scrub	W18-19
Rock outcrops, cliffs and scree	Inland rocks and scree	CG1, 9-10; OV37-40
	Montane rocks and scree	CG14; OV37, 40; U15-18, 21; W20

Task 2. Habitat representation in the NPMS (2015-16): Defining the current resource

Table 4 summarises habitat data received by the NPMS in 2015 and 2016. Surveyors within the NPMS can choose to survey habitat plots at one of three different levels: Wildflower, Indicator and Inventory (Walker *et al.*, 2015). Surveyors can also choose whether to record at the level of the NPMS broad habitat or at the fine-scale level (see WP1 Task 1; Walker *et al.*, 2015). Table 4 provides counts of annual survey visits at the fine habitat level over the two years of the NPMS to date. Counts for the more complete Indicator and Inventory level plots are shown only; the Inventory level plots are complete samples (*i.e.* all species should be recorded), whilst the shorter lists of species searched for by those surveying at the Indicator level have been shown to retain a high level of ecological information (Pescott *et al.*, In review).

The best represented fine-scale habitats are within the ‘Broadleaved woodland, hedges and scrub’ and the ‘Lowland grassland’ broad categories (see WP1 Task 1 above); arable field margins are also well represented. Work on power within the NPMS came to the general conclusion that around 30 plots for a species would be necessary for analysts to detect declines within 10 years (Pescott *et al.*, 2016, In review); many of the fine-scale habitats below reach this level (Table 4), although this does not mean that any given species within a habitat will have 30 recorded occurrences. More work on species-level power is planned in WP2 Task 2.

Tables 5 and 6 link NPMS habitats and species respectively to published estimates of responses to nitrogen deposition. In the case of habitats, these are given as critical load exceedance values; for species they are indicators of the direction of change source from individual studies and experiments. Overall, evidence for change linked to nitrogen deposition has been found for 51 NPMS indicator species, although this is sometimes contradictory between studies. Contradictory evidence may be due to statistical variance and/or bias, or real ecological (*i.e.* local context) difference. Note also that some evidence presented by primary studies is in the form of ordinations or other summaries, and that therefore individual effects on species cannot always be easily extracted.

Table 4. Counts of annual survey visits to plots recorded at the fine habitat level in 2015 and 2016.

Fine-scale habitat	Indicator samples	Inventory samples
Acid fens, mires and springs	30	22
Arable field margins	104	76
Base-rich fens, mires and springs	11	28
Blanket bog	18	20
Coastal saltmarsh	24	25
Coastal sand dunes	7	29
Coastal vegetated shingle	35	21
Dry acid grassland	23	41
Dry calcareous grassland	60	60
Dry deciduous woodland	158	167
Dry heathland	71	35
Hedgerows of native species	200	164
Inland rocks and scree	19	14
Maritime cliffs and slopes	27	5
Montane acid grassland	5	33
Montane calcareous grassland	1	5
Montane dry heathland	7	5
Montane rocks and scree	4	7
Native conifer woods and juniper scrub	7	10
Neutral damp grassland	55	60
Neutral pastures and meadows	105	161
Nutrient-poor lakes and ponds	15	7
Nutrient-rich lakes and ponds	20	41
Raised bog	0	3
Rivers and streams	28	37
Wet heath	41	27
Wet woodland	27	32

Table 5. NPMS fine scale habitat links to priority habitats and critical load values for nitrogen. The EUNIS classes for which the original critical loads were derived are given in the table below. The EUNIS classes that they have implicitly been extended to (through matching to NPMS fine habitats) can be derived from comparison with Table 2.

NPMS Fine Habitat	**Priority habitat**	**Critical load exceedence value (kg N ha^{-1} $year^{-1}$) and original EUNIS class (in parentheses). (From Hall *et al.*, 2011; Caporn *et al.*, 2016)**
Arable field margins	Arable Field Margins	Not covered
Raised Bog	Lowland raised bog	5-10 (D1)
Blanket Bog	Blanket bog	5-10 (D1)
Wet heath	Lowland Heathland	10-20 (F4.11)
Hedgerows of native species	Hedgerows	Not covered
Wet woodland	Wet Woodland	10-20 (G1)
Dry deciduous woodland	Lowland Beech and Yew Woodland	10-20 (G1.6)
	Lowland Mixed Deciduous Woodland	10-15, 15-20 (G1.8; G1; G4)
	Upland Oakwood	10-15, 15-20 (G1.8; G1)
	Upland Birchwoods	10-20 (G1)
	Upland Mixed Ashwoods	15-20 (G1.A)
Coastal saltmarsh	Coastal saltmarsh	20-30 (A2.53; A2.54; A2.55)
Coastal sand dunes	Coastal sand dunes	8-15 (B1.4), 10-20 (B1.3)
Machair	Machair	Not covered
Coastal vegetated shingle	Coastal vegetated shingle	Not covered
Maritime cliff and slopes	Maritime cliff and slopes	Not covered
Nutrient poor lakes and ponds	Oligotrophic and dystrophic lakes; Mesotrophic lakes; Ponds	3-10 (C1.1)
Nutrient rich lakes and ponds	Eutrophic standing waters; Ponds	Not covered
	Reedbeds	Not covered
Rivers and Streams	Rivers	Not covered
Dry heathland	Lowland Heathland	10-20 (F4.2)
Montane dry heathland	Mountain Heaths and Willow Scrub	5-15 (F2)
	Upland Heathland	10-20 (F4.11)
Dry calcareous grassland	Lowland Calcareous Grassland	15-25 (E1.26)
	Calaminarian Grasslands	Not covered
Dry acid grassland	Lowland Dry Acid Grassland	8-15, 10-15 (E1.95; E1.7))
Neutral pastures and meadows	Upland Hay Meadows	10-20 (E2.3)
	Lowland Meadows	20-30 (E2.2)
Neutral damp grassland	Coastal and Floodplain Grazing Marsh	Not covered
Acid Fens, mires and springs	Lowland Fens; Upland Flushes, Fens and Swamps	10-15, 15-30 (D2; D4.1)
	Purple Moor Grass and Rush Pastures	15-25 (E3.51)
Base-rich fens, mires and springs	Lowland Fens; Upland Flushes, Fens and Swamps	10-15, 15-25, 15-30 (D2; D4.1; D4.2)
Montane calcareous grassland	Upland Calcareous Grassland	5-10 (E4.4)
Montane acid grassland	Mountain Heaths and Willow Scrub	5-10 (E4.3)
Conifer woods and juniper scrub	Native Pine Woodlands	5-15 (G3; G3.4)
Montane rocks and scree	Mountain Heaths and Willow Scrub	5-15 (E4.2; F2)
	Inland Rock Outcrop and Scree Habitats	Not covered
Inland rocks and scree	Inland Rock Outcrop and Scree Habitats	Not covered
	Limestone Pavements	Not covered

Table 6. Linking NPMS Indicator species to evidence of links with nitrogen deposition.
See WP1 Appendix 1 below.

Task 3. Assessing the potential for recorder bias in the NPMS dataset

All surveys have the potential for issues with species' detectability or other biases, despite the existence of protocols and guidance (Morrison, 2016); however, this may be a particular issue with volunteer-based surveys (Tulloch *et al.*, 2013; Pescott *et al.*, 2015). The existence of a contemporaneous survey to the NPMS, covering an overlapping set of habitats in Wales, the Glastir Monitoring and Evaluation Program (GMEP; https://gmep.wales), allows for a test of the potential for this phenomenon to have affected NPMS data. The GMEP survey follows the methodology for unbiased plot locations used in the UK Countryside Survey (CS; Carey *et al.*, 2008), with recording performed by contracted surveyors. The main reason for using GMEP data instead of the CS is the temporal match with NPMS surveys – the use of the CS from 2007 would introduce an additional element of uncertainty given the ten years since the actual surveys for the CS were conducted. GMEP data for 2014 and 2016 (the two years of available survey data) were compared to Welsh NPMS data (for 2015 and 2016) on the following basis: plots were matched on general broad habitats (as opposed to NPMS broad habitat specifically; *e.g.* see Norton *et al.*, 2012) and were limited to comparable plot sizes. It was also established that the NPMS and GMEP 1 km square samples to be compared exhibited similar distributions of the land cover weightings used to randomly weight the selection of squares in the NPMS (Pescott *et al.*, 2014); this means that bias attributable to within-grid square land covers could be excluded as a significant process influencing the results. Subsequently, in order to establish a common benchmark against which to compare plot types, the plots that were subject to complete sampling (*i.e.* GMEP and NPMS Inventory plots) were both filtered according to the species on the NPMS Indicator list.

After performing this filtering step, two patterns were apparent: NPMS Indicator plots were significantly less species rich than their comparator-filtered GMEP plots, whilst filtered NPMS Inventory plots tended to be marginally more species rich than their GMEP comparators (Fig. 1); note, however, that this graphical analysis does not constitute a formal statistical model, but rather an indication of potential significance. If one assumes that GMEP data are largely without bias due to the fact that the plot placement methodology within squares follows a standardised protocol designed to minimise bias (Maskell *et al.*, 2008; note, however, that error of various types may still be present: plots in some GMEP squares still encountered access issues, for example), then we can conclude that NPMS Indicator plots may be under-recorded, whilst NPMS Inventory plots may suffer from some plot-level self-selection bias towards richer habitat patches (despite the NPMS protocol). Another consideration relevant to the NPMS Indicator plots is that, even if the distribution of NPMS land-cover weights across 1 km squares is similar to that across GMEP squares, the explicit NPMS guidance to place plots in patches of semi-natural habitat, along with the provision of maps to surveyors detailing the likely locations of such habitats in a square, may result in more species-rich plots, even if the within-square plot selection protocol (intended to minimise bias) is followed. This is in keeping with the design and aims of the scheme.

The NPMS was designed to report on semi-natural habitats of types under-represented by other national surveys, such as the CS (Carey *et al.*, 2008). The fact, then, that fully-surveyed plots may be richer than plots surveyed under a completely random framework is not unexpected, and may simply be the result of surveyors following the provided guidance. However, without a review of surveyor plot placement, we cannot rule out a contribution from bias (and even if the majority of surveyors could be shown to have followed the NPMS plot placement guidance, plots in such locations could still have been chosen preferentially over suggested alternative locations). Finally, in the specific context of air pollution impacts, richer initial plots may make it easier to detect negative effects such as eutrophication; the desire to have plots in lower quality stands of semi-natural vegetation, in order to detect improvements as well as declines (Pescott *et al.*, 2015), may be less important in this context.

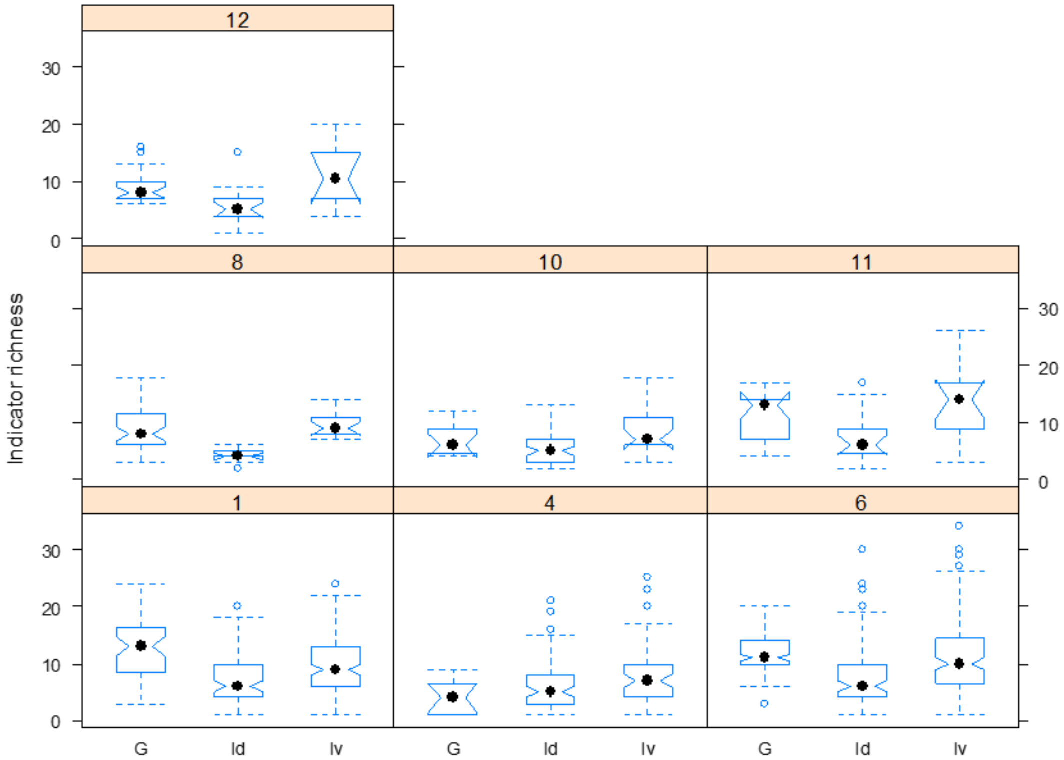

Figure 1. GMEP 2014/'16 quadrats compared with Welsh NPMS plots 2015/'16. Numbers above figure panels relate to broad habitats, see Table 7 below for descriptions and code numbers. Each box and whisker summarises the distribution of the values of each response variable in each of four datasets coded as follows: G = GMEP quadrat data from Wales; Id = NPMS quadrats in which only selected indicator species were recorded; Iv = NPMS quadrats in which all vascular plant species were recorded. Note, however, that the variables shown, irrespective of their survey origins, are based solely on presence of indicator species listed by NPMS (*i.e.* they were filtered according to the NPMS Indicator lists). Such a constraint ensures that all records in all quadrats were drawn from the same list of plant species, although the effort applied in searching for these species may still have differed between quadrats.

Table 7. Broad habitat codes and descriptions.

Broad Habitat code	Broad Habitat
1	Broadleaved woodland
4	Arable
6	Neutral grassland
8	Acid grassland
10	Dwarf Shrub Heath
11	Fen, Marsh & Swamp
12	Bog

Task 4. NPMS squares and their representation of important environmental gradients

Coverage of environmental driver gradients by structured monitoring schemes is of clear importance for subsequent correlative research (Smart *et al.*, 2012), but national variation in volunteer-based survey activity may result in unrepresentative coverage of such gradients, even if available locations for survey are stratified in order to increase geographical spread, as is the case in the NPMS. Analyses of gradient coverage may also serve to highlight where environmental drivers are largely confounded at the scale of investigation, and will therefore also help to manage expectations and/or plan complementary activity at different scales that might be required to strengthen claims of driver-impact associations.

Within the NPMS dataset, we compared all released, volunteer-allocated, and volunteer-surveyed monads (as of February 2017) according to a set of important environmental variables matched at the 10 x 10 km scale (hectads; variables listed in Table 8). Figure 2 illustrates the placement of these sets of squares according to the first two principal components of the ordinated data. The PCA indicates that, although there is little difference in the environmental coverage of allocated and surveyed monads, these are clearly unrepresentative of the complete set of released monads, with squares in the more humid, peat-covered parts of Britain being under-represented in volunteer interest and activity. The second principal component captures gradients of nitrogen oxides (NO_Y) and ammonia/ammonium (NH_X), and there does not appear to be a clear bias relating to the representativeness of allocated, surveyed and all released locations along this second gradient. Despite the fact that the ordination axes imply that NPMS monads cover the pollution gradients included in the analysis, it is, however, very likely that sensitive habitats typical of upland Britain (e.g. raised and blanket bog) will remain under-represented for this comparison: axis two appears to be more driven by the presence of points at the lower end of axis one, suggesting that coverage of the pollution gradient is likely to be largely restricted to lowland areas. This will work to restrict inference for sensitive upland habitats. The NPMS is focusing on recruitment in the uplands as a part of its strategic plan, and, elsewhere, the potential for joint analyses of data with future, professional-led, ground surveys is likely to contribute to the filling of these gaps (P. Henrys, CEH Lancaster, pers. comm.)

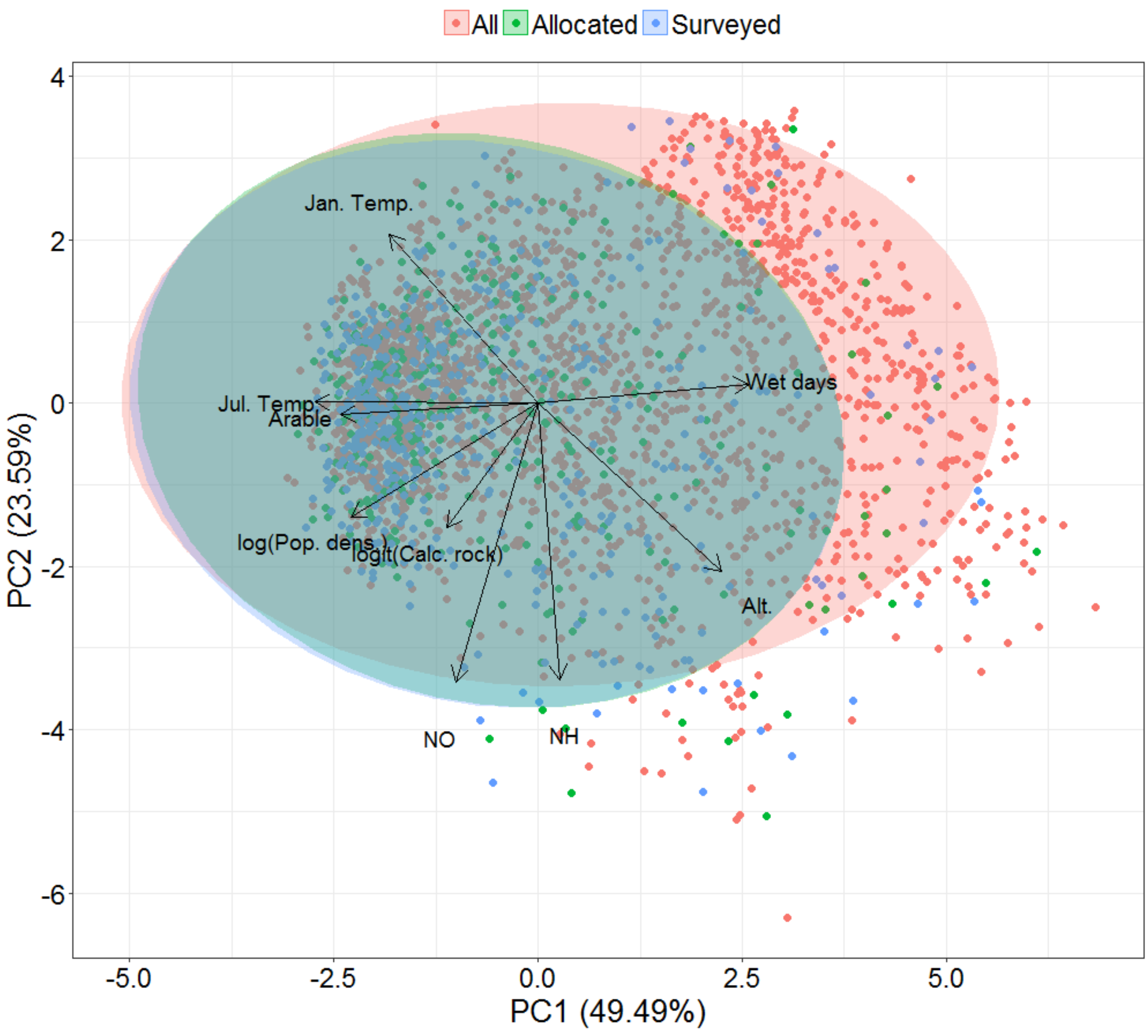

Figure 2. PCA of all GB hectad environmental variables available, grouped by NPMS survey status (to February 2017). Note that hectads with missing values for any of the environmental values are excluded. Calcareous bedrock (as a proportion) was logit transformed prior to entering into the PCA. Population density was log transformed. Ellipses represent 95% confidence intervals. The axis label percentages indicate the percentages of variance explained by a principal component.

Table 8. NPMS square PCA environmental data sources. For maps of these variables (with the exception of NO_y) see Blockeel *et al.* (2014) and Pescott *et al.* (2015).

Environmental variable	Source and notes
Mean altitude	Integrated Hydrological Digital Terrain Model (Morris & Flavin 1990, 1994)
January mean temperature and July mean temperature (both 1981-2010)	Met Office standard dataset
Mean number of wet days per year (1981-2010)	Met Office standard dataset
NO_y and NH_x (keq ha^{-1} yr^{-1}; 1990-1996 mean)	FRAME nitrogen deposition model (Dore *et al.* 2007). 1990-1996 was chosen to represent a recent period of high emissions – the important thing here is to capture relevant spatial variation, rather than absolute values.
Arable land	'Arable and horticulture' cover class, Land Cover Map 2007 (Morton *et al.* 2011)
Peaty soils	'Bog' cover class, Land Cover Map 2007 (Morton *et al.* 2011)
Calcareous rocks	BGS Parent Material Model Version 6 (Great Britain). Based on bedrocks with CaCO3 contents classified as High (e.g. chalk), Variable (high) (e.g. interbedded limestone and mudstone beds) and Moderate (e.g. dolomitic limestone, calcareous mudstone)
Population density	Gridded Population of the World, Version 3 (CIESIN 2005)

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Appendix 1. NPMS species and indicators of nitrogen deposition

Table 6. Linking NPMS Indicator species to evidence of links with nitrogen deposition.

Species	Indicator	Aggregate	ID level	Stevens *et al.* (2004)	Stevens *et al.* (2011)	Dupre *et al.* (2010)	Field *et al.* (2014)	Van Den Berg *et al.* (2011)	Southon *et al.* (2013)
Achillea millefolium	pos		1			positive			
Aethusa cynapium	pos		2						
Agrimonia eupatoria	pos		2						
Agrostis capillaris	neg		2			positive			
Ajuga reptans	pos		1						
Alchemilla alpina	pos		1		small magnitude				
Alisma plantago-aquatica	pos		1						
Allium ursinum	pos		1						
Alnus glutinosa	pos		1						
Alopecurus geniculatus	pos		2						
Alopecurus myosuroides	neg		2						
Ammophila arenaria	pos		2		negative		negative		
Anagallis arvensis	pos		1						
Anagallis tenella	pos		1						
Anemone nemorosa	pos		1						
Angelica sylvestris	pos		1						
Antennaria dioica	pos		1						
Anthemis cotula	pos		2						
Anthoxanthum odoratum	pos/neg		2			inconsistent			
Anthriscus sylvestris	neg		1						
Anthyllis vulneraria	pos		1						
Apium graveolens	pos		1						
Apium nodiflorum	pos		2						
Arabis hirsuta	pos		2						

Arctium minus / nemorosum	pos	1	1						
Arctostaphylos uva-ursi	pos		2		negative				
Armeria maritima	pos		1						
Arum maculatum	pos		1						
Asplenium ceterach (Ceterach officinarum)	pos		1						
Asplenium ruta-muraria	pos		2						
Asplenium scolopendrium (Phyllitis scolopendrium)	pos		1						
Asplenium trichomanes	pos		1						
Aster tripolium	pos		1						
Atriplex portulacoides	pos		2						
Atriplex sp.	pos	1	1						
Avenula pratense (Helictotrichon pratense)	pos		2						
Azolla filiculoides	neg		2						
Bellis perennis	pos		1						
Berula erecta	pos		2						
Beta vulgaris	pos		1						
Betonica officinalis (Stachys officinalis)	pos		1		negative				
Betula pubescens / pendula	pos/neg	1	1						
Bidens tripartita	pos		2						
Blackstonia perfoliata	pos		1						
Blechnum spicant	pos		1						
Brachypodium pinnatum	neg		2						
Briza media	pos		1					negative	
Bromopsis erecta	pos		2		negative				
Bromus hordeaceus	pos		2						
Buddleja davidii	neg		1						
Butomus umbellatus	pos		2						

Calluna vulgaris	pos		1	negative					
Caltha palustris	pos		1						
Calystegia sepium	pos		1						
Campanula glomerata	pos		1		negative				
Campanula latifolia	pos		2						
Campanula rotundifolia	pos		1	negative		Inconsistent		negative	
Campanula trachelium	pos		2						
Capsella bursa-pastoris	pos		1						
Cardamine pratensis	pos		1						
Carduus nutans	pos		2						
Carex arenaria	pos		2						
Carex bigelowii	pos		2						
Carex echinata	pos		2						
Carex flacca	pos		2						
Carex limosa	pos		2		negative				
Carex nigra	pos		2						
Carex otrubae	pos		2						
Carex panicea	pos		2				positive		
Carex paniculata	pos		1						
Carex pendula	pos		1						
Carex pulicaris	pos		2						
Carex remota	pos		2						
Carex rostrata	pos		2						
Carex sylvatica	pos		2						
Carlina vulgaris	pos		2		negative				
Carpobotus edulis	neg		1						
Centaurea scabiosa	pos		1		small magnitude/negative				
Centranthus ruber	neg		1						
Cerastium arvense	pos		2		negative				
Cerastium fontanum	pos		1						

Cerastium glomeratum	pos		2						
Ceratocapnos claviculata	pos		1						
Chaenorhinum minus	pos		2						
Chaerophyllum temulum	pos		2						
Chenopodium album	pos		1						
Chrysosplenium oppositifolium	pos		1						
Circaea lutetiana	pos		1						
Cirsium acaule	pos		1						
Cirsium arvense	neg		1						
Cirsium dissectum	pos		2						
Cirsium heterophyllum	pos		2						
Cirsium palustre	pos		1						
Cirsium vulgare	pos		1						
Clematis vitalba	pos		1						
Clinopodium acinos	pos		2						
Cochlearia sp.	pos	1	1						
Coeloglossum viride (Dactylorhiza viridis)	pos		2						
Colchicum autumnale	pos		1						
Comarum palustre (Potentilla palustris)	pos		1						
Conifer seedlings / saplings	neg	1	1						
Conopodium majus	pos		2						
Cornus sanguinea	pos		2						
Corylus avellana	pos		1						
Crambe maritima	pos		1						
Crassula helmsii	neg		2						
Crataegus monogyna	pos/neg		1						
Crepis paludosa	pos		2						
Crithmum maritimum	pos		1						

Cruciata laevipes	pos		1						
Cryptogramma crispa	pos		1		small magnitude				
Cuscuta epithymum	pos		1						
Cynoglossum officinale	pos		2		negative				
Cynosurus cristatus	pos		1						
Cystopteris fragilis	pos		2						
Cytisus scoparius	pos		2						
Dactylorhiza fuchsii	pos		2						
Dactylorhiza maculata	pos		2						
Dactylorhiza praetermissa	pos		2						
Daphne laureola	pos		2						
Daucus carota	pos		2		negative				
Deschampsia cespitosa	pos/neg		1						
Deschampsia flexuosa	pos/neg		2			inconsistent	positive		positive
Digitalis purpurea	pos		1						
Diphasiastrum alpinum	pos		2						
Dipsacus fullonum	pos		1						
Drosera anglica	pos		2						
Drosera intermedia	pos		2						
Drosera rotundifolia	pos		1						
Eleocharis multicaulis	pos		2						
Elodea canadensis / nutallii	neg	1	1						
Elytrigia atherica	pos		2						
Empetrum nigrum	pos		1						
Epilobium hirsutum	pos		1						
Equisetum arvense	pos		2						
Equisetum fluviatile	pos		2						
Equisetum palustre	pos		2						
Erica cinerea	pos		1						
Erica tetralix	pos		1						
Eriophorum angustifolium	pos		2						

Eriophorum vaginatum	pos		2				positive		
Euonymus europaeus	pos		2						
Euphorbia amygdaloides	pos		1						
Euphorbia exigua	pos		1						
Euphorbia helioscopia	pos		1						
Filago minima	pos		2						
Filipendula ulmaria	pos		1						
Filipendula vulgaris	pos		1						
Fragaria vesca	pos		1						
Fumaria sp.	pos	1	1						
Galium album (Galium mollugo)	pos		1						
Galium aparine	neg		1						
Galium boreale	pos		2						
Galium odoratum	pos		1						
Galium palustre	pos		2						
Galium saxatile	pos		1			inconsistent			
Galium sterneri	pos		2					negative	
Galium verum	pos		1						
Genista anglica	pos		2						
Genista tinctoria	pos		2						
Gentianella amarella	pos		2					negative	
Geranium robertianum	pos		1						
Geranium sanguineum	pos		2						
Geranium sylvaticum	pos		2						
Geum urbanum	pos		1						
Glaucium flavum	pos		1						
Glaux maritima	pos		2						
Glebionis segetum (Chrysanthemum segetum)	pos		2						
Glechoma hederacea	pos		1						

Glyceria maxima	neg		2						
Gymnadenia conopsea	pos		2						
Gymnocarpium dryopteris	pos		2						
Hedera helix	pos/neg		1						
Helianthemum nummularium	pos		1						
Helminthotheca echiodes (Picris echioides)	pos		2						
Heracleum mantegazzianum	neg		2						
Heracleum sphondylium	pos		1						
Hippocrepis comosa	pos		1						
Hippuris vulgaris	pos		1						
Holcus lanatus	pos		2			positive			
Honckenya peploides	pos		2						
Hordeum secalinum	pos		2						
Hottonia palustris	pos		2						
Huperzia selago	pos		2						
Hyacinthoides non-scripta	pos		1						
Hydrocharis morsus-ranae	pos		1						
Hydrocotyle vulgaris	pos		1						
Hypericum elodes	pos		2						
Hypericum tetrapterum	pos		2						
Hypochaeris radicata	pos		2			inconsistent			
Ilex aquifolium	pos		1						
Impatiens glandulifera	neg		1						
Inula conyzae	pos		2						
Iris pseudacorus	pos		1						
Jasione montana	pos		2						
Juncus gerardii	pos		2						
Juncus inflexus / effusus / conglomeratus	pos/neg	1	1						

Juncus squarrosus	pos		2						
Juniperus communis	pos		1						
Kickxia elatine	pos		1						
Kickxia spuria	pos		2						
Knautia arvensis	pos		2		hump-backed				
Lactuca serriola	pos		2						
Lamiastrum galeobdolon	pos		1						
Lamium amplexicaule	pos		2						
Lathyrus linifolius	pos		2						
Lathyrus pratensis	pos		1						
Lemna gibba	pos		2						
Lemna trisulca	pos		1						
Leucanthemum vulgare	pos		1						
Ligusticum scoticum	pos		2						
Limonium sp.	pos	1	1						
Linum catharticum	pos		1					negative	
Littorella uniflora	pos		1						
Lobelia dortmanna	pos		2						
Lonicera periclymenum	pos		1						
Lotus corniculatus	pos		1			inconsistent	negative		
Lotus pedunculatus	pos		1						
Luzula multiflora	pos		2						
Luzula sylvatica	pos		2						
Lycopodium clavatum	pos		2		small magnitude				
Lycopus europaeus	pos		1						
Lysichiton americanus	neg		1						
Lysimachia nemorum	pos		1						
Lysimachia vulgaris	pos		1						
Lythrum salicaria	pos		1						
Matricaria recutita	pos		2						
Medicago lupulina	pos		2						

Melampyrum pratense	pos		2						
Melica uniflora	pos		1						
Mentha aquatica	pos		1						
Menyanthes trifoliata	pos		1						
Mercurialis perennis	pos		1						
Milium effusum	pos		2						
Minuartia verna	pos		2						
Moehringia trinervia	pos		1						
Molinia caerulea	pos		2	negative		inconsistent			positive
Montia fontana	pos		2						
Mycelis muralis	pos		2						
Myrica gale	pos		2						
Myriophyllum aquaticum	neg		2						
Nardus stricta	pos		2			inconsistent	positive		
Narthecium ossifragum	pos		1						
Nasturtium officinale (Rorippa nasturtium-aquaticum)	pos		2						
Neottia cordata (Listera cordata)	pos		2						
Neottia ovata (Listera ovata)	pos		2						
Nuphar lutea	pos		1						
Nymphaea alba	pos		1						
Odontites vernus	pos		2						
Ophioglossum vulgatum	pos		1						
Orchis mascula	pos		1						
Oreopteris limbosperma	pos		2						
Origanum vulgare	pos		1						
Ornithopus perpusillus	pos		2		negative				
Osmunda regalis	pos		1						
Oxalis acetosella	pos		1						

Oxyria digyna	pos		1						
Pastinaca sativa	pos		2		small magnitude				
Pedicularis palustris	pos		2						
Pedicularis sylvatica	pos		2						
Persicaria amphibia	pos		1						
Persicaria bistorta	pos		1						
Persicaria lapathifolia	pos		2						
Persicaria vivipara	pos		2		small magnitude/hump-backed/negative				
Petasites hybridus	pos		2						
Phalaris arundinacea	pos		2						
Pilosella officinarum	pos		1						
Pimpinella major	pos		2						
Pinguicula lusitanica	pos		2						
Pinguicula vulgaris	pos		1						
Plantago coronopus	pos		1						
Plantago lanceolata	pos		1	negative		inconsistent			
Plantago maritima	pos		1						
Plantago media	pos		1						
Platanthera bifolia	pos		2		negative				
Polygala serpyllifolia	pos	1	1			negative			
Polygala vulgaris	pos	1	1			inconsistent			
Potamogeton crispus	pos		1						
Potamogeton perfoliatus	pos		2						
Potamogeton polygonifolius	pos		1						
Potentilla anserina	pos		1						
Potentilla erecta	pos		1			inconsistent			
Potentilla reptans	pos		1						
Potentilla sterilis	pos		1						

Poterium sanguisorba (Sanguisorba minor)	pos		1		inconsistent				
Primula veris	pos		1						
Primula vulgaris	pos		1						
Prunella vulgaris	pos		1						
Prunus spinosa	pos/neg		1						
Pteridium aquilinum	neg		1						
Puccinellia maritima	pos		2						
Ranunculus acris	pos		1						
Ranunculus bulbosus	pos		1						
Ranunculus repens	pos/neg		1						
Ranunculus sceleratus	pos		1						
Ranunculus sp. (water-crowfoots)	pos	1	1						
Reseda lutea	pos		1						
Reseda luteola	pos		1						
Rhamnus cathartica	pos		2						
Rhinanthus minor	pos		1						
Rhododendron ponticum	neg		1						
Rhynchospora alba	pos		2						
Rorippa palustris	pos		2						
Rosa rugosa	neg		1						
Rubus chamaemorus	pos		2		small magnitude				
Rubus fruticosus	neg		1						
Rubus saxatilis	pos		2		hump-backed				
Rumex acetosa	pos		1			positive			
Rumex acetosella	pos		1			inconsistent			
Rumex crispus	pos		1						
Rumex crispus / obtusifolius	neg	1	1						
Rumex hydrolapathum	pos		2						

Ruscus aculeatus	pos		2						
Sagina apetala	pos		2						
Sagina nodosa	pos		2						
Sagittaria sagittifolia	pos		1						
Salix repens	pos		1						
Salvia verbenaca	pos		2						
Sanguisorba officinalis	pos		1						
Sanicula europaea	pos		1						
Saxifraga aizoides	pos		2						
Saxifraga granulata	pos		1						
Saxifraga hypnoides	pos		2						
Saxifraga oppositifolia	pos		1						
Saxifraga stellaris	pos		2						
Saxifraga tridactylites	pos		1						
Scabiosa columbaria	pos		2						
Schoenoplectus lacustris (Scirpus lacustris)	pos		2						
Schoenus nigricans	pos		2						
Scirpus sylvaticus	pos		2						
Scutellaria galericulata	pos		2						
Scutellaria minor	pos		1						
Sedum acre	pos		2						
Sedum anglicum	pos		2						
Sedum rosea	pos		1						
Senecio aquaticus	pos		2						
Senecio erucifolius	pos		2						
Senecio jacobaea	pos/neg		1						
Serratula tinctoria	pos		2						
Sherardia arvensis	pos		1						
Silaum silaus	pos		2						
Silene dioica	pos		1						

Silene flos-cuculi (Lychnis flos-cuculi)	pos		1						
Silene latifolia	pos		1						
Silene uniflora	pos		1						
Solanum dulcamara	pos		1						
Solidago virgaurea	pos		2			inconsistent			
Sonchus arvensis	pos		2						
Sonchus asper	pos		2						
Sonchus oleraceus	pos		2						
Sorbus aucuparia	pos		1						
Sparganium erectum	pos		2						
Spartina anglica	neg		2						
Spergularia marina	pos		2						
Spergularia media	pos		2						
Spergularia rubra	pos		2						
Spirodella polyrhiza	pos		2						
Stachys palustris	pos		2						
Stellaria graminea	pos		1						
Stellaria holostea	pos		1						
Stellaria media	neg		1						
Suaeda maritima	pos		2						
Succisa pratensis	pos		1			negative			
Symphoricarpos albus	neg		1						
Symphytum officinale	pos		2						
Tamus communis	pos		1						
Teucrium scorodonia	pos		1						
Thalictrum flavum	pos		2						
Thalictrum minus	pos		2						
Thymus polytrichus / pulegioides	pos	1	1						
Torilis japonica	pos		2						

Trichophorum germanicum (Trichophorum cespitosum)	pos		2						
Trientalis europaea	pos		2		small magnitude/hump-backed				
Trifolium campestre	pos		2						
Trifolium dubium	pos		2						
Trifolium fragiferum	pos		2						
Trifolium pratense	pos		1						
Trifolium repens	pos		1						
Trifolium striatum	pos		2						
Triglochin maritimum	pos		2						
Triglochin palustre	pos		2						
Tripleurospermum inodorum	pos		2						
Trollius europaeus	pos		1						
Typha latifolia	pos		1						
Ulex europaeus	neg		1						
Ulex gallii / minor	pos	1	1						
Urtica dioica	pos/neg		1						
Vaccinium myrtillus	pos		1						
Vaccinium oxycoccos	pos		1						
Vaccinium vitis-idaea	pos		1		negative				
Valeriana dioica	pos		1						
Valeriana officinalis	pos		1						
Valerianella locusta	pos		2						
Veronica arvensis	pos		1						
Veronica beccabunga	pos		2						
Veronica montana	pos		2						
Veronica officinalis	pos		1			inconsistent			
Vicia cracca	pos		1						

Vicia hirsuta	pos		1						
Viola arvensis	pos		1						
Viola hirta	pos		2						
Viola lutea	pos		2						
Viola palustris	pos		1						
Viola riviniana / reichanbachiana	pos	1	1						

Work Package 2

Task 1: Metrics of pollution impacts and the NPMS

Introduction

The body of work on detecting the impacts of pollutants on biodiversity is extensive (Rowe et al., 2017). Where nitrogen (N) deposition is concerned, several recent projects have evaluated the types of metrics that are likely to provide the most information on change in species, communities, and other aspects of habitat structure and function (Emmett et al., 2011; Rowe et al., 2017; Stevens et al., 2011c). Given that work on N is the most developed among the various air pollutants, we briefly summarise and review the conclusions of this body of work below. This is followed by a short consideration of how these findings may relate to other pollutants, with a particular focus on ozone. The key suggested metrics contributable by the National Plant Monitoring Scheme (NPMS) are considered throughout.

Stevens et al. (2011c) examined species and habitat structural trends for acidic and calcareous grassland, heathland and bogs across eight large datasets. These included datasets that were largely the result of volunteer effort (typically larger-scale[1] 'Atlas'-type data), as well as several datasets from professional surveys (smaller-scale quadrat datasets). The larger-scale analyses of Stevens et al. (2011c) were also presented in Henrys et al. (2011; for vascular plants) and Stevens et al. (2012; for lichens). The significant relationships detected by Stevens et al. (2011c) for NPMS indicator species were extracted from this report and are presented within Appendix 1 of Work Package 1 (WP1; Pescott et al., In press.)

Emmett et al. (2011) further considered the results of Stevens et al. (2011c), placing them in the context of UK biodiversity conservation objectives, and making specific recommendations for the improvement of "broad scale vegetation surveillance schemes". These recommendations are reproduced in Table 1 below; information relating to the status of the recommendation in relation to the NPMS has been added for this report.

[1] "Small" and "large" scale are used here in the loose sense of most ecologists, rather than the strict sense of geographers. That is, small-scale is used to refer to small spatial units (e.g. on the order of metres), whereas large-scale is used to refer to much larger units (e.g. on the order of kilometres).

Table 1. The recommendations of Emmett et al. (2011) for changes to field-based surveillance schemes for improving the detection and attribution of N pollution effects at different scales. U = unsuitable; E = essential; D = desirable. Columns one to four are reproduced verbatim from Emmett et al. (2011).

Recommendation	**Broad-scale hectad / tetrad**	**Broad-scale quadrat**	**Site-specific**	**Currently used in NPMS? / (Possible future addition?)**
Use a defined area for recording, preferably 2x2m.	U	E	E	**Yes** (but a slightly larger unit)
Record quadrat locations so they can be re-found.	U	E	E	**Yes**
Record presence of taxa known to be N-sensitive.	D	D	D	**In part** (see WP 1, Appendix 1)
Identify all taxa to species level, including bryophytes and lichens.	D	D	D	No (Vascular plants only, and coverage dependent on participation level)
Estimate cover as well as presence.	U	D	D	**Yes**
Estimate grass:forb[2] cover ratio in the field.	U	D	E	No (Possible)
Record the Broad Habitat within which each measurement is taken.	E	E	E	**Yes** (See WP1 for more information on NPMS habitat correspondences)
Record the height of the vegetation (including tree canopy) e.g. by visual estimation or using a laser range-finder.	D	D	D	**In part**
Record the standing biomass of the vegetation e.g. by harvesting a defined area.	U	U	D	No (No)
Record the potential production of the vegetation e.g. by installing exclosures and measuring peak biomass.	U	U	D	No (No)
Sample soil and analyse for simple indicators of acidification and eutrophication (see below). Sampling the 0-15 cm depth layer allows comparison with the large Countryside Survey dataset.	U	D/E	D/E	No (Possible)
Measure soil pH.	U	E	E	No (Possible)
Maintain accessible databases, including metadata on units and methods, preferably on the National Biodiversity Network (http://www.nbn.org.uk/).	U	D	D	**Yes** (Data archived on NBN and EIDC)
Measure soil total N% and C%.	U	D	D	No (Possible)
Measure soil available N.	U	D	D	No (Possible)
Measure soil organic horizon depth.	U	D	D	No (Possible)
Measure N content in standard plant tissue.	U	D	D	No (Possible)
Record the location of soil sampling in relation to floristic recording.	U	E	E	No (Possible)
Install suction lysimeters to sample soil solution and analyses for NH_4, NO_3 and if possible dissolved organic N.	U	U	D	No (No)

Rowe et al. (2017) also reviewed a large set of ecological measures of N deposition (Ndep). Their most relevant conclusions for the current task are reproduced in Table 2 below. The various conclusions and recommendations of these reports are further considered below at the two different scales mentioned above: the plot scale, and the larger, 'Atlas'-type scales of 1 x 1, 2 x 2 and 10 x 10 km. Other relevant studies are also discussed as appropriate.

[2] See footnote in Table 3 below.

Table 2. Recommended metrics for monitoring the ecological impacts of Ndep. The first four columns are transcribed verbatim from Rowe et al. (2017), except for where square-bracketed clarifications have been inserted.

Metric	Appropriate for	Recommended calculation method	Evaluation	Considerations for NPMS
Mean Ellenberg N	Habitats where relationship with deposition has been demonstrated. All deposition rates.	Record plant species present, calculate mean Ellenberg N, and compare with typical values for the habitat e.g. using relationships from Stevens et al. (2011c)	**Pros**: well-related to theoretical and observed effects of N on species-assemblages; can be modelled and also easily measured. **Cons**: affected by other factors other than N; [precise ecological] meaning not immediately present.	Indicator (and Wildflower) plots only have a subset of species present. This may affect the sensitivity of the metric's response to gradients of, or temporal change in, N deposition.
Species richness	Grasslands, potentially other habitats such as mires. All deposition rates.	Record plant and lichen species present, calculate species richness, and compare with typical values for the habitat e.g. using relationships from Maskell et al. (2010)	**Pros**: readily understood. **Cons**: affected by other factors other than N; not applicable to all habitats.	As above.
Habitat Suitability Index (HSI)	All habitats. All deposition rates.	Mean simulated habitat suitability for 'species of interest' (Posch et al., 2014)	**Pros**: potentially better-related to 'favourable conservation status' [(JNCC, 2004)] than species richness. **Cons**: needs careful and transparent definition.	As above. The HSI is based on (empirically-based) models of species' responses to key gradients. However, the placement of plots in HSI 'space' will still be limited by the reduced species information available for NPMS Indicator and Wildflower plots.

Synthesis

Data at small-scales

Within quadrat datasets, ordination methods such as constrained correspondence analysis (CCA) have often been used to examine the relationships between pollutant driver variables and individual species' responses (Stevens et al., 2009, 2011b, 2011c). These analyses work on the basis of species' cover values (including zeros) within small plots (Kent, 2012). Other approaches to quadrat datasets include regression-type modelling of plot summary metrics (e.g. species richness, average Ellenberg Nitrogen values; Table 2). For acid grassland, Stevens et al. (2009) investigated the dataset of Stevens et al. (2004) and data from the UK Countryside Survey within this framework. These authors found that the ratio of graminoid:forb covers within a plot exhibited the strongest relationship to Ndep across the two datasets (a result also implied by the geographically broader study of Stevens et al., 2011a), although species richness and forb richness were also highlighted as Ndep indicators with potential value for this habitat. Maskell et al. (2010) expanded this general approach to the 1998 Countryside Survey dataset, identifying a number of relationships between Ndep and potential indicators in heathland, acid grassland, and calcareous grassland plots. In acid grassland and heathland, trait changes indicated that acidification was the main impact of Ndep, whilst a main impact of eutrophication was suggested for calcareous grasslands (note that other, apparently less significant impacts, were also suggested for these habitats by the study of Maskell et al. 2010).

The study of Maskell et al. (2010) is an important piece of evidence for correlative studies of Ndep, as it demonstrated that its effects are detectable in a national-scale observation network not specifically designed to detect the particular impacts of that pollutant class (unlike the focused study of Stevens et al. 2004), whilst controlling for other variables. Partly for this reason, Maskell et al. argued that the impact of Ndep on habitats is likely to be significant across the UK. More recently, Field et al. (2014) assembled a new dataset of field observations representing several habitats (acid grassland, bog, lowland heath, upland heath, and sand dune), and subjected these to a standard regression-based analysis in order to detect relationships between species richness and Ndep. Field et al. reported that, "in every habitat, we found reduced species richness and changed species composition associated with higher nitrogen deposition, with remarkable consistency in species loss across ecosystem types"; graminoid cover was also found to increase. Note that one potential weakness is many of these field-based studies is the failure to include spatial auto-correlation in analyses; additionally, many studies in this area (Field et al., 2014; Stevens et al., 2004, 2010) have also used stepwise variable selection to simplify regression models, which is not a recommended strategy (e.g. Whittingham et al., 2006).[3] Maskell et al. (2010) is one of the few studies not to have used this method, and to have taken into account hierarchical variance structures in analyses – another reason to place confidence in the relationships between Ndep and species richness estimated by this piece of work.

Rowe et al. (2016) took a rather different approach to the issue of Ndep impacts on habitats, surveying conservation managers in order to discover which metrics this group took as indicators of habitat quality. The number of positive indicator species (sensu JNCC, 2004) present was correlated with habitat quality in seven of the nine habitat classes addressed by the study. This result perhaps tells us more about how conservation managers are used to evaluating sites than about the ecology of Ndep impacts, but the priorities of conservation professionals will clearly be of importance in determining management, and knowing how 'quality' is perceived on the ground can be as important as abstract, generalised measures constructed from national datasets. It is perhaps also more likely to have links to the perceptions of users of the countryside, i.e. there is an obvious link to ecosystem "cultural services" (Kareiva et al., 2011).

Payne et al. (2013) used both the dataset of Stevens et al. (2004), and another two heathland datasets (Caporn et al., 2009; Edmondson et al., 2010), to establish multivariate regression-based "transfer functions" (Telford and Birks, 2005). These use estimated species-environment relationships to predict new environments based on new plot observations. Although they reported promising results, it is worth noting that Payne et al. (2013) only tested their transfer functions in the context of Ndep, and that the inclusion of other, confounding, environmental variables could strongly influence results (Telford and Birks, 2005). Typically transfer functions only include variables found to explain significant, non-confounded, variation in initial multivariate analyses (e.g. Davies et al., 2002).

Stevens et al. (2010) also used the dataset of Stevens et al. (2004) to investigate the potential of other indicator metrics at the plot scale in calcifugous (i.e. acidic) grassland. Specifically, mean 'CSR' (competitor-stress tolerator-ruderal) signatures (Grime et al., 2007), mean cover weighted and unweighted Ellenberg N (fertility) and R (acidity/pH) were investigated, as was an index of acid soil indicator species. Only Ellenberg R scores and the index of soil acidity indicated significant relationships with Ndep, although the authors note that a number of other studies have identified

[3] The biostatistician Frank E. Harrell, in his *Regression Modelling Strategies* (p. 67, 2015, 2nd Ed., Springer), notes "if this procedure [stepwise variable selection] had just been proposed as a statistical method, it would most likely be rejected because it violates every principle of statistical estimation and hypothesis testing."

significant links between Ndep and Ellenberg N at the plot scale across different habitats (e.g. Bennie et al., 2006; Pitcairn et al., 2002; Smart et al., 2005).

Emmett et al. (2011) summarised their recommendations for the "analysis of variables that can be derived from field data, to improve detection and attribution of N pollution effects". This is reproduced below (Table 3), with additional comments on the NPMS provided.

Table 3. Emmett et al. (2011) recommendations for the "analysis of variables that can be derived from field data, to improve detection and attribution of N pollution effects", with comments on the possibility of the use of NPMS data for the production of the recommended metrics.

Recommendation (Emmett et al. 2011)	**Cost** (Emmett et al. 2011)	**Comments in relation to NPMS**
Analyse floristic presence/cover data for species richness and Shannon diversity/evenness index	Low if data exist	Straightforward for Inventory (i.e. full) plots. Survey level would need to be included as a covariate to account for the effects of partial surveying at the Indicator and Wildflower levels.
Analyse floristic presence/cover data using species-groups based on:	Low if data exist	See individual metrics below.
• Proven N-sensitivity	Low if data exist	Straightforward for Inventory plots. See WP1, Appendix 1 for a cross-tabulation of key N indicators from the literature alongside NPMS Indicator status.
• Ellenberg N score	Low if data exist	Straightforward for Inventory (i.e. full) plots. Survey level could be included as a covariate to account for the effects of partial surveying at the Indicator and Wildflower levels.
• Ellenberg R score	Low if data exist	Straightforward for Inventory (i.e. full) plots. Survey level could be included as a covariate to account for the effects of partial surveying at the Indicator and Wildflower levels.
• Grass:forb cover ratio[4]	Low if data exist	Not estimated directly by surveyors, but could be calculated for Inventory plots.
• Typical height of present species	Low if data exist	Straightforward for Inventory (i.e. full) plots. Survey level could be included as a covariate to account for the effects of partial surveying at the Indicator and Wildflower levels.
• Typical Specific Leaf Area of present species	Low if data exist	Straightforward for Inventory (i.e. full) plots. Survey level could be included as a covariate to account for the effects of partial surveying at the Indicator and Wildflower levels.
Maintain accessible databases, including metadata on units and methods, preferably on the NBN.	Medium	NPMS data are archived in full on the EIDC and data.gov.uk. Species occurrence data are sent to the NBN at full resolution, except for Northern Ireland (1 x 1 km resolution).

[4] Note that Stevens et al. (2009) investigated graminoid:forb cover ratio, rather than grass:forb. Grass:forb was specifically investigated in Maskell et al. (2010). Graminoid means 'grass-like', and normally covers grasses, sedges and rushes.

Few Ndep studies have investigated the percentage-cover responses of individual species in a univariate context (i.e. outside of the context of plot data), although Stevens et al. (2009) present eight bivariate plots charting the relationship between species percentage cover and Ndep. However, all of these have low R^2 values, and it is not clear that the peculiar distributional properties of percentage cover data were properly accounted for in these analyses (e.g. the moss *Hylocomium splendens* appears to have a strong decreasing relationship with Ndep in Fig. 2b of Stevens et al. (2009), but the R^2 reported is one of the weakest at 0.02). These species-specific relationships could be profitably reinvestigated using the zero-inflated Beta distribution, which is better suited to percentage cover data, and can change (ecological) conclusions (Irvine et al., 2016).

Habitat quality metrics

One final area that should be considered for plot data is the construction of indicator metrics based on assessments of stand quality. These are essentially indices that are constructed in order to measure some conception of the intrinsic value of a community. They are closely related to metrics constructed from Ellenberg values or intrinsic plant traits such as Specific Leaf Area, in that they average over, or perform some other type of mathematical operation on, a number of species, each of which has been assigned some indicator value (e.g. of habitat quality or representativeness). Some examples of these follow:

- Rowe et al. (2011) construct a habitat-specific quality index from empirical species-specific niche models originally fitted to the extensive Countryside Survey datasets (Henrys et al., 2015; Smart et al., 2010). The site-specific suitability (based on modelled or measured abiotic variables, namely pH, soil C:N, N and S deposition) for each species is then estimated for the relevant set of habitat positive and negative indicators. In this case the Common Standards Monitoring species are used (JNCC, 2004). The final metric is the average of the positive indicator species relative suitability, minus the same measure for the negative indicators. That is, higher scoring sites will be, in theory, more likely to have positive than negative indicators. Rowe et al. (2009) describe the same method.
- Posch et al. (2014) present a similar approach to Rowe et al. (2011), this being the 'Habitat Suitability Index' (HSI) recommended by Rowe et al. (2017; see Table 2). The HSI is essentially the same as the indicator of Rowe et al. (2011), except for the fact that negative indicators are not included. That is, the HSI is the average relative suitability of the relevant positive indicators for the habitat under consideration. Where the site-specific and maximum suitabilities are the outputs of pre-existing niche models (previously parameterised using extensive national plot datasets as described above).

Ultimately these indicators either require well-parameterised niche models (Rowe et al., 2011, 2015), and the abiotic data with which to make site-specific estimates for sets of species, or the acceptance of a set of broader quality indicators, such as a species' national frequency (at some agreed scale) or standard conservation value, which have also been linked to models of abiotic processes and pollutant deposition (Van Dobben and Wamelink, 2009; Van Dobben et al., 2015). Van Dobben et al. (2015) review the area of "plant species diversity indicators for use in the computation of critical loads and dynamic risk assessments". These authors review a number of basic plant diversity indicators, and cover similar ground to Rowe et al. (2015) and van Dobben and Wamelink (2009).

Although the HSI, and related approaches, are well-developed techniques, they may not be immediately suitable for the NPMS. Plant communities as recorded in the NPMS would have to be back-transformed to the implied abiotic variables, which could then be back-transformed to estimated pollutant loads (the idea here is similar to the transfer functions of Payne et al. 2013). As mentioned above, another (not mutually exclusive) option would be to estimate, or measure, environmental variables at sites for use in existing empirical niche models, with the resulting species' suitabilities potentially then being combined into an HSI (Van Dobben et al., 2015; Table 2). In theory such measures could then be compared to what was actually recorded by surveyors. These options are similar to the integration of national datasets with local evidence suggested by Jones et al. (2016). These are interesting areas to consider for the future of the utility of NPMS data, but they do not appear to be short-term answers to the question of producing cost-efficient air pollution indicators using NPMS data.

Data at large-scales

Stevens et al. (2011c) and Henrys et al. (2011) investigated a number of UK surveillance networks for their potential to detect signals relating species frequencies to Ndep. At a large scale (2 x 2 and 10 x 10 km) two datasets from the Botanical Society for Britain and Ireland (BSBI) were used. One is a national Atlas dataset at 10 x 10 km (Preston et al., 2002), the other the result of a structured, systematic sample of UK tetrads (2 x 2 km) discussed in Braithwaite et al. (2006). (See also Pescott et al., 2015 for general discussion of these datasets).

These studies were restricted to species at intermediate frequencies in the datasets, this is required for the detection of spatial or temporal relationships with covariates at the scale analysed (Henrys et al. 2011). The smaller the scale the greater the number of species that are likely to be analysable. Species-broad habitat associations from PLANTATT (Hill et al., 2004), along with land cover mapping (Land Cover Map 2007) were used in order to restrict analyses of change to species populations' within particular habitats. That is, species presence data was only analysed within that set of grid squares containing land cover types associated with that species' characteristic broad habitat (Henrys et al., 2011; Stevens et al., 2011c).

Overall, mean Ellenberg N showed "a significant increase with increasing N deposition in almost all habitats across both surveys [Preston et al. 2002; Braithwaite et al. 2006] indicating increased fertility" (Henrys et al., 2011). In addition, "many individual species showed strong relationships with N deposition, and clear negative trends in species prevalence to increasing nitrogen were found in all habitats". (Such species were extracted from Stevens et al. 2011c under Work Package 1 of the current report).

The only other significant large-scale investigation into the impacts of air pollutants on plant species or vegetation of which we are currently aware is the work of McClean et al. (2011). These authors also utilised the 10 x 10 km Atlas data presented in Preston et al. (2002), but constructed lists of species lost from particular grid cells post-1986. The mean Ellenberg N of such 'lost' species (per grid cell) was compared with the mean post-1986 signal. This difference was then analysed in a regression framework, with estimates of both reduced and oxidised N included. Among other variables, reduced N was found to be a driver of the post-1986 change in Ellenberg N at the 10 km grid cell level.

Ozone

The only study that we have located to date that included some measure of ozone in its analyses of extensive vegetation data is that of Payne et al. (2011). Payne et al. (2011) also analysed the dataset of Stevens et al. (2004)[5] in an ordination framework (although using Redundancy Analysis, RDA,

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[5] Note the number of times this dataset has been analysed. Multiple re-analyses of the same dataset must inevitably generate false associations; moreover, citing the multiple papers that have arisen from the analysis of this dataset gives an impression of an evidence-base that is larger and, more independent, than is actually

rather than the Constrained Correspondence Analysis favoured by all of the other studies investigating this dataset).[6] The ozone data included in the models was tropospheric ozone extracted from APIS (the particular time frame is not specified). Payne et al. (2011) found that their ozone data explained around 4.5% of the variation in their vegetation dataset when it was the only variable in the model; in the full model (i.e. conditional on the influence of other covariates), ozone was the strongest variable, but explaining only 3.5% of the variation. An indicator species analysis of high versus low ozone sites revealed six plant species (bryophyte and vascular plants) that were associated with these conditions. Payne et al. (2011) reviewed experimental evidence of these species, providing some corroboration of their results.

More recently, Mill et al. (2016) have provided a review of the "process-oriented perspective on the combined effects of ozone (O_3), climate change and/or nitrogen (N) on vegetation", however, the process-focus of this paper is not particularly relevant for our current review. Hayes et al. (2012) report experimental impacts of ozone on flowering, which may be more relevant for developments to the NPMS monitoring strategy.

Recommendations

Here we make some brief recommendations for the development and use of air pollution impact metrics derivable from NPMS data. Specifically, the NPMS will consider the inclusion and derivation of the following measures:

- Mean Ellenberg N (and R) depending on habitat, have broad support at multiple scales, although, as with any correlative analysis, responses can be complicated by other covariates (e.g. contemporary/historic management).
- Species/indicator richness – again, varied evidence depending on habitat and study, but there is considerable evidence that this measure does respond to indices of Ndep pollution.
- NPMS Partnership to consider the inclusion of grass:forb ratio within its field protocol.
- NPMS Partnership to consider the inclusion of flower counts, or related measure, on its field protocol.

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the case. However, we are not suggesting that all re-use is bad: combining datasets can be a powerful route to inference.

[6] Note that these two ordination techniques imply different models of individual species' responses to environmental gradients, and that both cannot be optimal for any given dataset.

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Task 2: Power and sensitivity of NPMS data for the attribution of air pollution effects

Summary

- The models analysed here give a standard power analysis result, in that a correctly diagnosed decline or increase (here in species richness) becomes more likely with an increasing number of sites. Declines or increases in richness are also more easily detected from a higher starting richness, and when the decline or increase is of a larger magnitude. Declines are slightly easier to detect than increases, all else being equal. Comparisons of the number of NPMS plots for the 2015-16 seasons suggest that at the current time only the larger increases or declines in species richness are likely to be reliably detected.
- Considerations of power for other metrics, such as Ellenberg N or R, suggested that changes across time within plots are likely to be small, and that the numbers of plots required to detect such changes are likely to be outside the scope of the NPMS. However, Ndep impacts on these metrics have been detected in other studies using space-for-time substitution techniques (Maskell et al., 2010; Stevens et al., 2010), and a framework for analysing NPMS data in this way has been developed (WP2 Task 4). Average Ellenberg plot metrics calculated for NPMS Indicator-level surveys may be relatively robust across plots; the filtering created by recording a subset of species means that Ellenberg plot metrics are more likely to exhibit 'stepped', rather than gradual, changes in their values.
- In relation to nationwide ecological gradients, the species filtering imposed by recording at NPMS Indicator or Wildflower levels did cause changes in the position of plots along these gradients, but there was no evidence that this filtering left significant sections of these gradients under-represented. Indeed, apart from the impacts of NPMS Wildflower filtering on the ability of plots to represent succession and disturbance, both NPMS levels had remarkably little impact on recovery of the range of each ecological gradient. The reduced species recording of the lower levels of the NPMS may not seriously reduce the ability of analysts to detect change relating to these major gradients.

Section 1 – Power

What intensity of monitoring activity is sufficient to detect a statistically significant change in abundance, or some other measure, if one has actually occurred? This is the question addressed by power analyses (Jones, 2013; Pescott et al., 2016). Conducting an appropriate power analysis for a monitoring scheme involves deciding on a set of relevant scenarios to investigate, covering a range thought plausible once the proposed scheme is established. Important variables affecting the quality of inference include those that represent the underlying structure of the data, e.g. the number of years of monitoring, the number of sites monitored or the arrangement of repeated site visits in time and space (Urquhart, 2012), and those that represent the hypothetical effect that the monitoring is intended to capture, e.g. changes in species' abundances or distributions within a specified time frame. This could be, for example, a constant change of a fixed number of organisms or area of cover per year, or a proportional change in such a measure.

Here we investigate power in the context of two metrics that can be derived from National Plant Monitoring Scheme (NPMS) data (www.npms.org.uk), and which could potentially contribute to the detection of air pollution impacts on biodiversity (see WP2 Task 1 for more information).

Species and indicator richness

Task 1 of this work package concluded that species richness and Ellenberg N (i.e. fertility) metrics are likely to be the most tractable metrics of Ndep for spatially extensive, correlative studies, although Ellenberg R (i.e. acidity) may also be useful in certain habitats. Other metrics that are not explicitly collected by the NPMS at the current time, but which could be in the future, include the grass (or graminoid) to forb cover ratio; note, however, that the introduction of new measures to the NPMS will be dependent on volunteer engagement, as well as scientific need (Walker et al., 2015). As also indicated in WP2 Task 1 (Table 3), the power of certain metrics to detect change will be influenced by the amount of data available, which is in turn largely determined by the number of plots surveyed at the Inventory[7] level within a particular NPMS habitat. Any reduction in information on the presence of a certain species within a plot, as occurs if a survey is conducted at the Wildflower or Indicator levels, will reduce the information content of any summary metric. This can be partly adjusted for by introducing a survey-level covariate into data analyses (e.g. see WP2, Task 4); that is to say, the level at which a survey is conducted has an estimable effect on the metric of interest (e.g. species richness), which can then be accounted for when estimating the effects of other variables, such as that of a pollutant. Assuming that NPMS Indicator and Wildflower plots retain some ability to detect important ecological change (see Section 2 below), then amalgamating these reduced plot data with the full NPMS Inventory data is likely to increase the power to detect ecological changes of interest on average, even if the reduced species complement of these plots reduces their sensitivity to change.

In order to investigate some of these questions, NPMS Inventory (i.e. full plot) and Indicator richness trends were simulated in a generalised linear model framework (Gelman and Hill, 2007; Johnson et al., 2015; Miller and Mitchell, 2014). Here we focus mainly on acid grassland, given that it is one of the habitats for which the most evidence of Ndep impacts has been reported; we also consider calcareous grassland. In order to parameterise certain aspects of these models, we use a set of pseudo-quadrats based on the UK National Vegetation Classification, as well as actual NPMS data collected to date. The NVC pseudo-quadrats are a randomly generated set of 'fake' quadrats,

[7] Recall that the NPMS has three levels, Wildflower, Indicator and Inventory, where the Inventory is a full census of the vascular plants present within a plot, and Wildflower and Indicator plots are monitored for reduced sets of positive and negative indicator species (see www.npms.org.uk and/or Walker et al. 2015 for more background information).

generated from the synoptic plant community tables from the published NVC handbooks (Rodwell, 1991). The full detail of their generation is given in the online supplementary material of Tipping et al. (2013).

Acid grassland

The full set of the 44,753 NVC pseudo-quadrats was filtered to the NVC communities that are represented by the upland and lowland acid grassland communities in the NPMS (these being U1, U2, U3, U4, U5, U6, U7, U8, U9, U10, U11, U12, U13, U14, U19, U20, SD10, SD11; see Work Package 1). This left 1831 NVC pseudo-quadrats. The median species richness of these quadrats, after eliminating bryophytes and lichens, was 11 (17 before filtering). Filtering to retain only positive NPMS Indicator species of lowland and upland acid grassland resulted in a median species richness of 4 (1762 plots). These values were subsequently used as the mean starting richness values (i.e. the intercepts) in the following power analyses. These values are also very similar to the average empirical richnesses from the NPMS 2015 and 2016 data collected from the dry acid grassland and montane grassland habitats (Table 1).

Table 1. NPMS Average species richnesses by fine-scale habitat and survey level (rounded to the nearest integer).

2015	**Avg. spp. richness**
Dry acid grassland	**11**
Wildflower survey	4
Indicator survey	6
Inventory survey	15
Montane acid grassland	**11**
Wildflower survey	6
Indicator survey	7
Inventory survey	17
2016	**Avg. spp. richness**
Dry acid grassland	**10**
Wildflower survey	5
Indicator survey	7
Inventory survey	20
Montane acid grassland	**13**
Wildflower survey	2
Indicator survey	6
Inventory survey	16

The size of the current NPMS plot resource varies depending on habitat (Pescott et al., In press.); for acid grassland habitats, the variation is within the 0-100 range (note that we only consider plots recorded at the fine-scale habitat level here, the resource could potentially be increased by including those plots recorded at the broad-scale level; Table 2). This, therefore, is a conservative estimate of the NPMS resource.

Table 2. The number of samples of plots for the first two years of the NPMS, for the two acid grassland habitats covered by the scheme. These are subdivided by the level at which they were surveyed (where Inventory level is a full vascular plant survey of a plot).

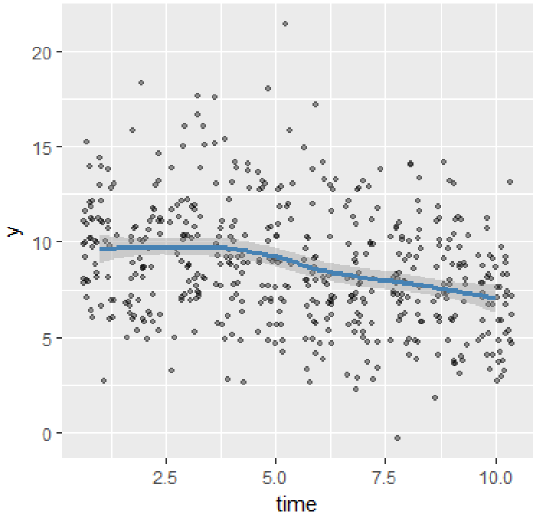
2015	**117**
Dry acid grassland	**76**
Wildflower survey	7
Indicator survey	25
Inventory survey	44
Montane acid grassland	**41**
Wildflower survey	15
Indicator survey	6
Inventory survey	20
2016	**129**
Dry acid grassland	**88**
Wildflower survey	41
Indicator survey	16
Inventory survey	31
Montane acid grassland	**41**
Wildflower survey	6
Indicator survey	4
Inventory survey	31

The power analyses for richness (whether full vascular plant richness or indicator richness) were conducted according to the scenarios detailed in Table 3 below. We simulate scenarios for numbers of sites between 1 and 250, rather than the 0-100 noted above, as the range within which the NPMS data resource currently sits. This was in order to explore the number of sites required for 80% power across as many scenarios as possible. The declines below are specified in terms of an overall linear loss relative to the original starting richness, rather than a year-on-year proportional (i.e. exponential) decline.

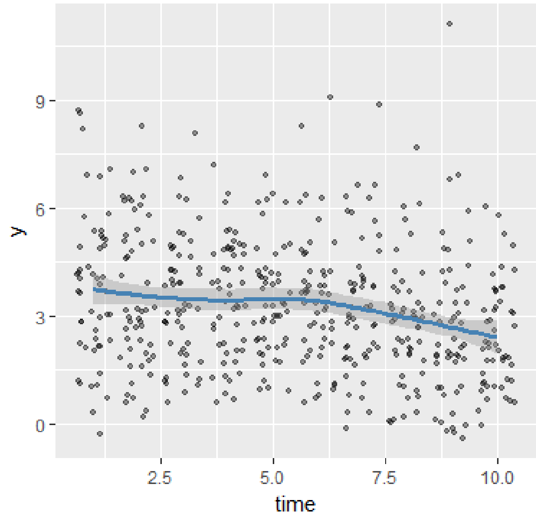
Table 3. Parameter values explored in the acid grassland power analyses.

Parameter	Values explored
Number of sites (i.e. annual plot samples)	1-250
Number of years	10
Starting species richness	4 (Acid grassland Indicator plots) 11 (Acid grassland Inventory plots)
Temporal change (i.e. the overall trend)	33% decline over ten years 10% decline over ten years 10% increase over ten years 33% increase over ten years

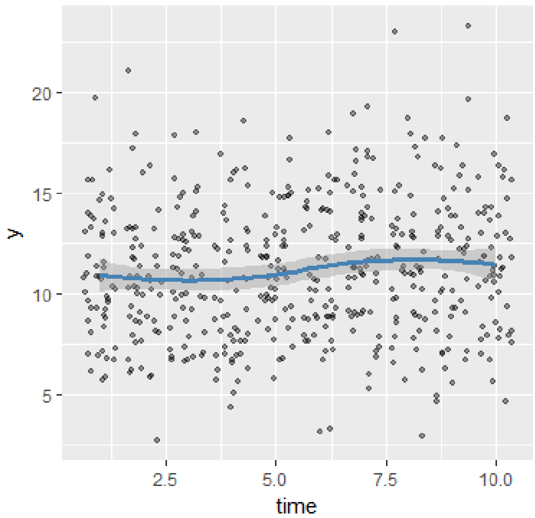
Poisson generalised linear models (GLM) were used for all power analyses. No additional random effects were explored (e.g. additional variance in starting abundances (the intercept), slopes, or overall error); this was because after exploring data simulated using the Poisson distribution, the variance inherent in this distribution appeared realistic in relation to the real world species richnesses expected for these habitats, and to the current NPMS dataset. Incorporating too many sources of variance into simulations using the Poisson distribution can result in unrealistic values being encountered in some individual simulations, leading to counter-intuitive conclusions regarding power (Pescott et al., 2015). Examples of individual simulations conducted using some of these parameter combinations are given below (Figure 1). For each scenario, 500 of these individual simulations were run. Poisson GLM fits were evaluated using the rtrim package[8]; temporal autocorrelation was taken into account when assessing trend significance (serialcor = TRUE in the trim function of rtrim).



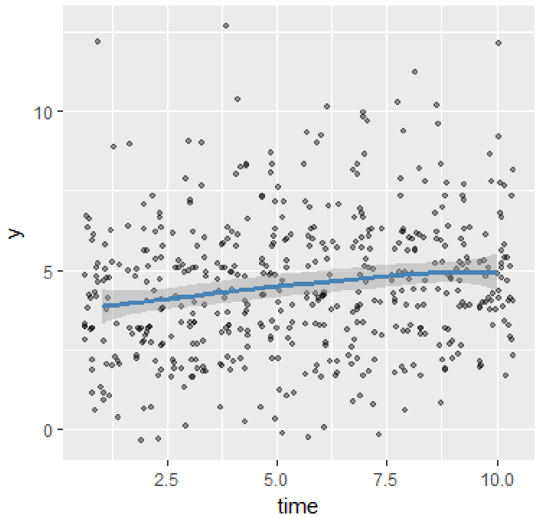
50 sites, 10 years, starting richness 11, decline 33% over 10 years



50 sites, 10 years, starting richness 4, decline 33% over 10 years



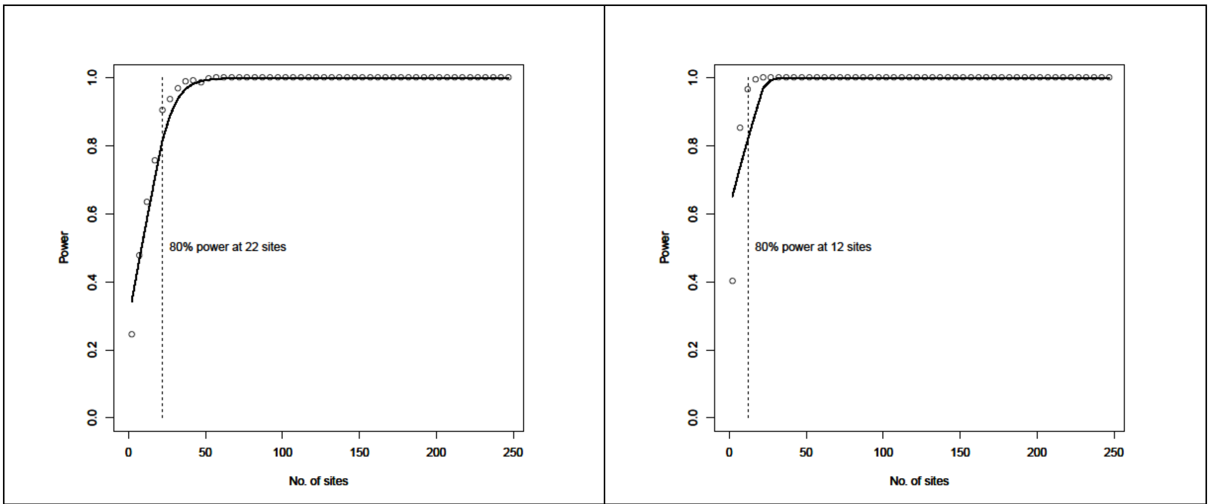
50 sites, 10 years, starting richness 11, increase 10% over 10 years



50 sites, 10 years, starting richness 4, increase 10% over 5 years

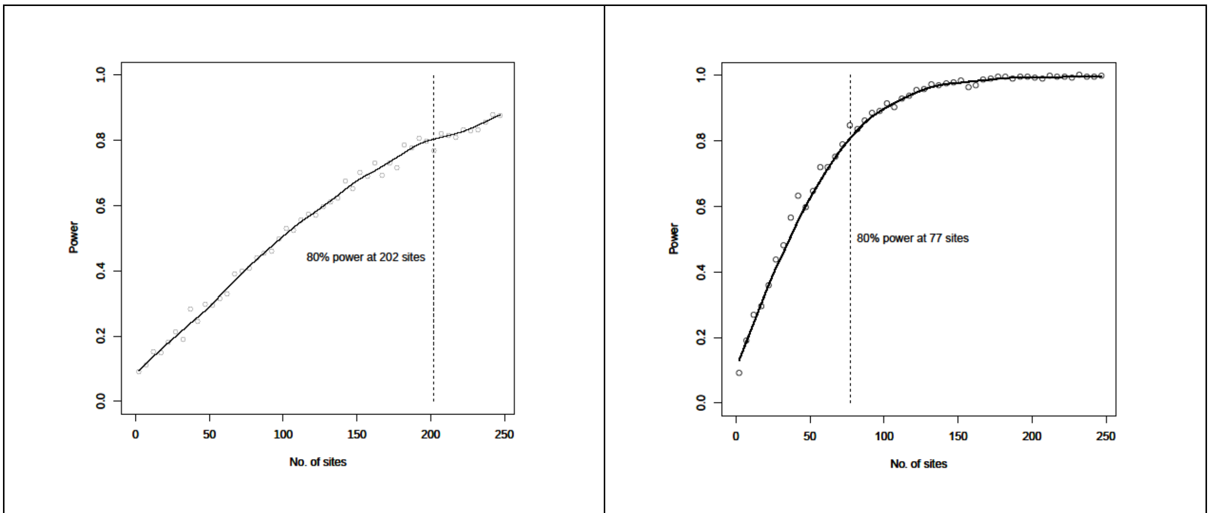
Figure 1. Examples of individual simulations for species richness power analyses. Points ‘jittered’ for clarity (this accounts for some points occasionally being below zero). Note the different scales on the y-axes. Smoothers are local regression (‘loess’) smoothers.

[8] https://cran.r-project.org/package=rtrim



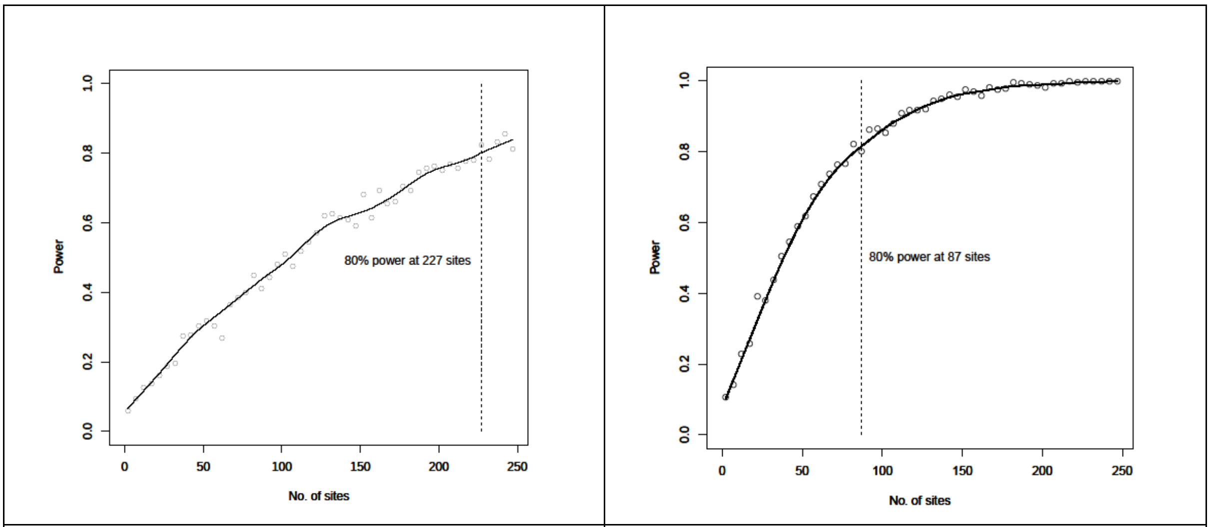
33% decline over 10 years, starting abundance 4	33% decline over 10 years, starting abundance 11

Figure 2. Power curves for acid grassland, indicator (starting abundance 4) and inventory (starting abundance 11) simulated data. Underlying simulations (500) exhibited an average overall decline of 33% in initial species richness over 10 years.



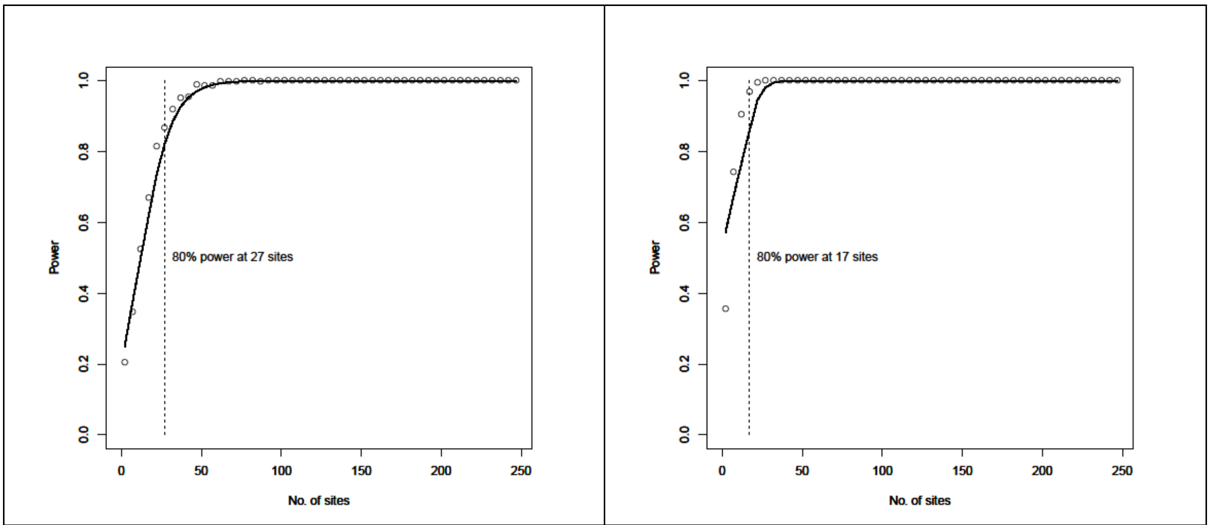
10% decline over 10 years, starting abundance 4	10% decline over 10 years, starting abundance 11

Figure 3. Power curves for acid grassland, indicator (starting abundance 4) and inventory (starting abundance 11) simulated data. Underlying simulations (500) exhibited an average overall decline of 10% in initial species richness over 10 years.



10% increase over 10 years, starting abundance 4	10% increase over 10 years, starting abundance 11

Figure 4. Power curves for acid grassland, indicator (starting abundance 4) and inventory (starting abundance 11) simulated data. Underlying simulations (500) exhibited an average overall increase of 10% in initial species richness over 10 years.



33% increase over 10 years, starting abundance 4	33% increase over 10 years, starting abundance 11

Figure 5. Power curves for acid grassland, indicator (starting abundance 4) and inventory (starting abundance 11) simulated data. Underlying simulations (500) exhibited an average overall increase of 33% in initial species richness over 10 years.

Calcareous grassland

In contrast to acid grassland, calcareous grasslands in Britain are typically relatively species rich (Lake et al., 2014; Rodwell, 1992). Therefore we repeated the above analysis for these communities within the NPMS, in order to bracket the range of species richnesses expected within the scheme. As before, the full set of the NVC pseudo-quadrats is 44,753. Filtering to the NVC communities represented by the NPMS, dry calcareous grassland and montane calcareous grassland (NVC communities CG1-CG14; see Work Package 1), leaves 2455 NVC pseudo-quadrats. The median species richness of these quadrats, after eliminating bryophytes and lichens, was 22 (25 before filtering). Filtering to retain only positive NPMS Indicator species of lowland and upland calcareous grassland resulted in a median species richness of 7 (2453 plots). These values were subsequently used as the mean starting richness values (i.e. the intercepts) in the power analyses. These values are, as for acid grassland, reassuringly similar (although slightly more variable) to the average empirical richnesses from the NPMS 2015 and 2016 data collected from the relevant habitats (Table 4).

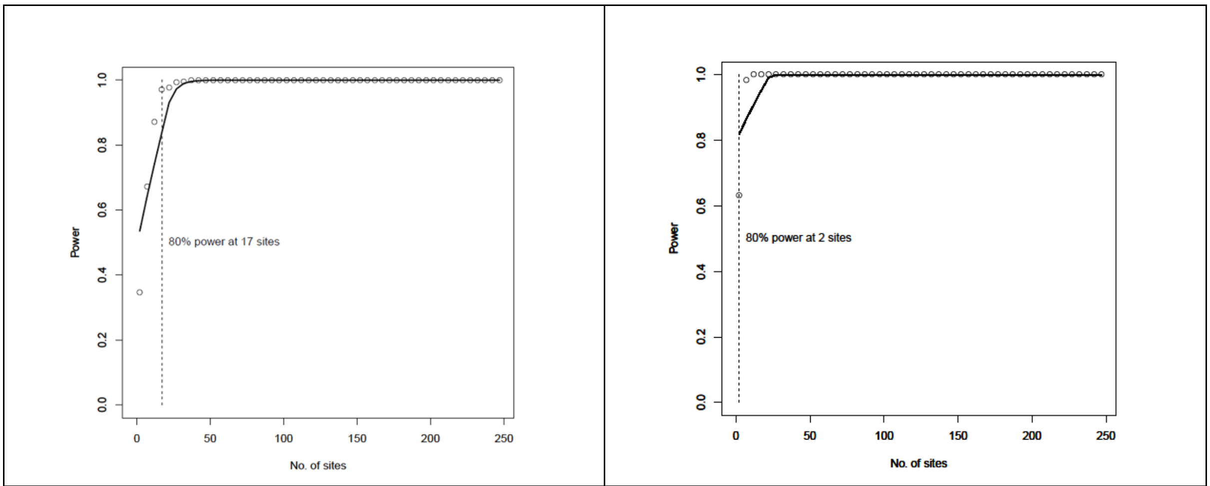
Table 4. NPMS Average species richnesses by fine-scale habitat and survey level (rounded to the nearest integer).

2015	Avg. spp. richness
Dry calcareous grassland	**10**
Wildflower survey	4
Indicator survey	7
Inventory survey	25
Montane calcareous grassland	**17**
Wildflower survey	19
Indicator survey	-
Inventory survey	20
2016	**Avg. spp. richness**
Dry calcareous grassland	**13**
Wildflower survey	6
Indicator survey	8
Inventory survey	21
Montane calcareous grassland	**6**
Wildflower survey	5
Indicator survey	3
Inventory survey	20

The scenarios explored for calcareous grassland are detailed below in Table 5.

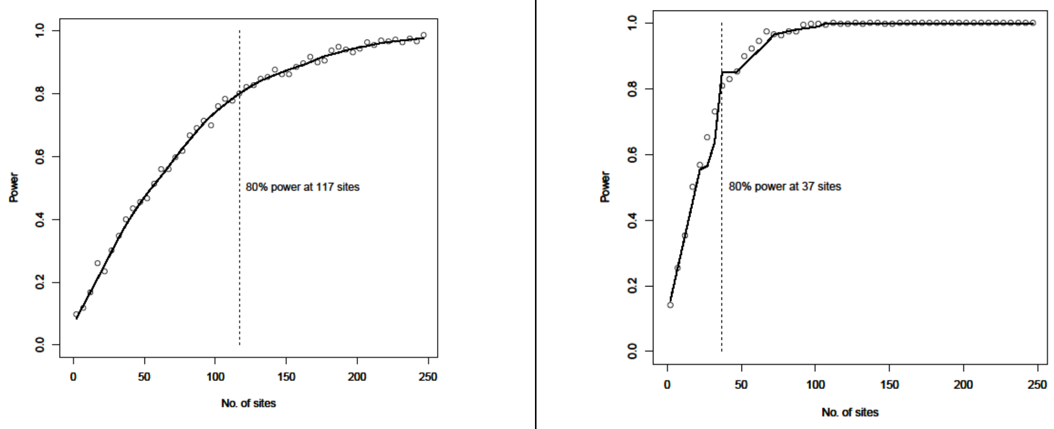
Table 5. Parameter values explored in calcareous grassland power analyses.

Parameter	Values explored
Number of sites (i.e. annual plot samples)	1-250
Number of years	10
Starting species richness	7 (Calcareous grassland Indicator plots) 22 (Calcareous grassland Inventory plots)
Temporal change (i.e. the overall trend)	33% decline over ten years 10% decline over ten years 10% increase over ten years 33% increase over ten years



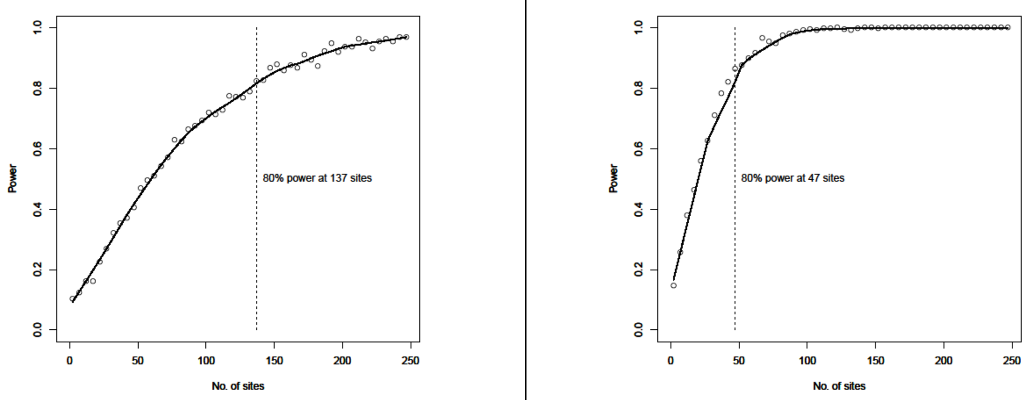
33% decline over ten years, starting abundance 7	33% decline over ten years, starting abundance 22

Figure 6. Power curves for calcareous grassland, indicator (starting abundance 7) and inventory (starting abundance 22) simulated data. Underlying simulations (500) exhibited an average overall decline of 33% in initial species richness over 10 years.



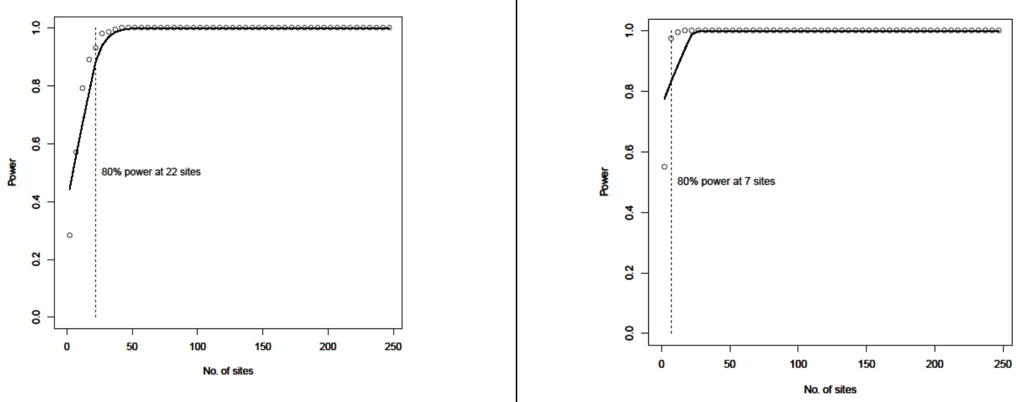
10% decline over ten years, starting abundance 7	10% decline over ten years, starting abundance 22

Figure 7. Power curves for calcareous grassland, indicator (starting abundance 7) and inventory (starting abundance 22) simulated data. Underlying simulations (500) exhibited an average overall decline of 10% in initial species richness over 10 years.



10% increase over ten years, starting abundance 7	10% increase over ten years, starting abundance 22

Figure 8. Power curves for calcareous grassland, indicator (starting abundance 7) and inventory (starting abundance 22) simulated data. Underlying simulations (500) exhibited an average overall increase of 10% in initial species richness over 10 years.



33% increase over ten years, starting abundance 7	33% increase over ten years, starting abundance 22

Figure 9. Power curves for calcareous grassland, indicator (starting abundance 7) and inventory (starting abundance 22) simulated data. Underlying simulations (500) exhibited an average overall increase of 33% in initial species richness over 10 years.

Average Ellenberg N and R (i.e. soil fertility and reaction)

The 2007 Countryside Survey examined change in unweighted (i.e. simple averaged) Ellenberg indicator values between the 1998 and 2007 surveys for most habitats. For the two grassland habitats that are our focus in this report, calcareous grassland plots showed no detectable change in any Ellenberg indicator value, whilst acid grassland plots showed changes in both Ellenberg fertility and pH indicators (Carey et al., 2008). The main acid grassland plots (i.e. those placed according to a robust methodology) showed a small, significant increase for Ellenberg R (reaction, a pH indicator) in England, whilst the targeted plots in this habitat showed increases in Ellenberg N at the scale of Great Britain, and for England and Wales at the country scale. Overall, the average changes reported for the whole of Great Britain were small; for example, the significant increase in Ellenberg N in the targeted acid grassland plots was from 3.03 to 3.12.

An analytical calculation of power (Chow et al., 2007) using these two means and a standard deviation of 0.51 (estimated empirically from the NVC acid grassland pseudo-quadrats discussed above), suggests that a sample size of 1570 plots would be required to detect this size of difference with power of 80% and a significance threshold of 0.05. Recall, however, that a statistically significant change is not necessarily the same as a biologically (or ecologically) significant one.

Maskell et al. (2010) examined the 1998 Countryside Survey data in isolation, within a 'space-for-time' substitution framework. That is to say, a contemporary gradient in N deposition pollution was used as an explanatory variable in a model of spatial variation in Ellenberg indicator values and other metrics. In this framework, community-weighted Ellenberg N and R both showed significant changes, declining in acid grassland with increasing Ndep. However, these authors did not specify the size of these changes, and therefore it is not possible to use them to inform prospective power analyses.

The NVC pseudo-quadrats may, however, be useful for examining the likely impact of the loss of information of the Indicator surveys (relative to the Inventory surveys) when calculating average Ellenberg indicator values for plots. Following the filtering steps explained above in the 'Acid grassland' section, the taxonomy in the NVC pseudo-quadrat dataset was matched to that in PLANTATT (Hill et al., 2004), and average Ellenberg N and R values for the quadrats were calculated for vascular plants. As before, the pseudo-quadrat data were subsequently filtered and the summaries recalculated.

As can be seen from Figure 10 below, the main impact on the plot Ellenberg N values is to create a more clumped, multi-modal distribution. The reason for this is that as plots are filtered to retain NPMS Indicator species only, many plots retain only one or two such species, and the rarer, less typical associate species, are lost. This means that particular mean plot values for the Ellenberg indicator become more common in the dataset. It is notable that visualising the full distribution is crucial for this insight; mean, median, and standard deviation statistics showed little change between the Inventory and Indicator datasets for Ellenberg N. A similar pattern was found for Ellenberg R plot values (Figure 11).

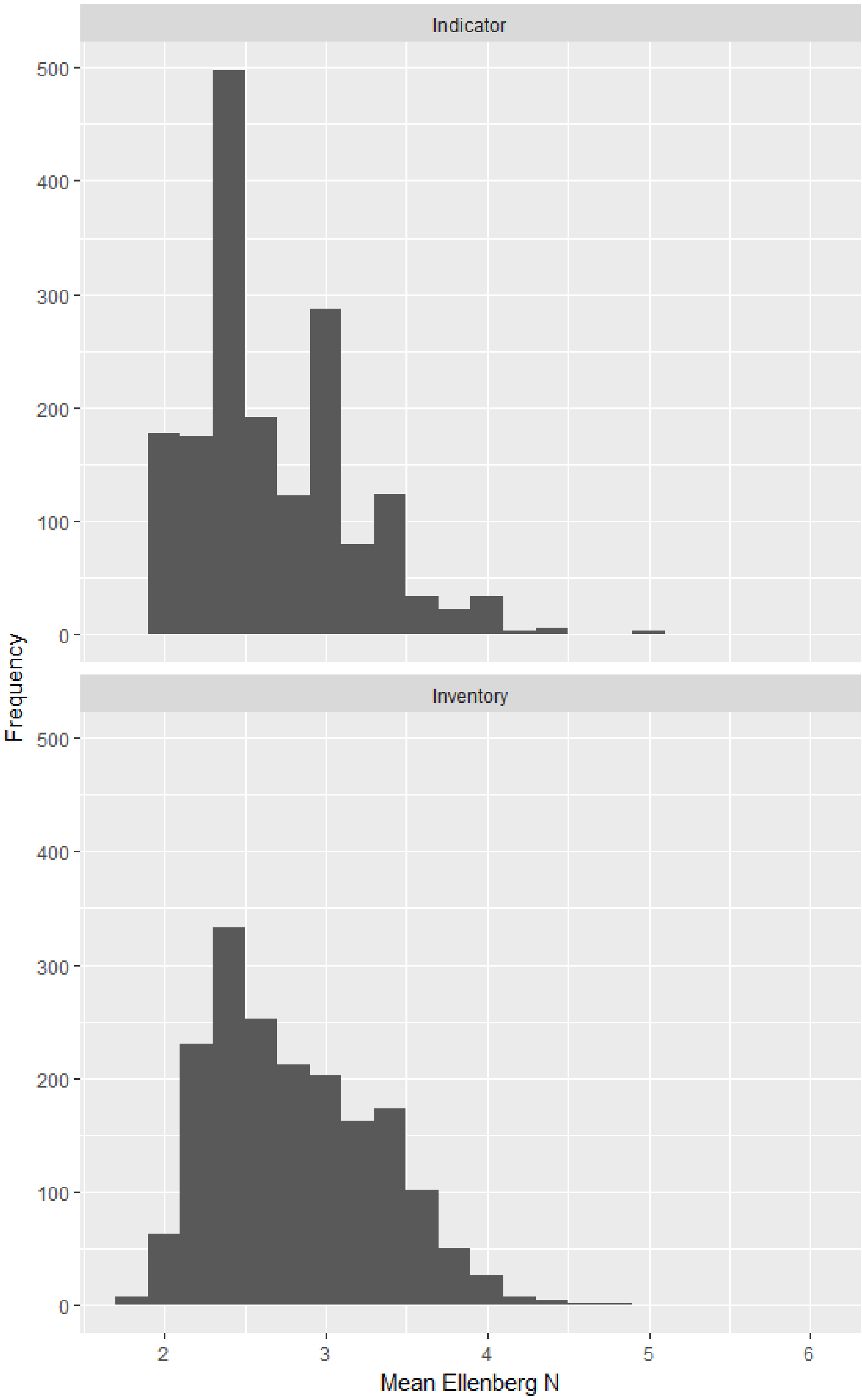


Figure 10. Distributions of average Ellenberg N plot values across all acid grassland NVC pseudo-quadrats, filtered to NPMS Indicator species, and unfiltered (Inventory).

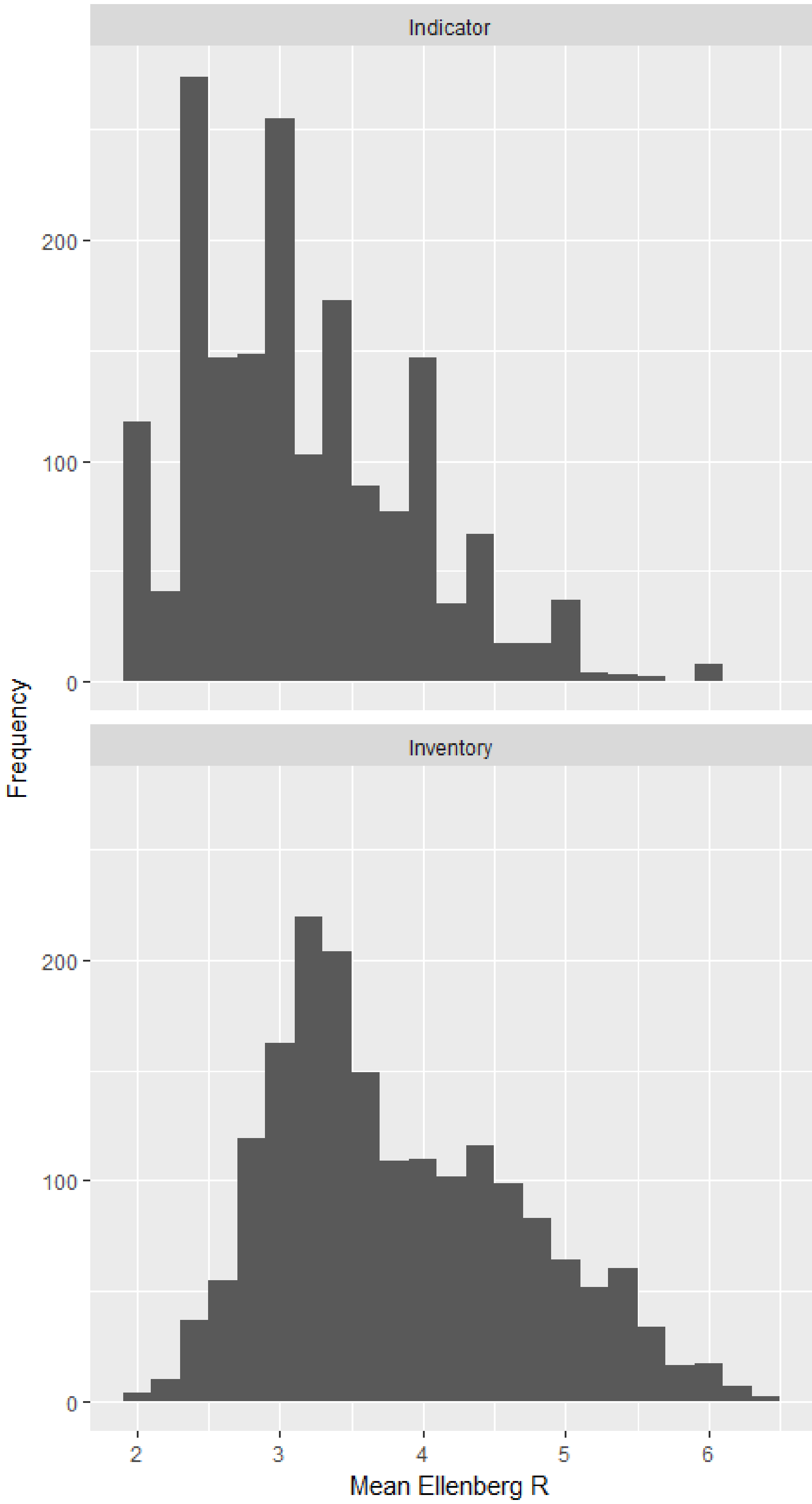


Figure 11. Distributions of average Ellenberg R plot values across all acid grassland NVC pseudo-quadrats, filtered to NPMS Indicator species, and unfiltered (Inventory).

Discussion

The models analysed here give a standard power analysis result, in that a correctly diagnosed decline increase (here in species richness) becomes more likely with an increasing number of sites. Declines or increases in richness are also more easily detected from a higher starting richness, and when the decline or increase is of a larger magnitude. Declines are slightly easier to detect than increases, all else being equal.

Tables 6 and 7 present the plot counts (i.e. the number of unique spatial locations being surveyed) for all NPMS fine and broad scale habitats recorded between 2015 and 2016. Comparison of these numbers to the power curves presented above provide a rule of thumb by which to judge the current likelihood of a particular change being detected over 10 ten years for any given habitat. For example, for dry acid grassland (Table 6), the total number of plots currently available is around 53 (a few more will have been added in 2017) when one considers both Indicator and Inventory plots. This suggests that at the current time only the larger increases or declines in richness are likely to be reliably detected (Figs 2-5).

Table 6. The numbers of plots recorded for NPMS fine habitats, 2015-2016.

NPMS fine habitat	Indicator survey	Inventory survey
Acid fens, mires and springs	25	23
Arable field margins	75	64
Base-rich fens, mires and springs	10	23
Blanket bog	18	18
Coastal saltmarsh	18	23
Coastal sand dunes	5	21
Coastal vegetated shingle	16	18
Dry acid grassland	22	31
Dry calcareous grassland	42	45
Dry deciduous woodland	110	125
Dry heathland	60	30
Hedgerows of native species	147	126
Inland rocks and scree	10	8
Maritime cliffs and slopes	18	8
Montane acid grassland	6	26
Montane calcareous grassland	1	5
Montane dry heathland	6	5
Montane rocks and scree	4	6
Native conifer woods and juniper scrub	7	12
Neutral damp grassland	46	51
Neutral pastures and meadows	87	118
Nutrient-poor lakes and ponds	12	6
Nutrient-rich lakes and ponds	15	32
Raised bog	-	3
Rivers and streams	21	31
Wet heath	35	26
Wet woodland	22	26

Table 7. The numbers of plots recorded for NPMS broad habitats, 2015-2016. Note that the information for arable margins and native pine woods is presented in Table 5, because in these cases there is no distinction made between fine or broad definitions.

NPMS broad habitat	Indicator survey	Inventory survey
Bog and wet heath	47	8
Broadleaved woodland, hedges and scrub	39	54
Coast	11	5
Freshwater	10	7
Heathland	17	3
Lowland grassland	77	62
Marsh and fen	8	10
Rock outcrops, cliffs and scree	2	4
Upland grassland	7	3

Considerations of power for other metrics, such as Ellenberg N or R, suggested that changes across time within plots are likely to be small, and that the numbers of plots required to detect such changes are likely to be outside the scope of the NPMS. However, Ndep impacts on these metrics have been detected in other studies using space-for-time substitution techniques (Maskell et al., 2010; Stevens et al., 2010), and a framework for analysing NPMS data in this way has been developed (WP2 Task 4).

The reduction in the number of species recorded at the NPMS Indicator level does affect the calculation of Ellenberg metrics, as would be expected, however, measures of central tendency such as the mean and median appear to be relatively robust to this loss of information. The resulting distributions of Ellenberg plot metrics after filtering are multi-modal, this suggests that a selective loss of species, driven by a process that preferentially retained or excluded particular species (e.g. eutrophication or acidification) could result in a 'jump' in a Ellenberg plot metric; at this point, changes in Ellenberg measures across time or space may become easier to detect. (This is akin to basing assessments of eutrophication on the appearance or disappearance of one or two species, i.e. an indicator-based approach). Section 2 investigates the effects of reducing the number of species recorded in plots in relation to broader environmental gradients.

Section 2 – Investigating the sensitivity of NPMS Indicator species to broad environmental gradients

Introduction

The National Plant Monitoring Scheme (NPMS) targets areas of semi-natural habitat within 1 km squares that have larger areas of rarer land cover types (according to the CEH Land Cover Map 2007). Contributors are provided with a set of randomly locations to sample within habitat parcels, but for practical reasons, plots in areas that are not publically accessible are likely to be under-represented. Further, in order to encourage the widest possible participation, taxonomic recording in the NPMS can be carried out at three levels that differ in the required botanical skill. Inventory level plots are assumed to be a complete census of the plants present; Indicator level plots allow the recorder to focus just on a predetermined list of species that includes more challenging grasses and sedges; Wildflower level allows recording of a smaller number of flowering plants that are easier to identify. These varying levels of difficulty allow for different points of entry into the scheme; however, given that around 60% of plots are being recorded at Indicator or Wildflower level, the utility of the plot data for answering ecologically interesting and policy-relevant questions about change in vegetation condition is likely to be affected.

Careful construction of the Indicator and Wildflower lists aimed to maximise the relevance and sensitivity of the data recorded at these reduced levels (Pescott et al., 2014; Walker et al., 2015). For example, the Indicator list comprises numerous Common Standards Monitoring indicators as defined by JNCC (2004). Changes in species composition among Indicator level plots should therefore be useful in monitoring condition change in the wider countryside in semi-natural habitats sampled by NPMS volunteers. An obvious and important question to ask is whether the NPMS plot data at the two lower levels of recording intensity (i.e. Wildflower, Indicator) could be used to express species compositional change along broad ecological gradients.

Assuming that recording all plant species in a plot maximises the chance of detecting ecological change, it follows that recording only species listed on the reduced Indicator and Wildflower lists could reduce the sensitivity of the NPMS data to detect change, and may introduce bias so that the position of a reduced-effort NPMS plot along key ecological gradients might be significantly different from its position if all species had been recorded. Here we investigate the effects of filtering quadrat samples using the NPMS species lists. Specifically, a subset of Countryside Survey (CS) quadrats were selected to, as far as possible, match the habitat coverage of the NPMS and the size of quadrat typically used. The principal ecological gradients in the CS data were identified by an ordination approach. The same dataset was then filtered to include only species included within NPMS Indicator or Wildflower sampling levels and plot scores on each axis were recalculated. An analysis of the differences in ordination scores of the original unaltered quadrat species lists versus the filtered lists was then carried out to determine the effect of reduced species recording on the position of plots along the principal environmental gradients.

Data assembly

CS quadrat data are an unbiased sample of British plant communities. Random sampling is stratified by physiographic zones and by Broad and Priority Habitat according to the UK Biodiversity Action plan of 2000 (Jackson, 2000) and subsequent country-level updates. Being a random sample, rarer habitats are under-represented. However samples are numerous and can be analysed to provide an unbiased representation of the species compositional range of Broad and more common Priority Habitats, and of the major ecological gradients along which species assemblages arrange themselves.

To match the target habitats and plot size of the NPMS we extracted all 'X'-plots from the 2007 Countryside Survey where any plot was located in a semi-natural broad habitat (BH) type with enough data to allow analysis at the within-habitat level (Table 7). In many instances the plot will have been assigned to a Priority Habitat (PH) in accordance with the UK Biodiversity Action Plan at that time. Since all PH can be referenced to a parent Broad Habitat the dataset will include sampled PH assemblages. However, using BH as the selection criterion ensured that a range of lower 'quality' assemblages were also included, since these are likely to represent the floristic endpoints of undesirable change occurring within a PH, and are likely to be sampled by the NPMS.

X plots in CS are 200 m^2 in size but are nested. We selected only species recorded up to nest 2, which is a 5 x 5 m square quadrat and therefore of the same size as most NPMS plots. A much larger number of plots are available in the CS database and these would increase sample sizes for the rarer habitats but plot sizes are different (2 x 2 m) and so we restrict the analysis here just to X plots, since these can be made equivalent in size to NPMS. Consequently Montane and Calcareous grassland were excluded.

Table 7. Broad Habitats and numbers of CS X plots that were selected for analysis.

Broad Habitat	**Number of X plots in 2007**
Bracken	46
Neutral grassland	322
Calcareous grassland	9
Acid grassland	212
Dwarf Shrub Heath	187
Bog	282
Fen, Marsh & Swamp	63
Montane	5
Broadleaved, Mixed & Yew Woodland	185

Ensuring consistent nomenclature

Species names and codes from NPMS and CS were compared. Both use the BRC code system yet both schemes amalgamate some taxa given difficulties with reliable identification. The principal requirement was to ensure that species grouped together in either NPMS or CS lists were identified and an entry for the aggregate inserted into each list. This was necessary to ensure that each amalgam received a species axis score so the information carried by the aggregate could still influence the position of each plot in the reduced datasets.

Applying NPMS recording levels to CS data

Taxa included in the CS data but not included in the NPMS Indicator or Wildflower lists were deleted, thereby reducing the information content of the CS data to the equivalent NPMS level. These two datasets – NPMS Indicators or Wildflowers only – therefore convey how each NPMS recording level would represent the vegetation in each CS plot.

Analysis

Detrended Correspondence Analysis (DCA) was applied to the unfiltered CS plot dataset. The species-environment relationship for each major axis was interpreted by inspecting the species points in the ordination space and by passively adding in mean Ellenberg values and cover-weighted canopy height and examining their correlation with each axis.

New plot scores for each axis were produced by calculating averages of the species scores for each plot but based on only the NPMS Indicators or NPMS Wildflowers present in each plot. These new plot scores were then added passively into the ordination space. Means and 95% confidence intervals on the difference between axis plot scores were calculated using a generalised linear mixed model (GLMM) on pairs of plots using the R package lme4 (Bates et al., 2015). That is, the axis plot score based on just the NPMS Indicators or Wildflowers present was subtracted from the axis plot score based on all species present. If the 95% confidence intervals derived from the samples of differences do not include zero then the mean scores are significantly different as a result of filtering by NPMS list. (This is the same statistical principle as the more familiar paired t-test).

Results

The CS dataset comprised 19228 records for 578 taxa distributed among 1311 plots. The number of records and species reduced to 12101 records for 257 species at indicator level and 7560 records for 156 species at wildflower level.

Ordination of CS2007 'X'-plot data

An ordination of the CS plot data extracted two principal axes, the first correlated with substrate fertility and the second with vegetation height, disturbance and successional stage (Fig. 12). The omission of improved grasslands is likely to have strengthened the correlation between high pH and high fertility at the right-hand end of axis 1. If included they would have been much more likely to characterise this end of the axis rather than high pH, less fertile habitat types. A number of arable weeds characterise the disturbed, high fertility, high pH region at top right. These are associated with weedy ex-arable neutral grasslands. Broadleaved woodland is associated with the bottom right region of the ordination. The narrow range of axis 2 scores at the least fertile left-hand side of the ordination reflects the dominance of bog, heath and fen, all habitats with a similar successional status, and the low frequency of wet, sphagnum dominated woodland. Broadleaved woodlands in Britain that are as wet, acid and infertile as Bog are rare.

A somewhat different pattern might have resulted from ordination of actual NPMS datasets in which different habitats may be represented with differing frequency of occurrence. However, it seems unlikely that different ecological gradients would emerge from analysis of a similarly representative NPMS sample of British plant communities.

The effect of NPMS species filtering on association with ecological axes

Differences between unfiltered CS scores and the scores for the NPMS-filtered plots were statistically significant in three out of four instances (Fig. 13). Along axis 1 (fertility; Fig. 12), filtering quadrat lists by NPMS Indicator resulted in a significant shift toward the less fertile end of the axis while filtering just on NPMS Wildflowers shifted mean scores in the opposite direction. However, the magnitude of these significant changes was small with these differences amounting to a 0.6% and 0.4% change respectively along the length of the fertility axis. Along axis 2 (disturbance/succession; Fig. 12) filtering by Indicators made no difference to mean scores, but filtering just on Wildflowers moved the mean axis score for the plots by 6.6% of the total length of the axis and toward taller vegetation (Fig. 12). It is likely that this reflects the omission of grass species that in general are characteristic of grazed, mid-successional vegetation. The minor impact of these changes on the distributions of amended plot scores is also shown in Figure 14. The most obvious effect is for the tail of scores associated with the most highly fertile vegetation to have been truncated when species lists were reduced to NPMS Wildflowers only. This may reflect the absence of arable grass weeds from the Wildflower list. They feature in the CS plot data because the Neutral Grassland subset includes a proportion of plots in weedy ex-arable grasslands.

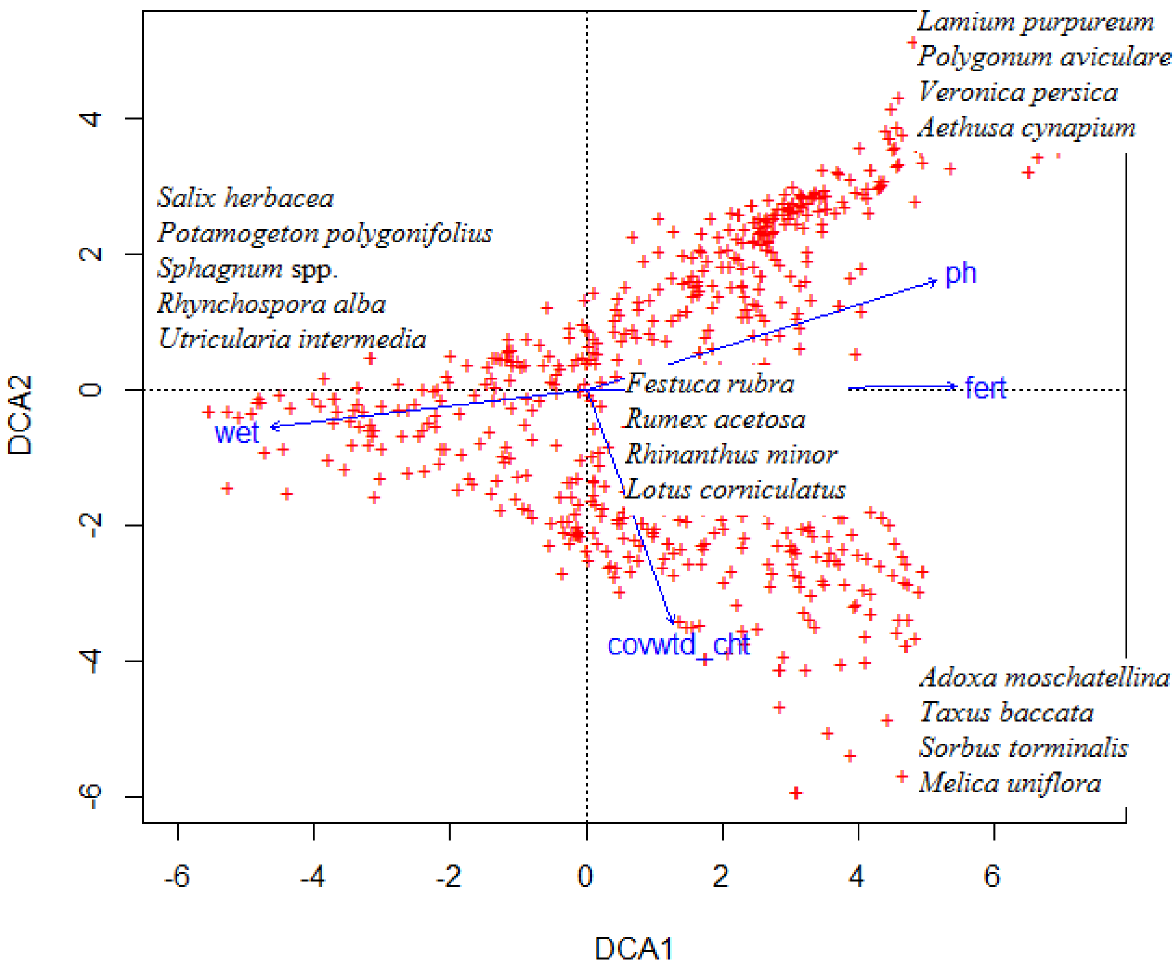


Figure 12. A DCA ordination showing axes 1 and 2. Mean Ellenberg values (N – fertility, R – pH, and F – moisture) and cover-weighted mean canopy height for each CS plot have been added passively to illustrate the association of each axis with ecological factors.

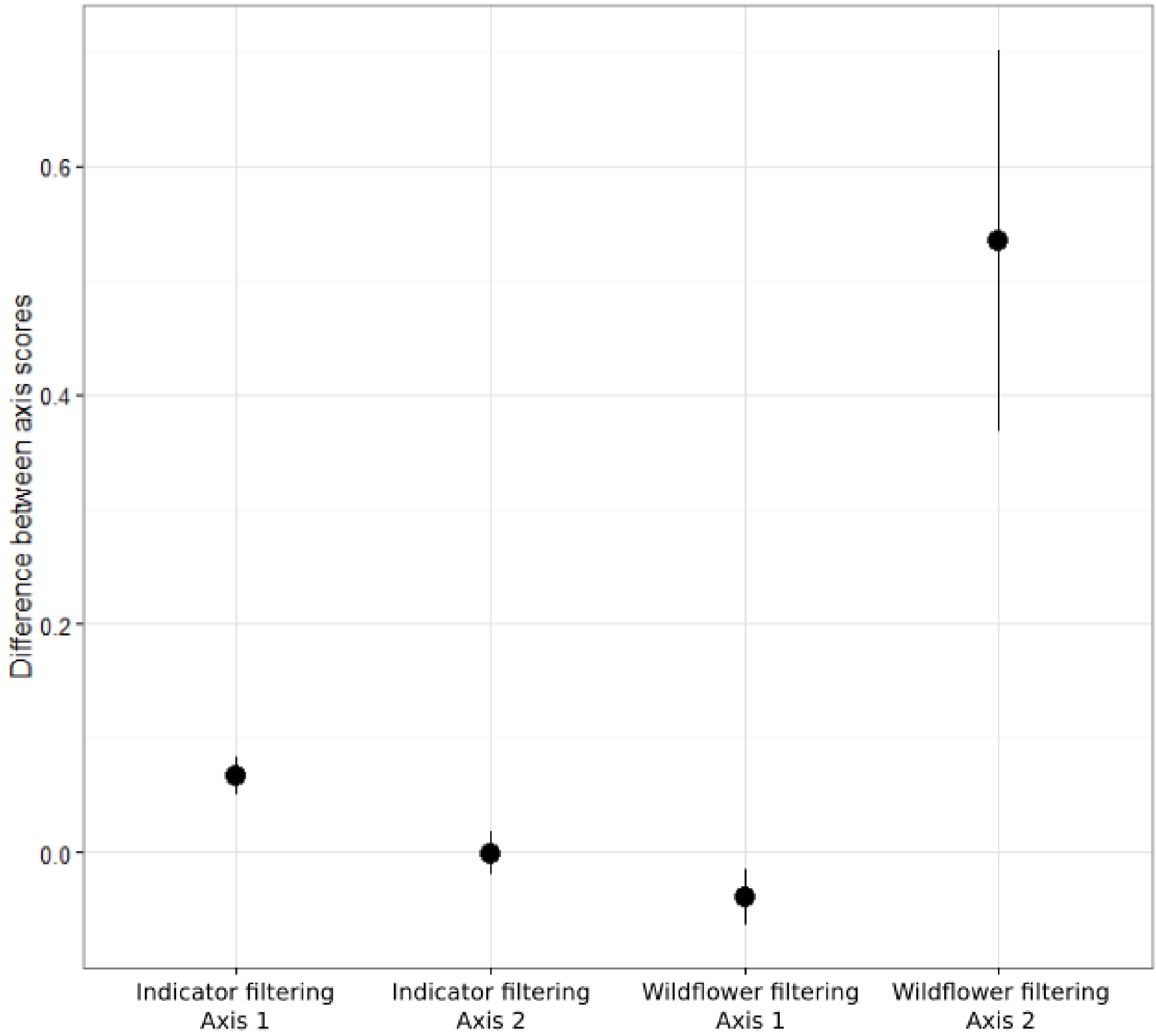


Figure 13. Plot of mean and 95% confidence intervals for the paired differences between axis 1 and 2 scores for CS plots and the same plots but filtered by the NPMS Indicator or Wildflower species lists.

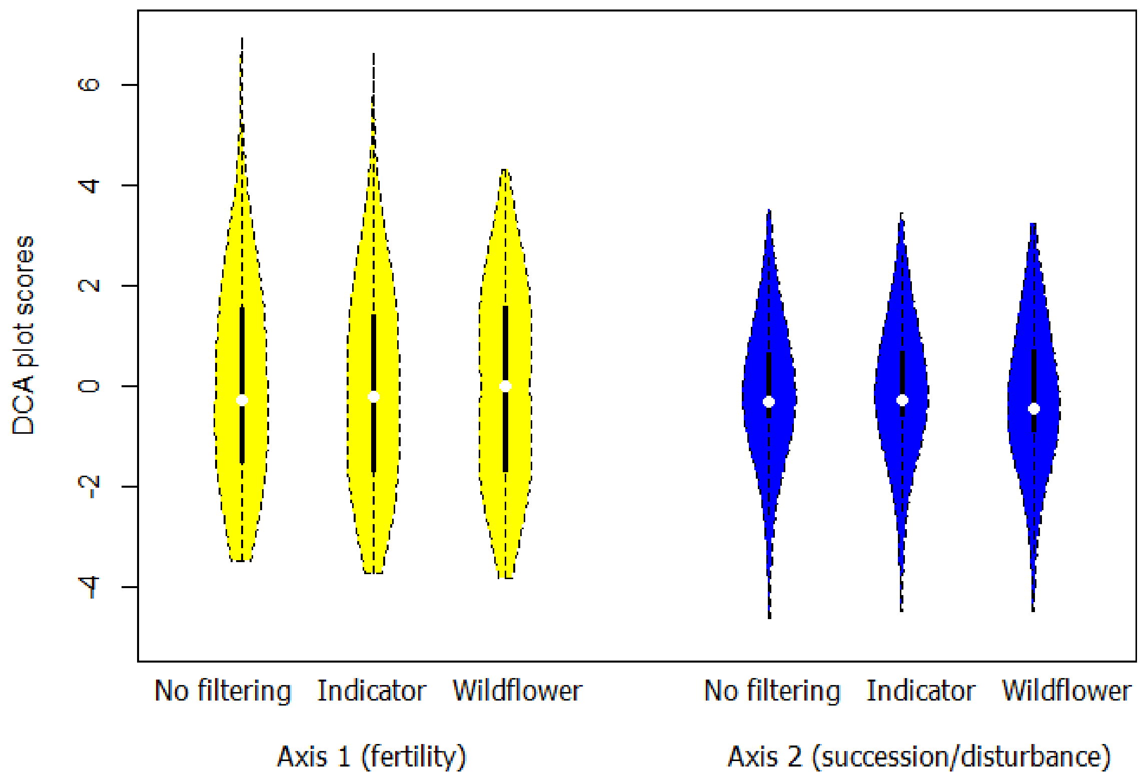


Figure 14. Density/distribution plots of DCA axis scores for the full CS dataset and for the same plots having been filtered by NPMS species lists.

Discussion

We identify the direction and magnitude of changes in the position of plots along major ecological gradients that result from reducing the number of species recorded in each plot according to the NPMS Indicator and Wildflower lists. Such changes are artefactual, in that they are a property of the recording scheme and not of nature. The difference between axis scores for the full versus filtered datasets are a measure of the information cost versus participatory benefit of increasing scheme uptake by reducing the number of species that need to be identified in each plot.

Different levels of change were associated with each DCA axis. This suggests that the absence of a species can have a different impact depending on the gradient concerned (i.e. different species carry different amounts of information in relation to specific environmental drivers). Significant differences in filtered versus unfiltered plot scores were demonstrated, but the size of the resulting shift along ecological gradients was small. Moreover, there was no evidence that filtering of the species lists left significant sections of ecological gradients under-represented. Indeed, apart from the NPMS Wildflower filtering on axis 2 (succession/disturbance), both NPMS filters had remarkably little impact on recovery of the range of each axis.

In conclusion it seems that reduced species recording is not likely to seriously reduce the representativeness of NPMS samples as a sample of the targeted habitats. However, it would be desirable to perform the same analysis carried out here but using an actual sample of NPMS plots to confirm the findings. Possible further areas for exploration include testing the feasibility of including the mean difference between NPMS Inventory level and filtered plots by habitat type as an analytical covariate to adjust for variation in recording intensity when it comes to actual analysis of NPMS data.

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Task 3: Air Pollution Datasets available for use with the National Plant Monitoring Scheme as Indicators of Pollutant Impacts

Air pollution datasets review

Three independent models are in operational use at the Centre for Ecology and Hydrology for the calculation of the spatial distribution of nitrogen (N) and sulphur (S) deposition across the United Kingdom. These are: Concentration Based Estimated Deposition (CBED), the Fine Resolution Atmospheric Multi-pollutant Exchange model (FRAME) and EMEP4UK.

CBED is an inferential model which makes use of measurements of concentrations of nitrogen and sulphur compounds in air and precipitation from the United Kingdom Eutrophying and Acidifying Pollutants (UKEAP) monitoring network (http://uk-air.defra.gov.uk/networks/network-info?view=ukeap). The concentrations of ammonium (NH_4^+), nitrate (NO_3^-) and sulphate (SO_4^{2-}) in precipitation from 37 sites are interpolated across the UK and combined with annual precipitation from the UK Met Office national precipitation monitoring network to calculate wet deposition of N and S at a 5 × 5 km resolution. Monitoring data of both gas (NH_3, NO_2, HNO_3, SO_2) and particulate (NH_4^+, NO_3^- and SO_4^{2-}) concentrations are used with spatial modelling to generate pollutant air concentrations across the UK. This is combined with estimates of deposition velocity from a 'big leaf' model (Smith et al., 2000) to calculate dry deposition to vegetation. CBED deposition data is updated each year as a three year rolling annual average and is available through the Air Pollution Information System (http://www.apis.ac.uk/apis-home?page=1) for the years 2005-2015. The data is used to calculate the exceedance of critical loads for acid deposition and nutrient nitrogen deposition, and to provide evidence to policy makers on the threat to natural ecosystems.

FRAME is a Lagrangian atmospheric chemistry transport model (ACTM) which simulates the emission of pollutants, their vertical diffusion, horizontal transport, atmospheric chemical transport and wet and dry deposition to the surface. The model calculates the annual average deposition of N and S, as well as gas and particulate concentrates, and can be run at either a 5 × 5 km or a 1 × 1 km resolution over the UK. Model performance is evaluated annually by comparison with measurements from the UKEAP monitoring network (Dore et al., 2015). The model has been applied to simulate the historic trends in N and S deposition since 1970 (Matejko et al., 2009) and used with projections of pollutant emissions to estimate deposition for the year 2030 and future changes to ecosystem impacts. Source-receptor data calculated with the model (Oxley et al., 2013) is used in integrated assessment modelling to provide advice to policy makers on cost-effective strategies to abate pollutant emissions and protect human health and natural ecosystems. Historical deposition data are available through the CEH Environmental Information Platform (https://eip.ceh.ac.uk/). A more recent dataset of N and S deposition in the UK for the period 1800 - 2015 is available, but not currently in the public domain. NH_3 concentrations are calculated at a high resolution (1 × 1 km) for the UK, updated annually as a three year rolling average and used to calculate the exceedance of the critical level (Hallsworth et al., 2010).

EMEP4UK is a nested Eulerian ACTM which runs at a 5 × 5 km resolution over the UK. This model is driven by dynamic meteorological data calculated with the Weather Research Forecast (WRF) model (www.wrf-model.org), and uses photo-oxidant chemistry to simulate the four-dimensional physical and chemical state of the atmosphere with data output at a 3-hourly time resolution. The model simulates a comprehensive suite of gas and particulate concentrations including O_3 as well as N and S deposition. A technical description of the model is given in Simpson et al. (2012). The model is applied at a European scale in support of the Convention on Long Range Transboundary Air Pollution (CLRTAP) and is used to assess the impact of air pollution on both human health and natural ecosystems across Europe. EMEP4UK has been applied over the UK to investigate surface ozone (Vieno et al., 2010) and the long range transport of nitrate particulates (Vieno et al., 2014). Deposition and concentration data for the period 2001-2015 are also available through the CEH Environmental Information Platform.

In addition, the Pollution Climate Mapping (PCM) model, developed at Ricardo-AEA, provides concentrations of NO_x and SO_2 at a 1 × 1 km resolution for the UK (Brooks et al., 2015). The model uses a combination of monitoring data for rural background sources, combined with dispersion modelling of large and small point sources and spatial mapping of the contribution from traffic sources.

The ability of ACTMs to represent the air concentration and deposition of pollutants relies on comparison with independent measurements to demonstrate that the models are fit for purpose. A number of different ACTMs, including FRAME and EMEP4UK, were compared with measurements of concentrations of gases, aerosols and precipitation for nitrogen and sulphur compounds as part of the Defra model inter-comparison exercise (Carslaw, 2011; Dore et al., 2015). Generally the models were demonstrated to be fit for purpose based on calculation of the normalised mean bias and the factor of two metric (i.e. the fraction of points greater than half and less than than twice the measured values). The Lagrangian FRAME model performed well for gas concentrations (SO_2, NO_2, NH_3), this was attributed to its high vertical resolution (1 m at the surface) and explicit plume rise representation for point sources. The Eulerian models performed well for particulate concentrations due to their more complex dynamics and chemical schemes. Overall the models tended to predict lower values of concentration in precipitation than those measured by the UKEAP monitoring network. This may be partly caused by the use of bulk collectors in the national chemistry precipitation monitoring newtwork which can over-estimate precipitation concentrations caused by dry deposition to the collector surface (Cape et al., 2009).

Estimates of uncertainty in the deposition calculated by ACTMs can be made using multiple simulations in which model parameters are varied within their range of uncertainty (i.e. Page et al., 2008; Aleksankina et al., 2018). However this only provides an estimate of uncertainty due to the selection of model parameters, and does not include the uncertainty associated with the fundamental assumptions made in constructing a numerical simulation which parameterises highly complex meteorological, physical and chemical processes. One advantage of the approach used in ACTMS over an approach based on interpolation of monitoring data is that ACTMs can be run both backwards and forwards in time provided reliable estimates of atmospheric emissions are available (e.g. Tipping et al., 2017). Monitoring-based approaches however are limited temporally by the availability of data.

Whilst the spatial grid resolution of CBED is 5 × 5 km, the FRAME model is operational at the higher spatial resolution of 1 km x 1 km. Some regional studies have also been undertaken with EMEP4UK at a higher 1 × 1 km spatial resolution. Acurate predictions of air concentration and deposition at high resolution requires accurate estimates of model parameters at this resolution, including atmospheric emissions (of NH_3, NO_x and SO_2), precipitation and land use, which are freely available (i.e. national atmospheric emissions data base, http://naei.beis.gov.uk/; CEH GEAR precipitation, https://eip.ceh.ac.uk/rainfall). However, it is also important to note that uncertainty in concentration and deposition estimates at individual grid squares can increase as the model grid resolution is increased. Hallsworth et al. (2010) applied the FRAME model at a high 1 × 1 km resolution to simulate NH_3 concentrations and the exceedance of the critical level. It was shown that the use of high spatial resolution was effective in spatially separating low emission areas (e.g. nature reserves) from high emission areas of intensive agriculture. This led to more realistic estimates of NH_3 concentrations, particularly over small nature reserves which in a model resolved to 5 x 5 km would not be spatially distinct from the surrounding agricultural land.

Table 1. Summary details for the models discussed.

Model:	CBED	FRAME	EMEP4UK
Pollutants modelled	Wet and dry deposition of N & S	Wet and dry deposition of N & S (plus gaseous and particulate components)	O_3, N and S deposition (plus gaseous and particulate components)
Spatial scales	5 km	1 or 5 km	5 km
Time period	2005-2015	1800-2015	2001-2015
Data platform	APIS	CEH EIP	CEH EIP

NPMS-specific considerations

NPMS surveyors are monitoring plots within 1 × 1 km squares; plot data are then collected at the finer scale of the plot (areas on the order of metres), and entered into the NPMS website at this resolution (www.npms.org.uk). The vast majority of records collected by the scheme then, are resolved to at least 10 x 10 m accuracy. In order to achieve the most sensitive inference from statistical, or other analytical, models applied to the data to uncover associations between air pollutant drivers and vegetation or plant response metrics, it is desirable to match NPMS plot or square data to indices of air pollution at the finest scale available. This will increase the ability of a model to discriminate finer relationships between an air pollutant and, e.g., species richness or a species cover, particularly if different types of land cover (e.g. semi-natural vegetation versus intensive agriculture; Hallsworth et al. 2010) are effectively discriminated at this finer scale. This argues that, with the exception of O_3 (available only from EMEP4UK at 5 km at the current time; Table 1), the 1 x 1 km predictions of the FRAME model are likely to be provide the most analytical power for inferring relationships with NPMS data at the current time. Distinguishing between the impacts of different components of N (i.e. reduced versus oxidised) is possible in theory, however, sensible answers in this regard, and for that matter with respect to the overall impact of N deposition *in toto*, is dependent on minimising bias and variance in the datasets being used. Inference will always be challenged by small effects (e.g. small 'true' regression coefficients) in noisy datasets.

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Task 4: Developing a framework for linking NPMS plant data to pollutant metrics using Bayesian spatio-temporal models

Summary

National Plant Monitoring Scheme data represent species richnesses within plots across a range of UK habitats. As for other monitoring schemes and surveys, such data can be used to quantify the relationships between covariates and response variables. However, such landscape-scale, correlative frameworks can create problems for statistical inference, in that various temporal and spatial correlations, and other effects (e.g. surveyor level), must be taken into account. Unbalanced data (e.g. variable numbers of data points per combination of covariates) can also create problems for the interpretation of such data. Here we investigate a series of models within a Bayesian inferential framework with the aim of finding the 'best' model for the data. 'Best' here is judged in terms of the deviance information criterion (DIC), a within-sample measure of the expected out-of-sample predictive accuracy; the models compared were specified a priori using our knowledge of the expected variables of importance; no further variables were entered into or deleted from models. Using integrated nested Laplace approximation (INLA), a model that jointly estimates an inferred spatial-temporal auto-correlation structure underlying the data was found to give the lowest DIC. Estimates of year and surveyor level effects can be extracted from this model for particular habitats, and examples are provided. The relationship between nitrogen deposition (Ndep) and species richness within two habitats considered to have been impacted by Ndep in the recent past was also investigated, and the marginal relationship between these two variables is displayed graphically. An unexpected sign for this relationship led to a post hoc investigation of possible confounding elements within our dataset, and an increased variability in Wildflower level survey returns at low levels (below the median for the samples considered) relative to high levels is suggested as a possible culprit. The model structure presented thus provides a way of exploring relationships between indices of pollution load and response variables of conservation importance (e.g. species richness), whilst accounting for complex dependency structures between data points.

Introduction

Indicators should be straightforward and easily interpretable. Here we develop a Bayesian framework within which plant species richness data from the NPMS can be related to environmental covariates, including pollution data, whilst also taking account of spatial and temporal correlations. The framework is flexible, and could be extended to incorporate plot data collected under other schemes (e.g. the Countryside Survey; CS).

Framework

NPMS volunteers record species within small plots (typically 5 x 5 m) across the British countryside (NPMS, 2015). Once the plots are selected (an activity restricted to the first year of a volunteer's engagement with the scheme), the volunteer makes two decisions: which level of habitat discrimination to record at (broad or fine) and which level of expertise to record at (Wildflower, Indicator, and Inventory). The volunteer then subsequently aims to visit their plots twice a year, per year, although some plots will be resampled less frequently.

The simplest approach to modelling these data, whilst also allowing for the focus on different subsets of species implied by the two decisions relating to participation level described in the previous paragraph, is to model species richness, adjusting for the different recording level and habitat type. Such an approach allows for the incorporation of all NPMS data and for the estimation of the effects of covariates (such as surveyor level) as an integral part of the modelling process. We note that additional plot data could also be jointly analysed under such a scheme by including additional covariates to account for scheme type (e.g. NPMS versus CS). Measures of pollutant deposition can be incorporated as covariates, as for other approaches to estimating the effects of pollutants on plant species richness using correlative modelling frameworks (e.g. Maskell et al., 2010).

Modern approaches to Bayesian statistics provide a relatively straightforward way of running hierarchical models (i.e. models with various types of non-independence between data points), whilst also allowing for missing data, and accounting for unbalanced and/or small samples (which is likely to be the case for some combinations of habitat and surveyor level within the NPMS). Integrated nested Laplace approximation (INLA), a deterministic algorithm used for rapid Bayesian inference (Rue et al., 2009), uses Laplace approximations to produce marginal posterior distributions for all estimated parameters (Blangiardo and Cameletti, 2015; Rue et al., 2009), and is used here due to its relative speed and flexibility compared to Markov chain Monte Carlo-based (MCMC) methods.

As noted above, the response (i.e. dependent) variable modelled here is species richness per sample (i.e. a visit by a surveyor to a plot). The dataset used is NPMS data collected between 2015 and 2017. The covariates considered are:

- Nitrogen deposition 1996-2000, a 5 x 5 km average (keq ha^{-1} y^{-1}) estimated from the FRAME model v.7.0 (Dore et al., 2007; Matejko et al., 2009) (see Task 3 for more information on this process model).
- The first two principal components of a principal components analysis (PCA) of 10 x 10 km climate data for the UK (including January average temperature 1981-2010; July average temperature 1981-2010; mean number of wet days per year 1981-2010; percentage of peaty soil; and percentage of calcareous rock. Variables likely to be correlated with estimates of Nitrogen deposition (e.g. percentage of arable land and population density), were not included in the PCA (cf. Pescott and Roy, 2016) to avoid confounding and unidentifiable parameters.
- Year – NPMS data for all years of the survey (2015-2017) were included.
- Survey type (i.e. Wildflower, Indicator, and Inventory; see NPMS, 2015 for more information on what these levels mean).
- NPMS habitat (again, see the NPMS website and guidance materials for a complete breakdown of the habitat types that surveyors are asked to look for).

The model development below builds on the framework laid out in Pescott et al. (In press.), in that models treating both spatial and temporal auto-correlation are ultimately considered, and in that both N deposition (Ndep) and climate variables are included. After a description of the model development process, predictions for species richness as a function of Ndep are presented for two NPMS fine-scale habitats of interest: dry acid grassland and dry calcareous grassland.

Model development

We start with a basic spatial model, whereby year, nitrogen deposition and the two climate principal components (PC1 and PC2) are modelled as fixed effects, surveyor type and broad habitat are modelled as random effects, and a random spatial field representing a continuous spatial 'process' (i.e. a particular model of spatial auto-correlation), also treated as a random effect, takes account of spatial autocorrelations. In the model descriptions this is denoted by f(SPDE), where an SPDE (stochastic partial differential equation) is a computationally efficient approach to modelling spatial and spatio-temporal processes (Blangiardo et al., 2013). In all models, species richness is modelled with a Poisson error distribution, as is often the case for low-prevalence count data. Models 1-4 include both spatial and temporal effects, but separately, whereas models 5-8 model them jointly as an intrinsic component of the spatio-temporal models.

Model 0:
The base model as described above.
SppRichness$_{ij}$ = $\beta_0 + \beta_1 * year + \beta_2 * Ndep + \beta_3 * PC1 + \beta_4 * PC2$ + f(type) + f(habitat) + f(SPDE) + $\varepsilon_{i,j}$
Where:
i = plot; j = monad (i.e. 1 x 1 km square);
$\varepsilon_{i,j}$ ~ Poisson(λ)
β_i is the year, Ndep, PC1 or PC2 effect, for i= 1,2,3,4 respectively;
f(SPDE) takes account of between-monad spatial autocorrelation;
f(type) takes account of the dependencies within surveyor level (i.e. level as a random effect);
f(habitat) takes account of the dependencies within NPMS broad habitat (i.e. habitat as a random effect).

This formulation states that beta0 is the mean intercept across all plots and monads; $\varepsilon_{i,j}$ is the random Poisson error term for a sample in plot *i* within monad *j*; and λ is the parameter that defines the Poisson process.

Model 1:
This modifies Model 0 by allowing year to follow a first-order auto-regressive process, AR1, whereby species richness observations at the same site, one year apart, follow a deterministic relationship, whereas observations two or more years apart do not follow the same relationship. This model omits surveyor type and broad habitat as random effects.

SppRichness$_{ij}$ = $\beta_0 + \beta_1 * Ndep + \beta_2 * PC1 + \beta_3 * PC2$ + f(SPDE) + f(year) + $\varepsilon_{i,j}$;
f (year) takes the AR1 format.

Model 2:
This modifies Model 0 by letting year follow a first-order random walk process, RW1, whereby species richness observations at the same site, one year apart, follow a stochastic process, whereas observations two or more years apart do not follow the same relationship. This model omits surveyor type and broad habitat as random effects.

SppRichness$_{ij}$ = $\beta_0 + \beta_1 * Ndep + \beta_2 * PC1 + \beta_3 * PC2$ + f(SPDE) + f(year) + $\varepsilon_{i,j}$;
f (year) takes the RW1 format.

Model 3:
This modifies Model 1 by adding the random effects for surveyor type and broad habitat.

SppRichness$_{ij}$ = $\beta_0 + \beta_1 * Ndep + \beta_2 * PC1 + \beta_3 * PC2$ + f(SPDE) + f(year) + f(type) + f(habitat) + $\varepsilon_{i,j}$;
f (year) takes the AR1 format.

Model 4:
This modifies Model 2 by adding the random effects for surveyor type and broad habitat.

SppRichness$_{ij}$ = $\beta_0 + \beta_1 * Ndep + \beta_2 * PC1 + \beta_3 * PC2$ + f(SPDE) + f(year) + f(type) + f(habitat) + $\varepsilon_{i,j}$;
f (year) takes the RW1 format.

Model 5 onwards can be written in the generic form:

$$SppRichness_{i,j} = \beta_0 + \sum_i \beta_i x_i + \omega_{i,t}$$

Where $\omega_{i,t}$ is a latent spatio-temporal process (i.e. mathematical 'structure'):

$$\omega_{i,t} = a\omega_{i,t-1} + \xi_{i,t}$$

which changes with first order autoregressive dynamics and spatio-temporal covariance function $\xi_{i,t}$.

Model 5:
This modifies Model 1 by changing the separate spatial and temporal processes into a jointly modelled spatio-temporal process, with the temporal process taking the AR1 structure.

$$SppRichness_{i,j} = \beta_0 + \beta_1 * Ndep + \beta_2 * PC1 + \beta_3 * PC2 + \omega_{i,t}$$

Model 6:
This modifies Model 2 by changing the separate spatial and temporal processes into a jointly modelled spatio-temporal process, with the temporal process taking the RW1 structure.

$$SppRichness_{i,j} = \beta_0 + \beta_1 * Ndep + \beta_2 * PC1 + \beta_3 * PC2 + \omega_{i,t}$$

Model 7:
This modifies Model 3 by changing the separate spatial and temporal processes into a jointly modelled spatio-temporal process, with the temporal process taking the AR1 structure.

$$SppRichness_{i,j} = \beta_0 + \beta_1 * Ndep + \beta_2 * PC1 + \beta_3 * PC2 + \omega_{i,t} + f(type) + f(habitat)$$

Model 8:
This modifies Model 4 by changing the separate spatial and temporal processes into a jointly modelled spatio-temporal process, with the temporal process taking the RW1 structure.

$$SppRichness_{i,j} = \beta_0 + \beta_1 * Ndep + \beta_2 * PC1 + \beta_3 * PC2 + \omega_{i,t} + f(type) + f(habitat)$$

Table 1 below summarises the estimates of spatial variance across models. The Deviance Information Criterion for each model is also provided as a comparator of model fit.

Table 1. Estimates of spatial variance and relative fits across models. DIC = Deviance Information Criterion.

Model	**Spatial variance: Median (95% credible interval)**	**DIC**
Model 0	2.19 (1.63, 3.21))	57,898.23
Model 1	7.82 (5.99, 10.21)	67,216.44
Model 2	8.02 (6.04, 11.07)	67,216.02
Model 3	2.66 (2.12, 3.66)	57,897.04

Model 4	2.04 (1.65, 2.73)	57,893.55
Model 5	345.5 (252.4, 550.0)	62,489.47
Model 6	162.8 (96.1, 281.6)	63,192.93
Model 7	9.07 (2.86, 31.55)	56,299.72
Model 8	75.3 (43.5, 134.8)	56,662.03

Note that the estimated spatial variance is much larger in the models which do not take account of surveyor type and broad habitat, and this increases when we move from separate spatial and temporal processes to a joint spatio-temporal process. Adjusting for these two random effects decreases the spatial variance, this is very likely to be due to the fact that the spatial variability in species richness may be partially explained by habitat type (and possibly also surveyor type to a lesser extent).

Model 7 is not only the best fitting model (lowest DIC), but also the model which takes explicit account of the spatio-temporal processes assumed to underlie the data. Model 7 is therefore taken forward as the model from which we extract habitat-specific and year-specific estimates of species richness, as well as estimates of the relationship between the Ndep covariate and species richness for two example NPMS habitats which have previously been the focus of Ndep impact research.

Table 2 presents the estimated species richness for a single habitat type, Broadleaf woodland, across years and survey types. The random spatial fields for 2015, 2016 and 2017 are plotted in Figure 1.

Table 2. Spatio-temporal Model 7 applied to various linear combinations of covariates, using standardised weights for the spatial field (pers. comm. to MJ, R-INLA Helpdesk/Finn Lindgren, 2017).

Broad habitat	**Year**	**Survey type**	**Estimated species richness (95% cred. interval)**
Broadleaf woodland	2015	**Overall**	**7.30 (3.94, 13.51)**
		Indicator	5.59 (4.90, 6.35)
		Inventory	14.43 (12.67, 16.39)
		Wild flower	4.82 (4.23, 5.49)
Broadleaf woodland	2016	**Overall**	**7.32 (3.96, 13.56)**
		Indicator	5.61 (4.92, 6.38)
		Inventory	14.48 (12.72, 16.45)
		Wild flower	4.84 (4.25, 5.51)
Broadleaf woodland	2017	**Overall**	**7.33 (3.96, 13.57)**
		Indicator	5.61 (4.92, 6.38)
		Inventory	14.49 (12.73, 16.46)
		Wild flower	4.85 (4.25, 5.51)

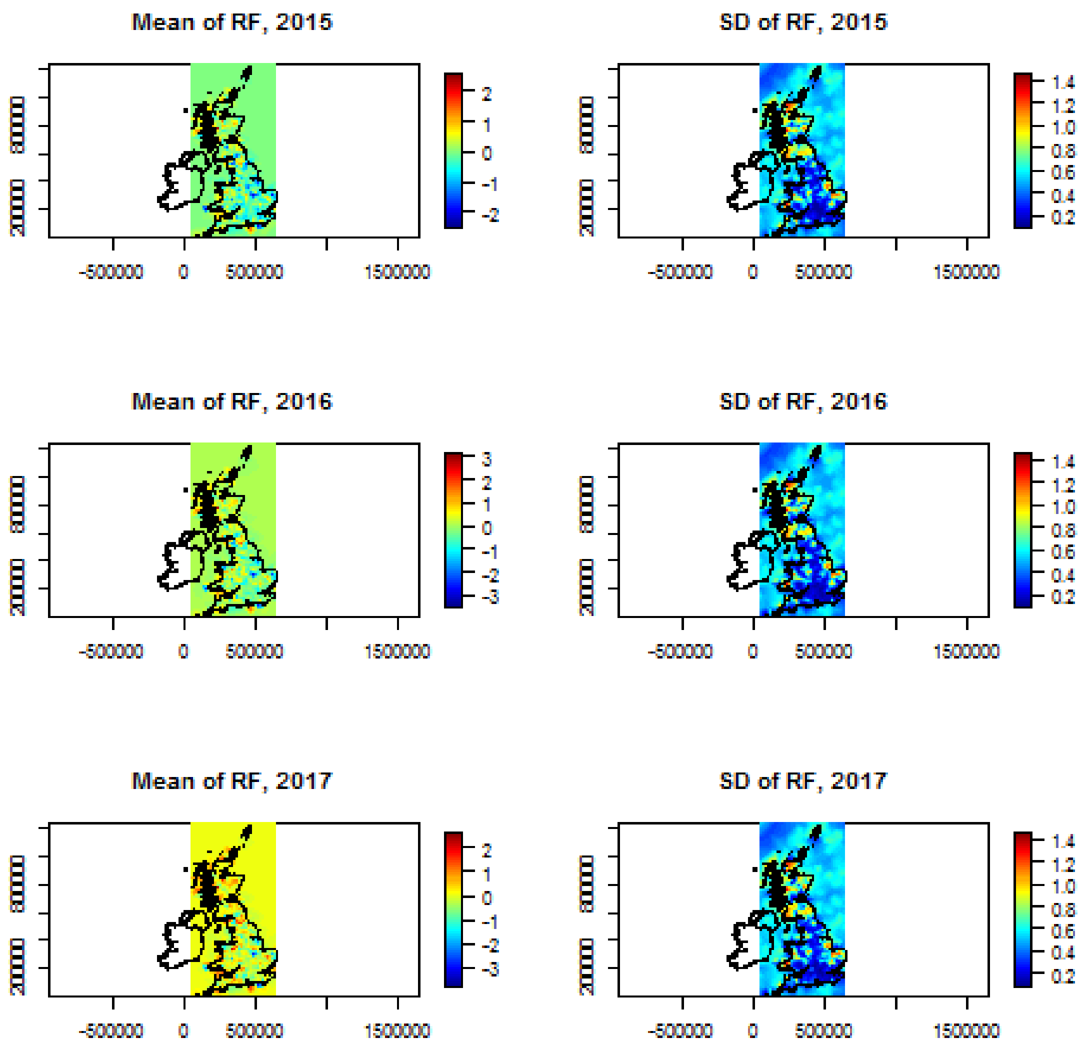

Figure 1. Spatial random fields (mean and standard deviation) for Model 7, split by year.

Both the mean and the standard deviation of the (spatial) random field provide an estimate of the mean species richness (and its standard deviation) on the log scale. Although there are only about 900 monads for which we have data, the remainder of Great Britain has been estimated using the spatial representation of GB modelled by the SPDE.

Model 7 was also used to examine the estimated relationship between species richness and Ndep across all the NPMS data 2015-2017. The relationships for dry acid grassland and dry calcareous grassland (NPMS fine habitats) were estimated using Model 7 and are plotted in Figure 2. The Ndep values (keq ha^{-1} y^{-1}) for which estimates of species richness were made are given in Table 3.

Table 3. Nitrogen deposition deciles (kiloequivalents ha-^{1} y^{-1}) across all NPMS records for the spatio-temporal analysis (Model 7). These values are based upon 9,214 plot samples from 684 surveyed monads, 2015-2017; plots were matched to the corresponding 5 x 5 km grid cell Ndep value from the FRAME v.7.0 model (1996-2000, 5 x 5 km average).

0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
0.32	0.77	0.9	1.03	1.14	1.23	1.35	1.45	1.62	1.85	3.17

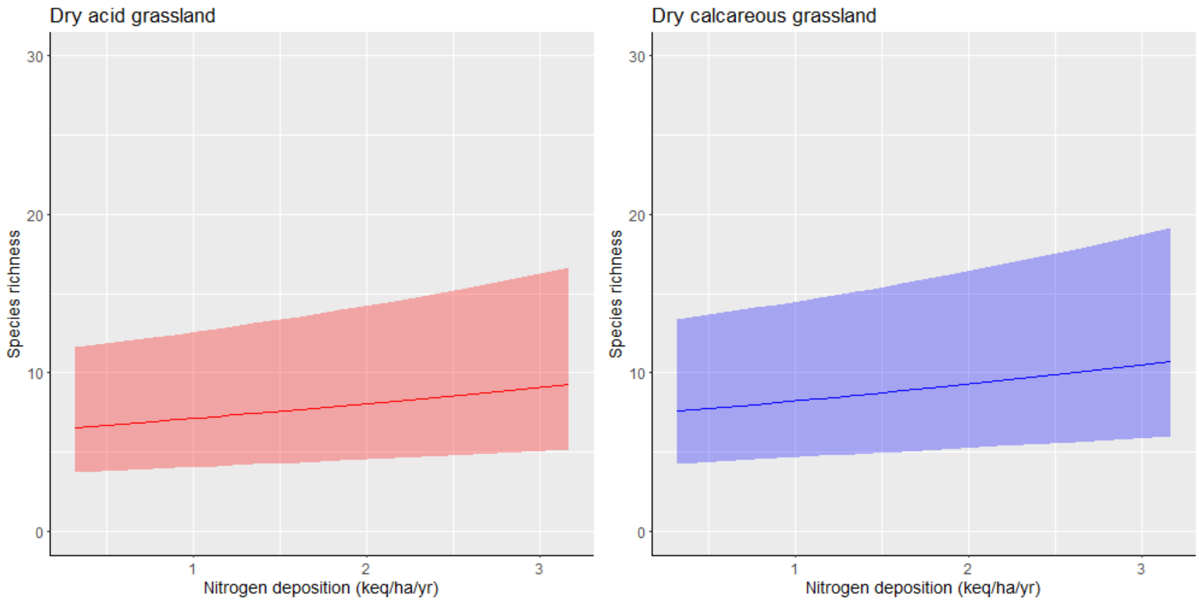

Figure 2. Estimated species richness for the decile levels of nitrogen deposition (Table 3) for dry acid grassland and dry calcareous grassland.

Figure 2 (estimated species richness against nitrogen deposition for both dry acid grassland and dry calcareous grassland) suggests a positive relationship, this is somewhat at odds with previous relationships reported in the literature (see WP2, Task 1).

To confirm this relationship we examined the dry acid grassland data in isolation within a mixed effects modelling framework using the R package 'lme4' (Bates et al., 2015). We used a model structure based on our INLA Model 7 to ensure that the results were as comparable as possible: nitrogen deposition, year, PC1 and PC2 were entered into the model as fixed effects, whilst monad and surveyor type were added as random effects (random intercepts); a Poisson error structure was used. We again found an average positive relationship between species richness and Ndep; however, after removing surveyor type as a random effect this relationship became negative. When surveyor type was reinstated in the model, but with the nested level for Wildflower surveyors being dropped, the association between nitrogen deposition and species richness remained negative. This suggests that the subcategories within surveyor type may be confounding the relationship between nitrogen deposition and species richness (see below). It should also be borne in mind that the relationships plotted in Figure 2 contain enough uncertainty that both zero and negative slopes remain possible.

We investigated the possibility of a confounding effect between surveyor level and species richness through graphical exploration (Figs 3, 4, 5). Splitting the dry acid grassland plots into two categories, above and below the median Ndep across all dry acid grassland plots, and subsequently visualising the distribution of species richnesses across surveyor levels, revealed no clear relationship between these variables (Figure 3), although there is an indication that richnesses recorded at the Wildflower level were considerably more variable in the lower range of Ndep (Fig. 3). Given the current uncertainty in the model outputs generally (Fig. 2), this increased variability in the Wildflower data may be the reason that the estimated average relationship between Ndep and species richnesses changes sign when the Wildflower data are dropped from the analysis.

We also investigated the distribution of surveyor levels across monads for dry acid grassland. It is possible that an association between surveyor level and regional within-habitat variation could also affect the average relationship between Ndep and species richness. There are 20, 27 and 18 unique monads surveyed across the three years for Indicator, Inventory and Wildflower surveys, respectively and their distributions are given in Figure 4. There was, however, no obvious relationship between surveyor level and geography within the dry acid grassland data.

A similar situation to that described above pertained to dry calcareous grassland; the same types of graphical exploratory data analysis revealed a similar situation to dry calcareous grassland (e.g. see Fig. 5).

In conclusion, we have demonstrated the possibility of using a novel Bayesian modelling framework to explore relationships between indices of pollution load and response variables of conservation importance (species richness), whilst accounting for hierarchical dependency structures between data points. We expect that this basic model framework will allow for estimates of both within-year relationships between Ndep (or other) pollutant indices and richness (or other responses), and also changes in these relationships across time, as data accumulate. Note, however, that, if, as hoped, Ndep declines further across the British Isles, then change in Ndep within grid cells, or other model configurations, rather than absolute deposition estimates, may provide better inferences with respect to recovery in the medium term.

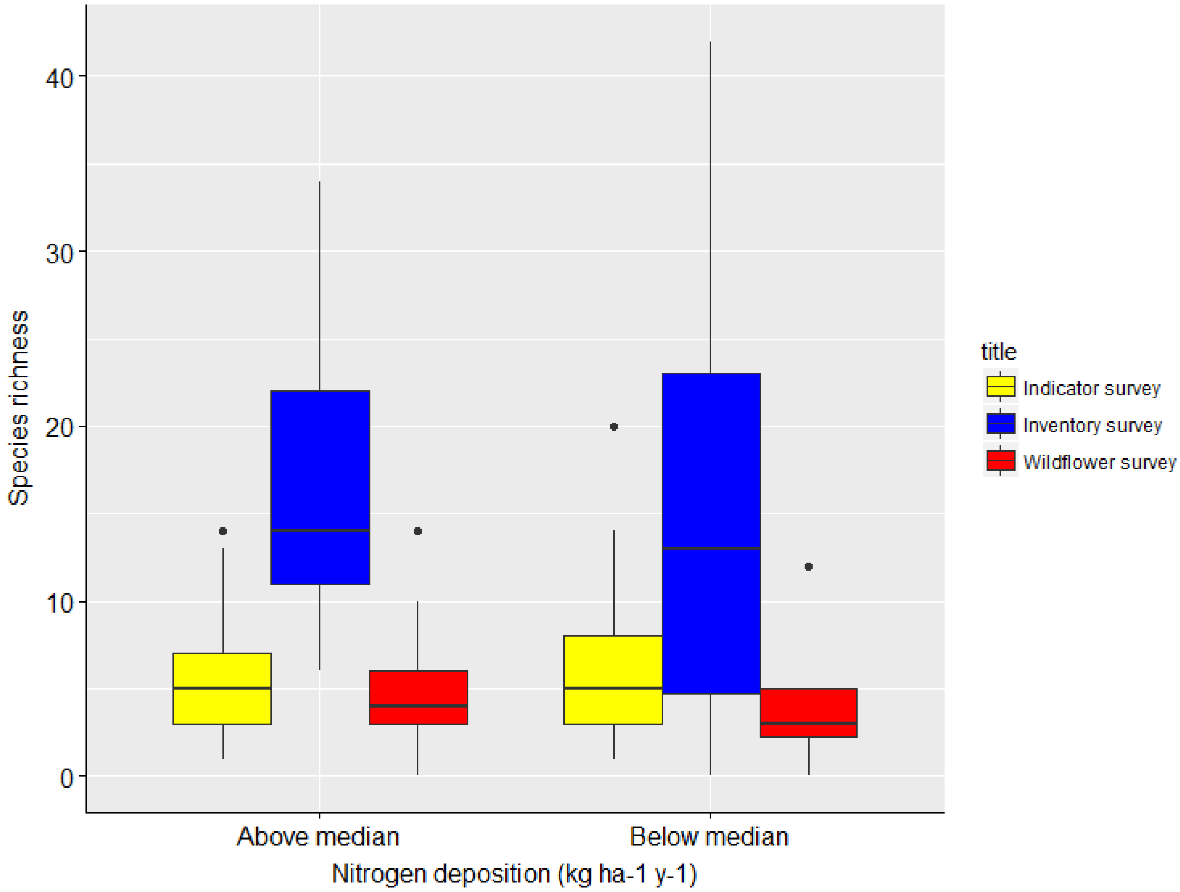

Figure 3: Boxplot of species richness by surveyor type, split by nitrogen deposition (below and above the median).

Figure 4. Distribution of monads surveyed, by surveyor type, for dry acid grassland.

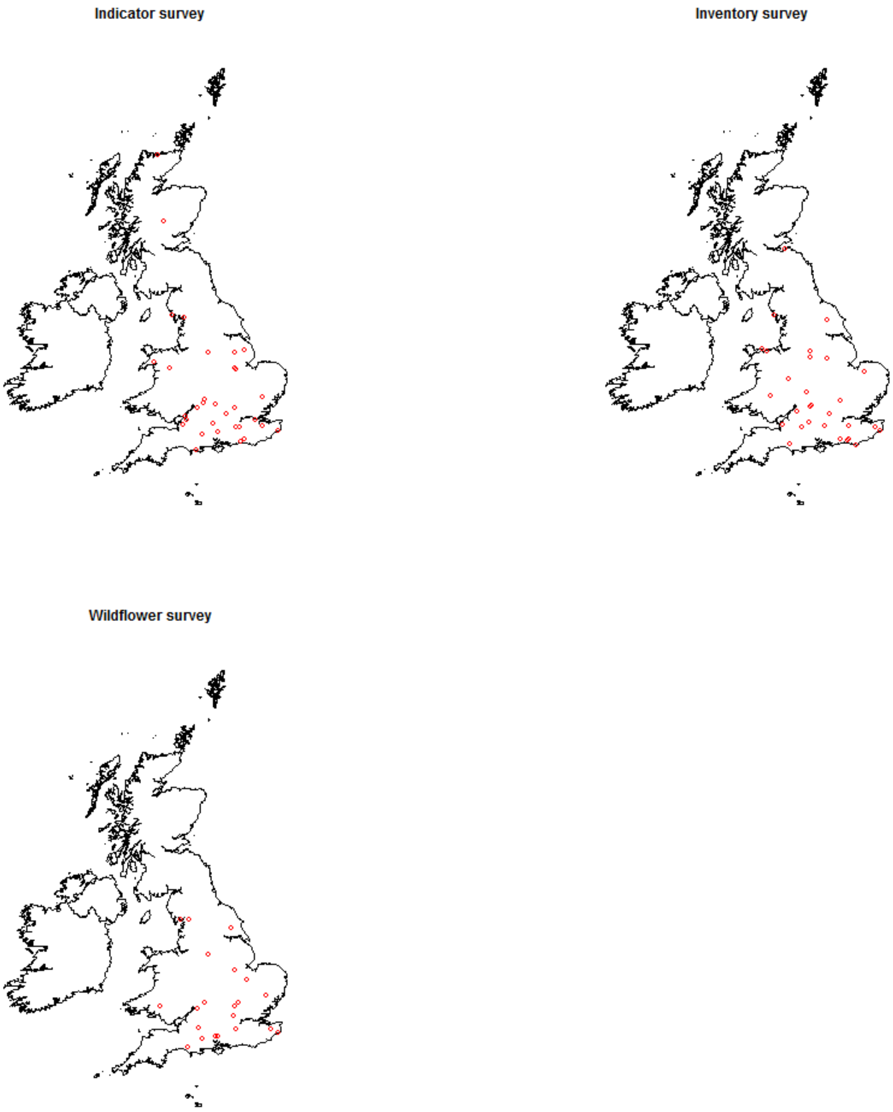

Figure 5. Distribution of monads surveyed, by surveyor type, for dry calcareous grassland.

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