England's statutory biodiversity metric enhances plant, but not bird nor butterfly, biodiversity

Received: 17 February 2023 | Accepted: 9 May 2024

DOI: 10.1111/1365-2664.14697

RESEARCH ARTICLE

Journal of Applied Ecology BRITISH ECOLOGICAL SOCIETY

England's statutory biodiversity metric enhances plant, but not bird nor butterfly, biodiversity

Cicely A. M. Marshall[1] | Kristian Wade[2] | Isla S. Kendall[2] | Hannah Porcher[1] | Jakob Poffley[1] | Andrew J. Bladon[2,3] | Lynn V. Dicks[2] | Jo Treweek[4]

[1]Conservation Research Institute and Department of Plant Sciences, University of Cambridge, Cambridge, UK

[2]Department of Zoology, University of Cambridge, Cambridge, UK

[3]Ecology and Evolutionary Biology Division, Health and Life Sciences Building, School of Biological Sciences, University of Reading, Whiteknights, Reading, UK

[4]eCountability, Chancery Cottage, Cullompton, UK

Correspondence
Cicely A. M. Marshall
Email: cm997@cam.ac.uk

Funding information
Leverhulme Trust, Grant/Award Number: EM-2018-073/2; King's College, University of Cambridge; Ecological Continuity Trust; Basque-Country Government, Grant/Award Number: J.G.A BFI-2010-235; Biotechnology and Biological Sciences Research Council, Grant/Award Number: BBS/E/C/000J0300; Department for Environment, Food and Rural Affairs, UK Government, Grant/Award Number: BD1226; Lawes Agricultural Trust; Heather Trust

Handling Editor: Martin Nuñez

Abstract

1. Biodiversity net gain is a policy focus worldwide, acknowledging ongoing losses of biodiversity to development, and a commitment to offsetting any residual impacts on biodiversity elsewhere. At least 37 countries have mandatory offsetting policies, and a further 64 countries enable voluntary offsets. Offsets rely on credible and evidence-based methods to quantify biodiversity losses and gains.
2. Following the introduction of the United Kingdom's Environment Act in November 2021, all new developments requiring planning permission in England must demonstrate a biodiversity net gain of at least 10% biodiversity net gain from 2024, calculated using a statutory biodiversity metric framework. The metric uses habitat as a proxy for biodiversity, scoring habitats' intrinsic distinctiveness and current condition.
3. We carried out a study of the metric's performance across England in terms of outcomes for biodiversity. We used generalized linear mixed models to regress baseline biodiversity units against five long-established single-attribute proxies for biodiversity (species richness, individual abundance, number of threatened species, mean species range and mean species range/population change). Data were gathered for species belonging to three commonly used indicator taxa (vascular plants, butterflies and birds) from 24 sites, including all terrestrial broad habitats except urban.
4. In baseline assessments, metric-derived biodiversity units correlated with most plant biodiversity variables, but not with any of the bird or butterfly biodiversity variables used in this study. Plant species recorded in habitats with higher baseline biodiversity units had slightly more restricted ranges (slope -16.22 ± 1.52, $p < 0.001$) on average and had shown stronger past declines (slope -0.02 ± 0.00, $p < 0.001$) than those in habitats with lower baseline biodiversity units. Each additional baseline biodiversity unit was associated with a 1% increase in plant species richness ($p < 0.01$).
5. *Synthesis and applications*: Using the statutory biodiversity metric to define 10% biodiversity net gain without additional species-focused conservation management

is likely to translate into small gains for plant biodiversity, and negligible gains for birds and butterflies. We make specific recommendations to improve the metric's efficacy in achieving desirable biodiversity outcomes. Our results provide a valuable case study for other countries interested in developing metrics to support biodiversity net gain policies.

KEYWORDS
biodiversity net gain, biodiversity offset, ecological impact assessment, Environment Act, nature recovery, no net loss, restoration, statutory biodiversity metric

1 | INTRODUCTION

Biodiversity no net loss and net gain have become the focus of biodiversity policies worldwide, prompting the addition of a biodiversity offset step to the mitigation hierarchy (Maron et al., 2016; Zu Ermgassen et al., 2019). Biodiversity offsets are distinguished from other forms of ecological compensation by the formal requirement for measurable outcomes: the losses due to impact and the gains achieved through the offset must be measured in the same way, even if the habitats concerned are different (BBOP, 2012; Treweek et al., 2010). No net loss policy began with the United States Clean Water Act (1972), with offsets emerging as a mechanism to compensate for the residual losses of wetlands that were occurring in the United States, despite application of the mitigation hierarchy to avoid and minimize impacts. At least 37 countries have mandatory offsetting policies, and a further 64 countries enable voluntary offsets (Bull & Strange, 2018). Biodiversity offsetting metrics are now in use in the United States, Germany, France, Brazil, Canada, South Africa and Australia (McVittie & Faccioli, 2020).

In the Environment Act of November 2021, the UK set a legally binding agenda to deliver 'the most ambitious environmental programme of any country on earth', improving or creating habitats to halt the decline in species by 2030 (Department for Environment Food & Rural Affairs, Forestry Commission, Environment Agency, Natural England, and The Rt Hon George Eustice MP, 2021). Under the Act, all new developments requiring planning permission in England must achieve a mandatory biodiversity net gain of at least 10% from 2024, assessed using the statutory biodiversity metric. Nationally significant infrastructure projects are expected to be included in 2025. With the UK committed to building 300,000 homes a year by the mid-2020s, the new net gain requirement is expected to generate a market for biodiversity credits worth an estimated £135 m to £274 m annually, substantially increasing the funding for nature conservation in England (eftec, WSP, and ABPmer, 2021). The metric is thus set to play an increasingly prominent role in nature conservation nationwide. Implementing this policy effectively requires a credible, evidence-based biodiversity loss and gain metric to support consistent determination of biodiversity change. Over time the use of biodiversity offset metrics has become more prescriptive, in recognition of poor outcomes from metrics used in a consultative capacity only, and lack of evaluation capacity among local regulators (Institute for European Environmental Policy, 2014). England follows this trend by mandating use of a statutory biodiversity metric, the success of which will be of interest to other countries developing similar policies and regulations.

As biodiversity cannot be measured in its entirety, single attributes (e.g. invertebrate biomass and species richness), or more commonly combinations of multiple attributes, are used as surrogates for the overall biodiversity associated with a particular area (Defra, 2012). Most metrics used in offsetting programmes are habitat-based, combining habitat extent with some measure of habitat quality (Quétier & Lavorel, 2011). Habitat quality assessment may be detailed, as in the Australian and South African systems (Parkes et al., 2003), or more subjective and simplistic, as in the habitat hectares approach developed for use in the United States and practised in Germany and France (BBOP, 2012; Briggs et al., 2009). In addition, ecological functionality is included in the Canadian system, the economic value of habitat replacement is included in the German system, and the benefits derived by people are included in the German and US systems (McVittie & Faccioli, 2020). The design and implementation of worldwide biodiversity metrics for securing long-term conservation benefits have been reviewed at the Institute for European Environmental Policy, 2014, and some national reviews of metric performance are also available (Quétier et al., 2014; Wende et al., 2005).

In the UK, a national metric for biodiversity accounting has been in development for at least 12 years, using habitat extent, distinctiveness and condition as a proxy of overall biodiversity (Defra, 2012; Treweek, 2009; Treweek et al., 2010). For England, the Environment Act mandates use of an official metric framework that builds on this experience, currently published as the statutory biodiversity metric (Defra, 2023). The metric is expected to be updated periodically. Developers must use the metric to present a post-development scenario that achieves at least a 10% gain in calculated biodiversity units relative to the baseline state, to obtain planning permission under the amended Town and Country Planning Act 1990. The devolved nations (Scotland, Wales and Northern Ireland) are not currently planning to mandate the use of a statutory metric, nor to set minimum legal thresholds for biodiversity net gain, though they share the same commitment to overall nature enhancement during development and planning officers face similar challenges in determining whether net gain is likely to be achieved (CIEEM, 2022).

The UK biodiversity offsetting schemes have been developed largely by government and industry (Collingwood Environmental Planning Limited and IEEP, 2014; Defra, 2012; Treweek et al., 2010), with the topic being hitherto relatively neglected by academics (Hawkins et al., 2022; Robertson, 2021). The choice of metric is a key determinant of success in achieving no net loss of biodiversity (Bull et al., 2014; Zu Ermgassen et al., 2019). The use of habitats as a proxy for biodiversity may overlook the value of habitats to certain species' populations and may also fail to address the needs of species in cases where the amount, type and quality of habitat is not the main driver for population viability (Burrows et al., 2011). Invertebrates may be especially poorly accommodated as they may require several habitats during their lifecycle, or depend on elements of a habitat that are overlooked, undervalued or even identified as a detrimental feature (Pedley & Dolman, 2020; Wilson, 2021). In a preliminary study using an earlier version of Natural England's biodiversity metric 2.0, no consistent relationship was found between metric scores for test locations in southern England and the number of conservation priority species recorded in them (Hawkins et al., 2022).

Here, we evaluate the statutory biodiversity metric's performance by comparing baseline biodiversity unit values with five long-established single-attribute proxies for biodiversity (species richness, individual abundance, number of threatened species, mean species range/population size and mean species range/population change), gathered for three taxa (vascular plants, butterflies and birds), from sites across England representing all natural and semi-natural terrestrial broad habitats. We use our results to make recommendations to improve distinctiveness scores, condition scores and net gain trading rules, relevant not only for future versions of the biodiversity metric in England, but also for all nations grappling with the quantification of biodiversity offsets and biodiversity net gain in the age of nature recovery.

2 | MATERIALS AND METHODS

2.1 | Study sites

Biodiversity units were calculated following field visits by the authors, whilst species data (response variables) were derived from long-term ecological change monitoring datasets collected by the sites and mostly held in the public domain (Table S1). We studied 24 sites across the Environmental Change Network (ECN), Long Term Monitoring Network (LTMN) and Ecological Continuity Trust (ECT). The ECN is the United Kingdom's long-term ecosystem monitoring and research programme that began in 1993, now continued in England as the LTMN; the ECT is a charity dedicated to preserving the national resource of long-term ecological field experiments and facilitating data reuse. We used all seven ECN sites in England. We selected a complementary 13 LTMN sites to give good geographic and habitat representation across England. We included four datasets from sites supported by the ECT where 2×2 m vascular plant quadrat data were available for reuse (Figure 1). All field visits were conducted with permission from the site managers, and in some cases with research permits (Wytham Woods visit number 90687; Chippenham Fen no permit number). The 24 sites included samples from all terrestrial broad habitats (sensu Defra, 2023) in England, except urban and individual trees: grassland (8), wetland (6), woodland and forest (5), sparsely vegetated land (2), cropland (2), heathland and shrub (1). Non-terrestrial broad habitats (rivers and lakes, marine inlets and transitional waters) were excluded. Our samples ranged in biodiversity unit scores from 2 to 24, the full range of the metric. Not all 24 sites had long-term datasets for all taxa: 23 had vascular plant data, eight had bird data, and 13 had butterfly data. We chose these three taxa as they are the most comprehensively surveyed taxa in England's long-term biological datasets, and are thus used as indicators for the national state of nature (Burns et al., 2023). Together they represent a taxonomically broad, although by no means representative, sample of English nature. Permits for animal research were not required as these data were reused from the public domain, and were originally collected non-invasively using observation only.

2.2 | Biodiversity unit calculation

Baseline biodiversity units were attributed to each vegetation quadrat using the statutory biodiversity metric (Defra, 2023; Equation 1). Sites were visited by the authors between April and October 2022, that is within the optimal survey period indicated in the metric guidance. Sites were assessed initially using metric version 3.1 (Panks et al., 2022), which was current at the time of survey, and were subsequently updated to the statutory metric for analysis using field notes and species data. Following the biodiversity metric guidance, we calculated biodiversity units at the habitat parcel scale, such that polygons with consistent habitat type and condition are the unit of assessment. We assigned habitat type and condition score to all quadrats falling within the parcel. Where the current site conditions (2022) and quadrat data (2010 to 2020) differed from each other in habitat or condition, for example the % bracken cover, we deferred to the quadrat data in order to match our response and explanatory variables more fairly. Across all samples, area was set to 1 ha arbitrarily, and strategic significance set to 1 (no strategic significance), to allow comparison between sites. To assign biodiversity units to the bird and butterfly transects, we averaged the biodiversity units of plant quadrats within the transect routes plus a buffer of 500 m (birds) or 100 m (butterflies). Quadrats were positioned to represent the habitats present at each site proportionally, and transect routes were also positioned to represent the habitats present across each site. Although units have been calculated as precisely as possible

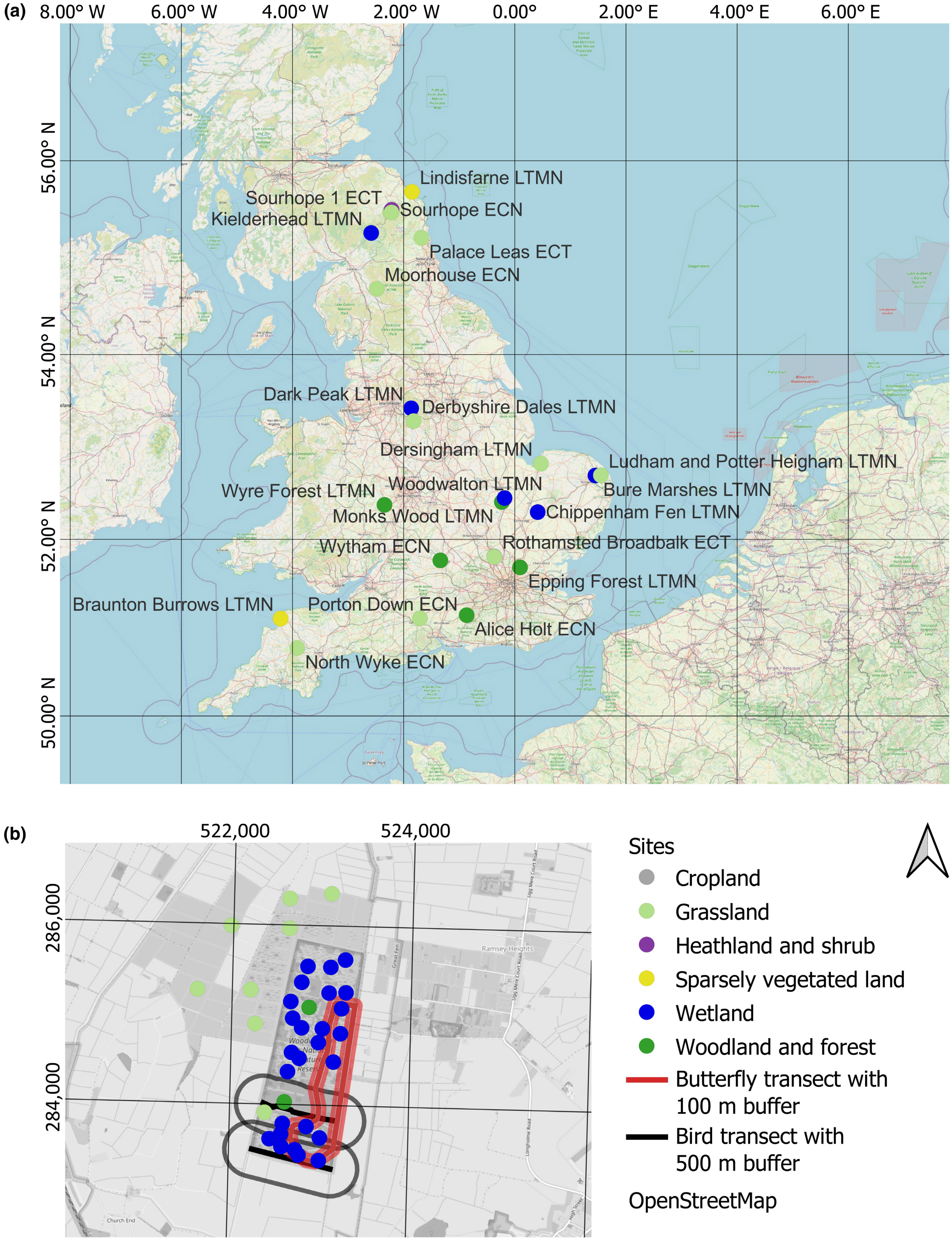


FIGURE 1 (a) Map of study sites across England with latitude and longitude. ECN = Environmental Change Network, ECT = Ecological Continuity Trust, LTMN = Long term Monitoring Network. (b) Example study site, Woodwalton Fen, showing plant sample locations, bird and butterfly transect routes with British National Grid references. Symbols are coloured by broad habitat. OpenStreetMap data are available under the Open Database License (openstreetmap.org/copyright).

for all taxa, we recognize that biodiversity units are calculated more precisely for the plant dataset than the bird and butterfly dataset: the size of transect buffer is subjective, and some transects run adjacent to offsite habitat that could not be accessed. Further details about biodiversity unit calculation can be found in the Supporting Information.

Biodiversity baseline unit calculation following the statutory biodiversity metric (Defra, 2023).

$$\text{Size of habitat parcel} \times \text{Distinctiveness} \times \text{Condition} \times \text{Strategic Significance} = \text{Biodiversity Units} \tag{1}$$

2.3 | Species response variable calculation

We reused species datasets for plants, birds and butterflies recorded by the sites to calculate our response variables (Table S1). Plant species presence data were recorded using 2×2m quadrats of all vascular plant species at approximately 50 sample locations per site (mean 48.1, SD 3.7), stratified to represent all habitat types on site. If the quadrat fell within woodland or scrub, trees and shrubs rooted within a 10×10m plot centred on the quadrat were also counted and added to the quadrat species records, with any duplicate species records removed. We treated each quadrat as a sample point, and the most recent census year was analysed (ranging between 2011 and 2021). Bird data were collected annually using the Breeding Birds Survey method of the British Trust for Ornithology: two approximately parallel 1km long transects were routed through representative habitat on each site. The five most recent census years were analysed (all fell between 2006 and 2019), treating each year as a sample point (Bateman et al., 2013). Butterfly data were collected annually using the Pollard Walk method of the UK Butterfly Monitoring Scheme: a fixed transect route taking 30 to 90min to walk (*c*. 1–2km) was established through representative habitat on each site. The five most recent census years were analysed (all fell between 2006 and 2019), treating each year as a sample point. Full details of how these datasets were originally collected in the field can be found in Supporting Information.

For species richness estimates, we omitted any records with vague taxon names not resolved to species level. Subspecies records were put back to the species level, as intraspecific taxa were recorded inconsistently across sites. Species synonyms were standardized across all sites prior to analysis. For bird abundance we used the maximum count of individuals recorded per site per year for each species as per the standard approach (Bateman et al., 2013). For butterfly abundance, we used sum abundance over 26 weekly visits each year for each species at each site, using a GAM to interpolate missing weekly values (Dennis et al., 2013). Designated taxa were identified using the Great Britain Red List data held by Joint Nature Conservation Committee [JNCC] (2022); species with any Red List designation other than Data Deficient or Least Concern were summed. Plant species range and range change index data followed PLANTATT (Hill et al., 2004). Range was measured as the number of 10×10km cells across Great Britain that a species is found in. The change index measures the relative magnitude of range size change in standardized residuals, comparing 1930–1960 with 1987–1999. For birds, species mean population size across Great Britain followed Musgrove et al., 2013. We used the breeding season population size estimates to match field surveys. Bird long-term population percentage change (generally 1970–2014) followed Department for Environment, Food, and Rural Affairs [Defra] (2017). For butterflies, range and change data followed Fox et al., 2015. Range data were occupancy of UK 10km squares 2010–2014. Change was per cent abundance change 1976–2014. For all taxa, mean range and mean change were averaged from all the species present in the sample, not weighted by the species' abundance in the sample.

2.4 | Analysis

We fitted generalized linear mixed effects models for 14 response variables using the packages *lme4* (Bates et al., 2015) and *nlme* (Pinheiro & Bates, 2022) in R version 4.2.1 (R Core Team, 2022). For all models, biodiversity units were fitted as a fixed effect, whilst site was fitted as a random intercept to account for spatial autocorrelation (plants) and temporal autocorrelation (birds and butterflies). There were 1005 quadrats across 23 study sites for plants, eight transects across eight sites for birds (40 transects total including 5years of data), and 13 transects over 13 study sites for butterflies (65 transects including 5years of data). We checked the validity of modelling assumptions by plotting Pearson residuals against fitted values and observed values. Error distributions for each variable are given in Table 1. Variables were not transformed. *p*-values were calculated using likelihood ratio tests, comparing the specified model to a null model with only the random effect. To visualize variation in community composition across sites and broad habitats, non-metric multidimensional scaling (NMDS) ordinations, grouping samples in multivariate space by their species composition, were carried out using R package *vegan* (Oksanen et al., 2022). For all three taxa, distance matrices were Jaccard, three axes were specified, and all analyses converged with stress <0.2. Biodiversity units were fitted to the ordinations using the function *envfit*. Mapping was conducted in QGIS 3.22.12.

3 | RESULTS

Biodiversity units correlated with three of four plant biodiversity responses, but no bird nor butterfly variables. Habitat parcels with higher biodiversity units had a greater species richness of plants than low biodiversity unit habitats (slope 1.01 ± 1.00, $p = 0.006$), and the plant species in higher biodiversity unit habitats were nationally rarer (slope -16.22 ± 1.52, $p < 0.001$) and had shown greater long-term declines in range size (slope -0.02 ± 0.00, $p < 0.001$), than species in lower biodiversity unit habitats (Table 1, Figure 2). However, all three effect sizes were small. Each additional baseline biodiversity unit was associated with a 1% increase in plant species richness. A decrease in mean plant species range of 16 hectads per one biodiversity unit is small in the context of the variation in range size in the dataset (936 to 2797 hectads). A decrease in mean plant species range change index of −0.02 standardized residual per one biodiversity unit is also small, given variation in the dataset from −0.73 to 1.54. The biodiversity unit results were consistent with the condition score results, with habitats assessed to be in good condition having plant species with smaller mean range sizes, and more

TABLE 1 Generalized linear mixed effects models of the relationship between 14 biodiversity indicators and biodiversity units measured using England's statutory biodiversity metric. For all models, biodiversity units were fitted as a fixed effect and site was fitted as a random effect influencing the intercept of the model. Fitted error distributions are given in brackets after each response variable. Coefficient estimates of the fixed effects, and their standard error, are reported. *p*-values were calculated using likelihood ratio tests between models with and without the fixed effect of biodiversity units. Models in bold type had a significant fixed effect. Estimates have been exponentiated if a log-link model was fitted.

Taxon	Response variable	Term	Estimate	Std. error	Chi	df	*p*-value
Plants (1004 quadrats at 23 sites)	**Species richness (negative binomial)**	Intercept	11.00	1.13			
		BUs	**1.01**	**1.00**	**7.53**	**1**	**0.006**
	Number of designated taxa (Poisson)	Intercept	0.00	10.62			
		BUs	0.97	1.07	0.19	1	0.667
	Mean range (Gaussian)	Intercept	2491.22	55.57			
		BUs	**−16.22**	**1.52**	**107.84**	**1**	**<0.001**
	Mean range change (Gaussian)	Intercept	0.39	0.08			
		BUs	**−0.02**	**0.00**	**127.24**	**1**	**<0.001**
Birds (40 transects at eight sites)	Species richness (Poisson)	Intercept	29.18	1.77			
		BUs	1.00	1.04	0.05	1	0.946
	Individual abundance (negative binomial)	Intercept	125.10	2.01			
		BUs	1.01	1.05	0.05	1	0.825
	Number of designated taxa (Poisson)	Intercept	4.08	1.96			
		BUs	1.02	1.04	0.18	1	0.672
	Mean population size (millions) (Gaussian)	Intercept	1.99	0.36			
		BUs	−0.04	0.02	2.10	1	0.147
	Mean population change (Gaussian)	Intercept	92.57	46.27			
		BUs	−0.76	2.98	0.07	1	0.794
Butterflies (65 transects at 13 sites)	Species richness (Poisson)	Intercept	17.34	1.40			
		BUs	0.99	1.02	0.21	1	0.645
	Individual abundance (negative binomial)	Intercept	1093.46	1.42			
		BUs	1.02	1.02	0.50	1	0.480
	Number of designated taxa (Poisson)	Intercept	0.78	1.00			
		BUs	1.03	1.00	0.35	1	0.556
	Mean range (Gaussian)	Intercept	2025.02	160.05			
		BUs	−4.77	10.84	0.20	1	0.656
	Mean abundance change (Gaussian)	Intercept	37.02	6.65			
		BUs	0.17	0.45	0.15	1	0.698

negative change indices, than moderate and poor condition habitats (Figure 3). For birds and butterflies, there were no significant relationships between biodiversity units and any of the five response variables investigated (Table 1, Figure 2).

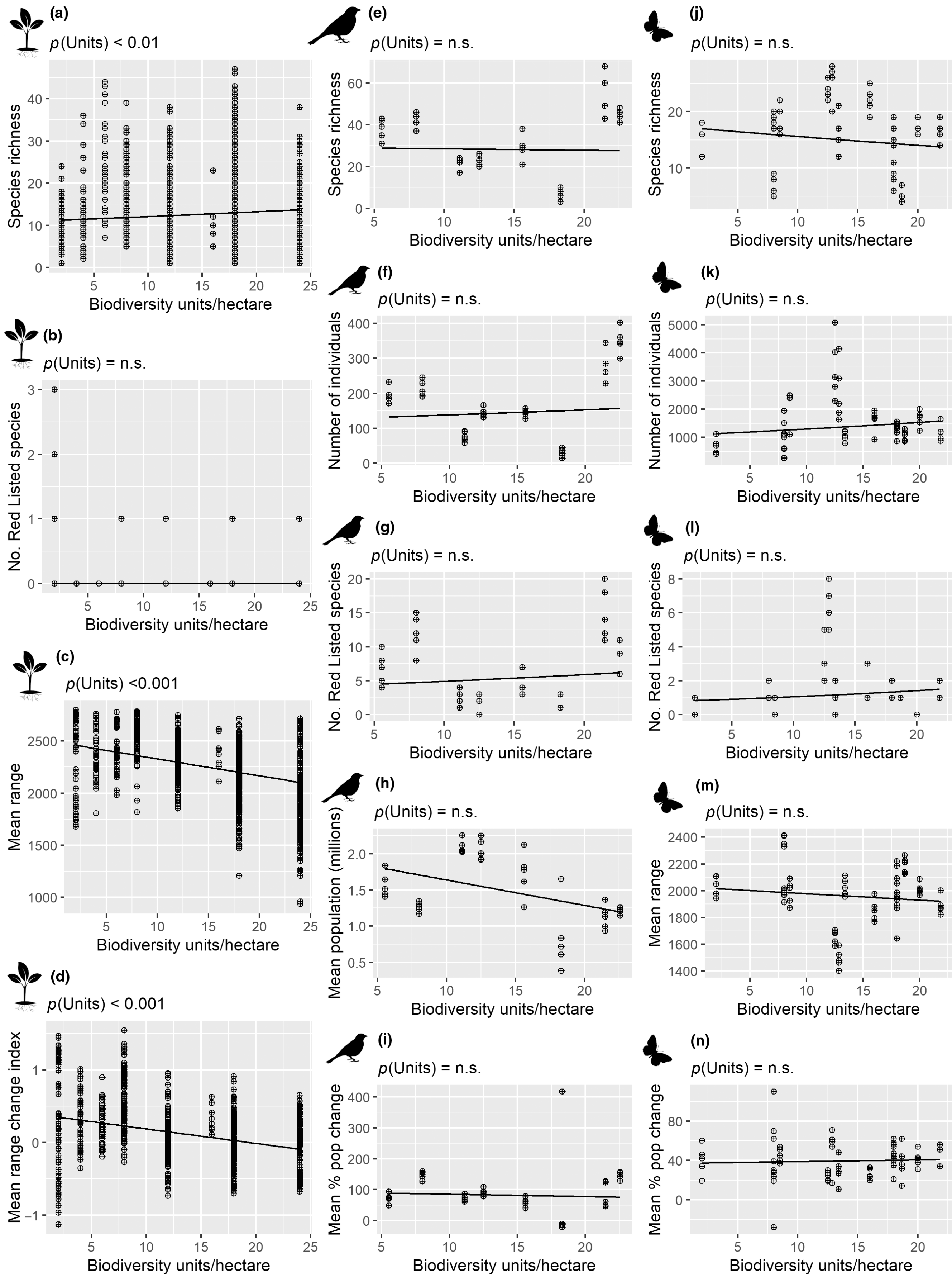
For plants, the NMDS ordination space clearly separated samples by broad habitat, with Axis 1 separating cropland samples from wetland (blanket bog) samples, and Axis 2 separating woodland from grassland samples. Biodiversity units were well correlated with Axis 1. For birds and butterflies, the five survey years of each site were clustered, and there was also clustering by broad habitat (Figure 4).

4 | DISCUSSION

4.1 | Metric performance

The extent to which biodiversity units reflect established biodiversity variables differed by taxon and the choice of biodiversity variable. We found that biodiversity units had explanatory power for three of four metrics of plant biodiversity (species richness, mean species range and mean range change), but no explanatory power for number of threatened plant species, nor any measure of bird nor

FIGURE 2 Scatterplots of statutory biodiversity metric biodiversity units against 14 biodiversity response variables, for (a–d) plants, (e–i) birds and (j–n) butterflies. Points show data from individual quadrats (plants) or transects in different years (birds and butterflies) at all sites. Lines show the fixed effect relationship between biodiversity units and response variables calculated using the generalized linear mixed effect models presented in Table 1. *p*-values follow Table 1 and were calculated using likelihood ratio tests between the model with and without the fixed effect of biodiversity units; n.s. = non-significant.


(a)
p(Units) < 0.01
Species richness
Biodiversity units/hectare
(b)
p(Units) = n.s.
No. Red Listed species
Biodiversity units/hectare
(c)
p(Units) <0.001
Mean range
Biodiversity units/hectare
(d)
p(Units) < 0.001
Mean range change index
Biodiversity units/hectare
(e)
p(Units) = n.s.
Species richness
Biodiversity units/hectare
(f)
p(Units) = n.s.
Number of individuals
Biodiversity units/hectare
(g)
p(Units) = n.s.
No. Red Listed species
Biodiversity units/hectare
(h)
p(Units) = n.s.
Mean population (millions)
Biodiversity units/hectare
(i)
p(Units) = n.s.
Mean % pop change
Biodiversity units/hectare
(j)
p(Units) = n.s.
Species richness
Biodiversity units/hectare
(k)
p(Units) = n.s.
Number of individuals
Biodiversity units/hectare
(l)
p(Units) = n.s.
No. Red Listed species
Biodiversity units/hectare
(m)
p(Units) = n.s.
Mean range
Biodiversity units/hectare
(n)
p(Units) = n.s.
Mean % pop change
Biodiversity units/hectare

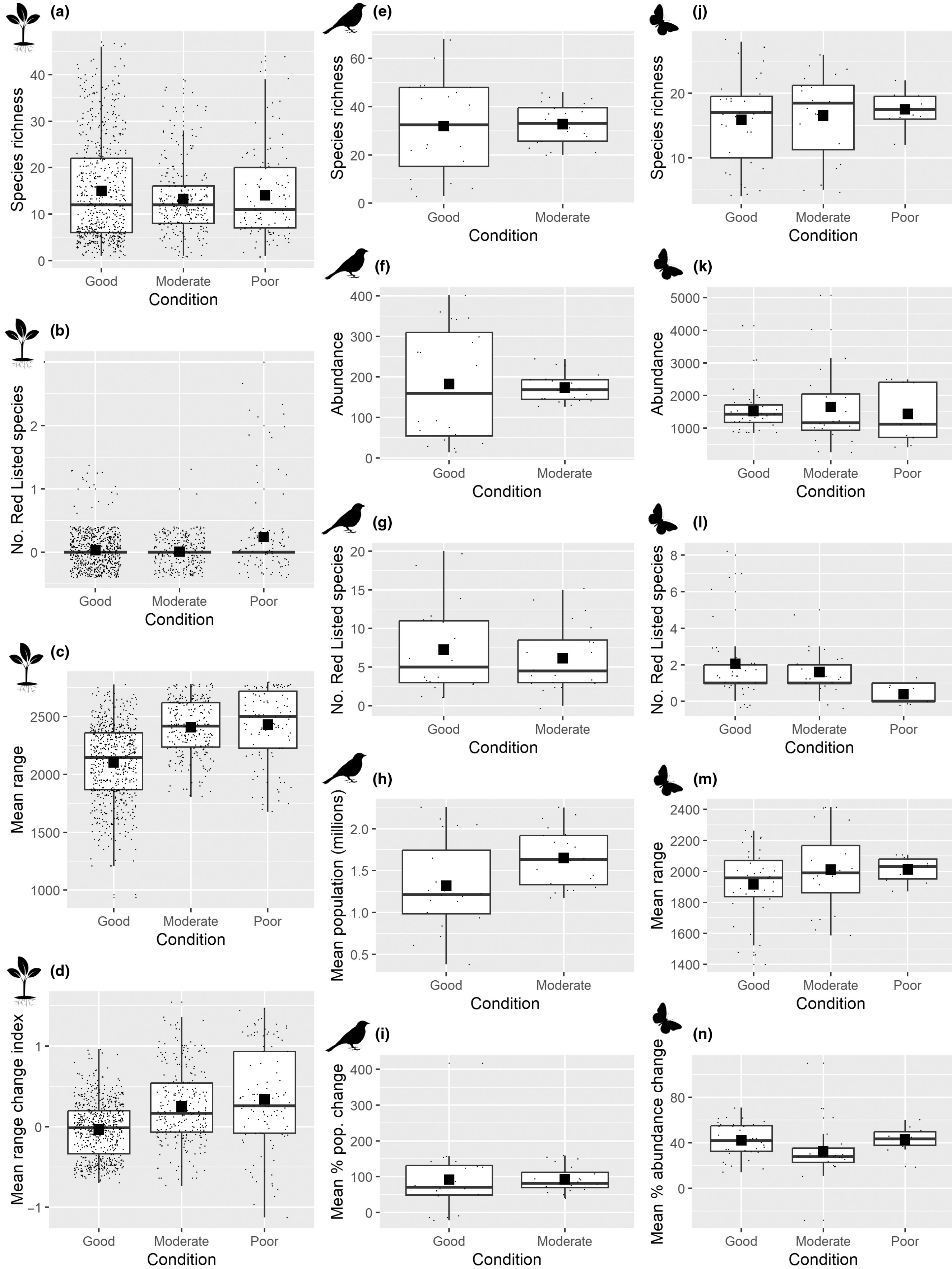

(a)
Species richness
0
10
20
30
40
Good
Moderate
Poor
Condition
(b)
No. Red Listed species
0
1
2
Good
Moderate
Poor
Condition
(c)
Mean range
1000
1500
2000
2500
Good
Moderate
Poor
Condition
(d)
Mean range change index
−1
0
1
Good
Moderate
Poor
Condition
(e)
Species richness
0
20
40
60
Good
Moderate
Condition
(f)
Abundance
0
100
200
300
400
Good
Moderate
Condition
(g)
No. Red Listed species
0
5
10
15
20
Good
Moderate
Condition
(h)
Mean population (millions)
0.5
1.0
1.5
2.0
Good
Moderate
Condition
(i)
Mean % pop. change
0
100
200
300
400
Good
Moderate
Condition
(j)
Species richness
10
20
Good
Moderate
Poor
Condition
(k)
Abundance
1000
2000
3000
4000
5000
Good
Moderate
Poor
Condition
(l)
No. Red Listed species
0
2
4
6
8
Good
Moderate
Poor
Condition
(m)
Mean range
1400
1600
1800
2000
2200
2400
Good
Moderate
Poor
Condition
(n)
Mean % abundance change
0
40
80
Good
Moderate
Poor
Condition

FIGURE 3 Box plots of statutory biodiversity metric condition scores against 14 response variables, for (a–d) plants; (e–i) birds and (j–n) butterflies. Boxes represent median values and the interquartile range (IQR), whiskers are 1.5 x IQR, square symbols represent mean values. Data points from individual quadrats (plants) and transects (birds and butterflies) are plotted as dots (jittered on the *x*-axis for all panels, and also on the *y*-axis for panel b). 'Poor' includes 'fixed at N/A' condition habitats (cereal crops, bracken), as the metric treats these equivalently.

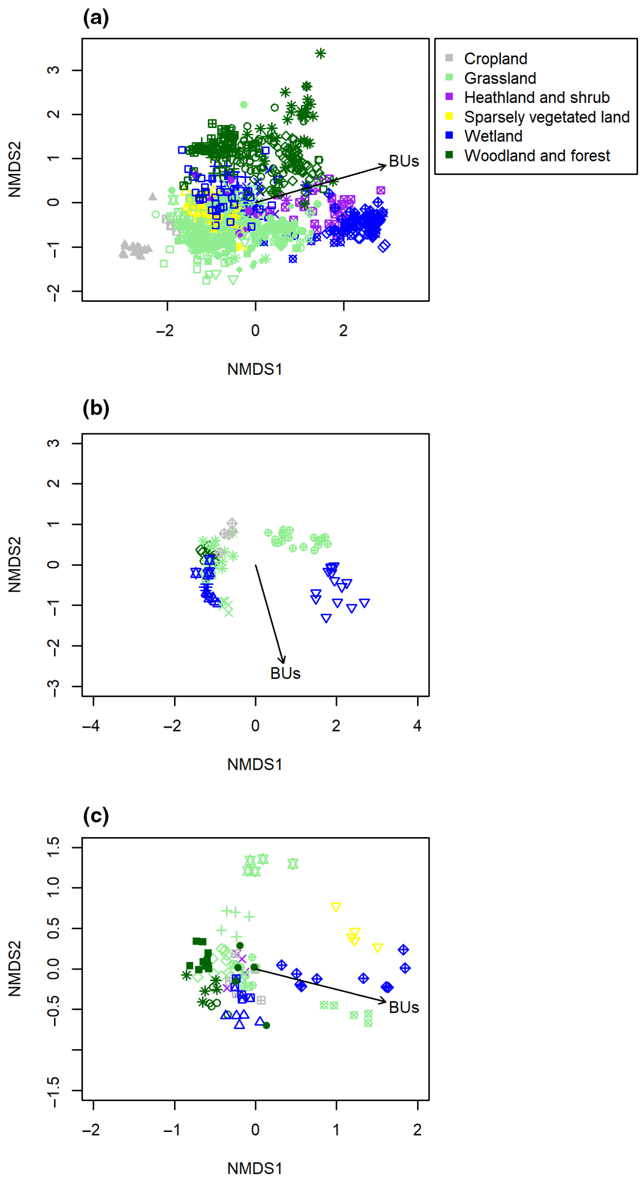


FIGURE 4 Non-metric multidimensional scaling (NMDS) ordinations for (a) plants, (b) birds and (c) butterflies. Samples are coloured by broad habitat type. Plotting symbol types represent up to 24 different sites (legend not shown); each point represents a quadrat (plant data) or a transect in different years (birds and butterflies). The arrow represents the fit of biodiversity units (BUs) to the ordination space ($p<0.05$ for all taxa) showing the direction of increasing BUs, with length proportional to significance.

butterfly biodiversity (Figure 2). It is encouraging that with our large sample of 1004 plant quadrats across all natural and semi-natural terrestrial broad habitats, samples with higher biodiversity units did, on average, have more restricted range species and species showing stronger past range declines than those with lower biodiversity units. The same pattern is found in condition scores for plants, with poor condition habitat supporting more wide-ranging species with growing populations (Figure 3). As well as condition scores, distinctiveness scores are also likely to contribute to the significant relationships found for plants. Distinctiveness scores are pre-assigned for each habitat type based on (i) the total amount of remaining habitat in England, (ii) proportion of habitat protected in sites of special scientific interest, (iii) UK priority habitat status and (iv) European Red List habitat categories. These are derived from or closely related to plant range and range change data, so a close relationship is expected, albeit somewhat obscured by the mixture of designations.

In the context of nature recovery, the lack of any significant relationships between biodiversity units and established biodiversity variables for birds and butterflies is concerning. Hawkins et al. (2022) also found no consistent relationship between a habitat's distinctiveness or condition score (the two main components of biodiversity units) and the number of conservation priority species of any taxon recorded by biological records offices or ecologist walkovers. This may be attributed to the biodiversity metric not capturing aspects of habitat which are important for birds and butterflies, or the scale of metric calculation being inappropriate for taxa with wider-ranging individuals. Habitat condition scores do not take account of the availability of floral resources, a crucial predictor of flower-visiting insect abundance and insect species richness. Habitat heterogeneity, particularly habitat richness and edge density, are important predictors of biodiversity for mobile species like birds, butterflies and other insects (Benton et al., 2003; Martin et al., 2019). This is important at relatively small spatial scales for invertebrates because they use resources from different habitat features within their life cycles (Jachuła et al., 2021) and at larger scales for birds (Morelli et al., 2013). Habitat heterogeneity is barely accounted for by the metric, though small-scale heterogeneity like varied sward heights for grassland and complex storey structure for woodland do feature in some of the condition scoring criteria. Finally, our power to detect a real trend is lower for birds and butterflies compared with plants, as fewer study sites were available for analysis, and the full range of condition scores was not available for birds (no transects were in overall poor condition). Unit calculations for bird and butterfly transects were also subject to greater error than for plant quadrats, for example where a transect buffer included inaccessible vegetation adjacent to the study site. For birds in particular, the relationships are in the directions expected, and it may be that including more sites than has been possible here would reveal significant trends.

Most of the sites we assessed are national nature reserves or benefit from some other form of substantial protection. This is by no means typical of the sites likely to be considered for development, which is the main application of the metric, nor even of proposed offset sites, which need to have potential for enhancement to achieve a net gain. Typical sites encountered by users of the metric are likely to fall at the lower end of the unit scale investigated here, from 2 units/ha for a low distinctiveness habitat in poor condition (e.g. modified grassland), to 12 units/ha for a medium-distinctiveness habitat in good condition (e.g. neutral grassland). Nevertheless, we found a full range of condition and distinctiveness scores across our sites (from 2 to 24, the full range of biodiversity units, and including the lower distinctiveness habitats in poor condition likely to be targeted for development), and we were able to evaluate a full range of habitat types including those with high and very high distinctiveness not commonly encountered in developer datasets (Hawkins et al., 2022). We had no data from urban habitats, aquatic habitats or linear habitats (hedgerows and rivers), nor have we investigated biodiversity unit values when calculated for created or enhanced habitat rather than baseline habitat, which has the additional complexity of negative temporal, spatial and difficulty multipliers (Moilanen et al., 2009).

4.2 | Recommendations for distinctiveness scoring and a proposed change index

Distinctiveness scores are currently derived from a mixture of existing designations, which mix together habitat rarity (in the UK and in Europe) and habitat threat, though these are conceptually distinct and may even be contradictory, introducing noise to the metric. Instead, distinctiveness could be assigned objectively using the mean range of the species supported by the habitat, using existing data for plants and butterflies. Defra should take a view on whether the extent of rarity is to be assessed within England, the UK or the EU, and apply the same to all habitats.

Each habitat could additionally be assigned a change index, derived from an average of its component species' range or population change indices (published at least for plants, birds and butterflies) or extent of remaining habitat. Change scores would be fixed, as they are for distinctiveness, but both sets of scores could be updated every 5–10 years as new datasets are released. Trading rules could then be implemented to prevent nationally declining habitats from being replaced by very large areas of nationally increasing habitats (Glenister, 2022; Zu Ermgassen et al., 2019).

Standardizing the recording of English habitats and their condition for the first time presents a substantial opportunity to collect these data centrally, which would offer the opportunity to evaluate the state of the nation's habitats periodically. Many of England's threatened and declining habitats have not been monitored systematically since the UK Biodiversity Action Plan was retired. Doing this could allow mandatory BNG habitat gains to be explicitly and strategically directed at nationally or locally agreed nature recovery targets (Treweek et al., 2010).

The metric treats biodiversity as linear, with very high distinctiveness habitat accorded a score of 8, high a score of 6, medium 4 and low a score of 2. However, many natural distributions and relationships in ecology are log-linear, with disproportionate influence, thresholds or tipping points observed at one end of a scale (Clark & Luis, 2020). This can weaken the ecological resilience of offsets, so that no net loss does not equate to zero loss (Buschke & Brownlie, 2020). Instead of a linear relationship, a power relationship could be used to account quantitatively for the irreplaceability of very high and high distinctiveness habitats, for example scores of 16 (very high), 8 (high), 4 (medium) and 2 (low), similarly for condition 4 (good), 2 (medium), 1 (poor) rather than 3, 2 1. This extends the idea already in the trading rules that some habitats are so exceptional that losses to them are unacceptable, even if later compensated (Pilgrim et al., 2013).

Whilst almost all habitat types in the metric follow UKHab definitions (UKHab Ltd, 2023), there are some important omissions. For example, the UKHab habitat 'other wetland' does not appear on the statutory metric habitat list. Our data suggest it should be added to the metric, assigned a distinctiveness score of very high, and be subject to the wetland condition scoring sheet. We have only one 'other wetland' sample in the dataset, a *Phalaris arundinacea* reed canary grass swamp at Woodwalton Fen, but the mean range of its plant species is lower than the mean range of any other habitat in the dataset. Other species which may be dominant in the habitat type according to its definition are *Typha latifolia* bulrush and *T. angustifolia*, which both have relatively limited British ranges (1860 and 776 hectads respectively), and *Schoenoplectus* species which also have limited British ranges (8–1202 hectads).

4.3 | Recommendations for condition scoring

The condition scoring sheets for each habitat need revising in light of ecological survey experience. Derived from the Higher Level Agri-environment Scheme Farm Environment Plan condition assessment method (Natural England, 2010), some condition scoring criteria have more to do with describing good condition of land and vegetation from an agricultural perspective than an ecological one. For example, one grassland criterion stipulates that species of sub-optimal condition should account for <5% of the sward, but the listed 'undesirable' species like Creeping Thistle *Cirsium arvense* and Stinging Nettle *Urtica dioica* are nectar-rich and very important to pollinators and several other Critically Endangered native insects (Falk, 2021; Wilson, 2021), though disliked by farmers as they reduce grazing area. There has been refinement of the condition criteria through the initial metric revisions, with the introduction of species richness thresholds for grassland types being particularly helpful to distinguish higher distinctiveness grassland.

The grassland habitat bracken, all cropland habitats, and all low or very low distinctiveness urban habitats like vegetated gardens, have condition fixed at 'N/A' (for which the metric workbook accords a score of 1, i.e. equivalent to poor). This has been justified on the basis that condition has a negligible effect on the overall value of low distinctiveness habitat, but our results do not support this. Our

22 cereal crop samples have the third lowest mean change index of 24 habitats, with only blanket bog and lowland heathland showing more negative indices. Cropland samples at the Rothamsted Broadbalk experiment, where wheat has been grown without the application of herbicide since 1968, are the only samples from any habitat with more than one Red Listed plant species (*Scandix pecten-veneris* shepherd's needle and *Ranunculus arvensis* corn buttercup). Nature-friendly farming systems and plant conservation charities would surely welcome condition scoring applied to cropland, which could produce funding for cornfield annual conservation and sustainable farming practices (Byfield & Wilson, 2005). Our 17 bracken samples have mean plant species richness equal to that of neutral grassland (13.9), and plant species mean range of 2509 tetrads, comparable with that of the other grassland habitats (2221-2606). Often occurring in matrix with other grassland habitats, the grassland condition scoring criteria should be applied to bracken habitat. Vegetated gardens can have many highly valuable habitat features, suppling as much nectar for pollinating insects as nature reserves do (Tew et al., 2021); the urban condition scoring sheet should be applied here.

4.4 | Recommendations for net gain trading rules

Worldwide, there are very few net gain policies that specify a rationale for the gain amount required (Simmonds et al., 2022). England's 10% biodiversity net gain threshold appears to be arbitrary. Though 10% minimum net gain is specified in the Environment Act, local planning authorities may go further, seeking 15% or 20% net gains to be adopted in their counties (Kent County Council, 2022). Our dataset suggests that achieving a 10% net gain would result in no or trivial benefit to nature, thanks to small effect sizes even for significant relationships. England's net gain goal could be both more ambitious and transparent, for example aligned to the Global Biodiversity Framework objective of a tenfold reduction in extinction risk by 2050. We acknowledge that a major conceptual shift has occurred, from the once commonplace scenario of development having large negative impacts on the environment, to no net loss of biodiversity, and now to a mandatory net gain requirement. Comparing a 10% gain to historical major losses rather than no net loss, the gains can be considered much more significant, and perhaps a one-way ratchet.

Our ordination showed plant community composition, and to a large extent bird and butterfly community composition, to be very different by broad habitat. Thus, the current metric requirement to replace at least medium or higher distinctiveness habitat with the same habitat or broad habitat is justified. However, net gain could be targeted better towards nature recovery if compensatory gains were required to match the needs of the specific species and habitats impacted. On the contrary, like-for-like requirements could constrain the flexibility needed to provide habitat mosaics that would be more resilient to climate change, if interpreted very strictly. Most of the priority habitats (following Annex 1 of the Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora, included at Level 5 of the UKHab hierarchy) are not specified in the biodiversity net gain worksheets, instead they are subsumed within their Level 4 habitat types. They should be named directly and be accorded very high distinctiveness so they cannot be traded away, whilst also becoming eligible for named habitat creation.

The biodiversity unit calculations for created and enhanced habitats post-development are subject to negative multipliers to account for spatial, temporal and difficulty of creation risks so that promised habitats must be over-supplied relative to what is lost to guarantee delivery of a net gain (Moilanen et al., 2009). More distinctive and higher condition habitats have large negative difficulty and temporal risk multipliers for habitat creation, discouraging developers from creating high or very high distinctiveness priority habitats and good condition habitats (Glenister, 2022). Creating medium-distinctiveness habitats like mixed scrub or other neutral grassland is usually the most efficient way for developers to achieve biodiversity net gain (Zu Ermgassen et al., 2021). This is particularly regrettable given Defra's early focus on the creation of priority habitat (p. 4 Point 22, Defra, 2012). Negative multipliers should be adjusted so that higher distinctiveness habitats in good condition are incentivized; one way to do this would be to put the high multipliers for high distinctiveness and good condition habitat on the baseline side of the calculation (or at least in the distinctiveness scores, which influence both pre- and post-development calculations), as in our proposed power relationship scoring system, rather than on the habitat creation side of the calculation.

With several quantitative variables interacting with each other within the pre and post-development metric calculations (distinctiveness, condition, strategic significance, temporal and difficulty multipliers), the possibility space of unit outcomes, and thus outcomes for nature, within the metric is very large. A dynamic model of the metric in R will be needed in order to document and explore all possible combinations, to check that trends in unit calculation outcomes are performing as intended and are aligned with national targets for nature and development, and to allow the consequences of any recommended changes to the metric to be explored before they are signed into law.

5 | CONCLUSIONS

We have investigated the performance of Defra's statutory biodiversity metric against five established attributes of biodiversity. We have shown that a 10% net gain calculated using the current metric can be expected to translate into small gains for plant species, but not for other taxa. When supported by species' surveys and analysis, as currently legislated, a habitat metric has a useful place, but targeted conservation action is likely to be necessary to benefit species' populations across taxa. We have made specific recommendations to improve condition scoring, distinctiveness scores, trading rules and country-wide habitat monitoring, which would be relatively cheap and easy to implement, helping the metric to become a more powerful and widely applicable tool. Our results provide a valuable case study for other countries developing biodiversity metrics to support net gain policies.

AUTHOR CONTRIBUTIONS

Cicely A. M. Marshall, Kristian Wade, Isla S. Kendall, Hannah Porcher and Jakob Poffley designed the methodology. Cicely A. M. Marshall, Kristian Wade and Isla S. Kendall collected the data. Cicely A. M. Marshall and Andrew J. Bladon analysed the data. Cicely A. M. Marshall, Jo Treweek and Lynn V. Dicks led the writing of the manuscript. All authors gave final approval for publication.

ACKNOWLEDGEMENTS

We thank the Ecological Continuity Trust for funding the 2022 fieldwork. We thank the ECM and LTMN site managers and data managers for permission to access land and advice on data reuse. CM acknowledges financial support from King's College, Cambridge. AJB and LVD were funded by the NERC Highlight topic GLiTRS project NE/V007173/1. We thank the Lawes Agricultural Trust and Rothamsted Research for the information and data from the e-RA database. The Rothamsted Long-term Experiments National Bioscience Research Infrastructure (RLTE-NBRI) is supported by the UK BBSRC (Biotechnology and Biological Sciences Research Council, BBS/E/C/000J0300, 2017-2023) and the Lawes Agricultural Trust. We thank Rob Marrs for providing the Sourhope ECT dataset for reuse. The Sourhope experiments were funded by Defra (Project BD1226, 1993-2007), the Basque-Country Government (J.G.A BFI-2010-235), the Heather Trust (2013) and the Leverhulme Trust (2019, EM-2018-073/2). The James Hutton Institute and R.W. and S.D. Flintoff allowed the experiments to be sited and maintained on the Sourhope Estate. We thank Simon Peacock and Janet Simkin for permission to use the Palace Leas dataset and for access to the site. We thank David Coomes for the comments on the manuscript. We thank Sky Wallis, Peter Hadfield, Jill Marshall and David Marshall for assistance with fieldwork. Contains JNCC/NE/NRW/NatureScot/NIEA data © copyright and database right 2022. Contains data supplied by UK Centre for Ecology & Hydrology.

CONFLICT OF INTEREST STATEMENT

Prof Lynn Dicks is a non-executive board member of Natural England and serves as a co-chair of Natural England's Science Advisory Committee (NESAC).

DATA AVAILABILITY STATEMENT

Data are available via the Dryad Digital Repository https://doi.org/10.5061/dryad.n5tb2rc0k (Marshall et al., 2024).

ORCID

Cicely A. M. Marshall https://orcid.org/0000-0002-7397-6472

REFERENCES

Bateman, I. J., Harwood, A. R., Mace, G. M., Watson, R. T., Abson, D. J., Andrews, B., Binner, A., Crowe, A., Day, B. H., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Lovett, A. A., Munday, P., Pascual, U., … Termansen, M. (2013). Bringing ecosystem services into economic decision-making: Land use in the United Kingdom. *Science (80-)*, *341*, 45–50. https://doi.org/10.1126/science.1234379

Bates, D., Maechler, M., Bolker, B., & Walker, S. (2015). Fitting linear mixed-effects models using lme4. *Journal of Statistical Software*, *67*, 1–48.

BBOP. (2012). *Biodiversity offset design handbook*. Forest Trends.

Benton, T. G., Vickery, J. A., & Wilson, J. D. (2003). Farmland biodiversity: Is habitat heterogeneity the key? *Trends in Ecology & Evolution*, *18*, 182–188. https://doi.org/10.1016/S0169-5347(03)00011-9

Briggs, B. D. J., Hill, D. A., & Gillespie, R. (2009). Habitat banking-how it could work in the UK. *Journal for Nature Conservation*, *17*, 112–122. https://doi.org/10.1016/j.jnc.2008.12.006

Bull, J. W., Milner-Gulland, E. J., Suttle, K. B., & Singh, N. J. (2014). Comparing biodiversity offset calculation methods with a case study in Uzbekistan. *Biological Conservation*, *178*, 2–10. https://doi.org/10.1016/j.biocon.2014.07.006

Bull, J. W., & Strange, N. (2018). The global extent of biodiversity offset implementation under no net loss policies. *Nature Sustainability*, *1*, 790–798. https://doi.org/10.1038/s41893-018-0176-z

Burns, F., Mordue, S., al Fulaij, N., Boersch-Supan, P., Boswell, J., Boyd, R., Bradfer-Lawrence, T., de Ornellas, P., de Palma, A., de Zylva, P., Dennis, E. B., Foster, S., Gilbert, G., Halliwell, L., Hawkins, K., Haysom, K. A., Holland, M. M., Hughes, J., & Jackson, A. C. (2023). *State of Nature 2023*. The State of Nature Partnership.

Burrows, L., Butcher, B., & Treweek, J. (2011). Offsets for species in the UK planning context: A possible methodology. *In Practice: Bulletin of the Institute of Ecology and Environmental Management*, *71*, 41–43.

Buschke, F., & Brownlie, S. (2020). Reduced ecological resilience jeopardizes zero loss of biodiversity using the mitigation hierarchy. *Nature Ecology & Evolution*, *4*, 815–819. https://doi.org/10.1038/s41559-020-1177-7

Byfield, A., & Wilson, P. J. (2005). *Important arable plant areas: Identifying priority sites for arable plant conservation in the United Kingdom*. Plantlife International.

CIEEM. (2022). *Welsh Government's Approach to Net Benefits for Biodiversity and the DECCA Framework in the Terrestrial Planning System CIEEM Briefing Paper.*

Clark, T. J., & Luis, A. D. (2020). Nonlinear population dynamics are ubiquitous in animals. *Nature Ecology & Evolution*, *4*, 75–81. https://doi.org/10.1038/s41559-019-1052-6

Collingwood Environmental Planning Limited, and IEEP. (2014). *Evaluation of the biodiversity offsetting pilot programme volume 2 (annex): Pilot Reports*. (pp. 1–71).

Defra. (2012). *Technical Paper: The Metric for the Biodiversity Offsetting Pilot in England*. 27.

Defra. (2023). *Statutory biodiversity metric tools and guides*. https://www.gov.uk/government/publications/statutory-biodiversity-metric-tools-and-guides

Dennis, E. B., Freeman, S. N., Brereton, T., & Roy, D. B. (2013). Indexing butterfly abundance whilst accounting for missing counts and variability in seasonal pattern. *Methods in Ecology and Evolution*, *4*, 637–645. https://doi.org/10.1111/2041-210X.12053

Department for Environment, Food & Rural Affairs (Defra). (2017). *Wild bird populations in the UK. All species, species trends, UK*. 1970–2014. http://data.defra.gov.uk/Wildlife/all_species_index_species_trends_UK_1970_to_2014_rev.csv

Department for Environment Food & Rural Affairs, Forestry Commission, Environment Agency, Natural England, and The Rt Hon George

Journal of Applied Ecology BRITISH ECOLOGICAL SOCIETY

Eustice MP. (2021). *World-leading Environment Act becomes law*. https://www.gov.uk/government/news/world-leading-environment-act-becomes-law
eftec, WSP, and ABPmer. (2021). *Biodiversity Net Gain*: *Market analysis study*.
Falk, S. (2021). *A review of the pollinators associated with decaying wood, old trees and tree wounds in Great Britain*. Woodland Trust. https://doi.org/10.13140/RG.2.2.31078.14408
Fox, R., Brereton, T. M., Asher, J., August, T. A., Botham, M. S., Bourn, N. A. D., Cruickshanks, K. L., Bulman, C. R., Ellis, S., Harrower, C. A., Middlebrook, I., Noble, D. G., Powney, G. D., Randle, Z., Warren, M. S., & Roy, D. B. (2015). *The state of the UK's butterflies 2015*. Butterfly Conservation.
Glenister, O. (2022). *Does the biodiversity metric 3.1 discourage the creation of priority habitats?* (pp. 11–15). Bull Chart Inst Ecol Environ Manag.
Hawkins, I., Zu Ermgassen, S. O. S. E., Grub, H., Treweek, J., & Milner-Gulland, E. (2022). No consistent relationship found between biodiversity metric habitat scores and the presence of species of conservation priority. *In Practice: Bulletin of the Institute of Ecology and Environmental Management*, *118*, 16–20.
Hill, M. O., Preston, C. D., & Roy, D. B. (2004). *Plantatt Atributes of British and Irish plants: Status, size, life history, geography and habitats*. Centre for Ecology and Hydrology, 80.
Institute for European Environmental Policy. (2014). Study on specific design elements of biodiversity offsets: Biodiversity metrics and mechanisms for securing long term conservation benefits. https://doi.org/10.2779/010692
Jachuła, J., Denisow, B., & Wrzesień, M. (2021). Habitat heterogeneity helps to mitigate pollinator nectar sugar deficit and discontinuity in an agricultural landscape. *Science of the Total Environment*, *782*, 146909. https://doi.org/10.1016/j.scitotenv.2021.146909
Joint Nature Conservation Committee (JNCC). (2022). *Conservation designations for UK Taxa 2022*. https://hub.jncc.gov.uk/assets/478f7160-967b-4366-acdf-8941fd33850b
Kent County Council. (2022). *Viability Assessment of Biodiversity Net Gain in Kent*.
Maron, M., Ives, C. D., Kujala, H., Bull, J. W., Maseyk, F. J. F., Bekessy, S., Gordon, A., Watson, J. E. M., Lentini, P. E., Gibbons, P., Possingham, H. P., Hobbs, R. J., Keith, D. A., Wintle, B. A., & Evans, M. C. (2016). Taming a wicked problem: Resolving controversies in biodiversity offsetting. *Bioscience*, *66*, 489–498. https://doi.org/10.1093/biosci/biw038
Marshall, C., Wade, K., Kendall, I., Peacock, S., Simkin, J., Bladon, A., Porcher, H., Poffley, J., Treweek, J., & Marrs, R. (2024). Data from: A database of Defra statutory biodiversity metric unit values for terrestrial habitat samples across England, with plant, butterfly and bird species data. *Dryad Digital Repository*. https://doi.org/10.5061/dryad.n5tb2rc0k
Martin, E. A., Dainese, M., Clough, Y., Báldi, A., Bommarco, R., Gagic, V., Garratt, M. P. D., Holzschuh, A., Kleijn, D., Kovács-Hostyánszki, A., Marini, L., Potts, S. G., Smith, H. G., al Hassan, D., Albrecht, M., Andersson, G. K. S., Asís, J. D., Aviron, S., Balzan, M. V., … Steffan-Dewenter, I. (2019). The interplay of landscape composition and configuration: New pathways to manage functional biodiversity and agroecosystem services across Europe. *Ecology Letters*, *22*, 1083–1094. https://doi.org/10.1111/ele.13265
McVittie, A., & Faccioli, M. (2020). Biodiversity and ecosystem services net gain assessment: A comparison of metrics. *Ecosystem Services*, *44*, 101145. https://doi.org/10.1016/j.ecoser.2020.101145
Moilanen, A., Van Teeffelen, A. J. A., Ben-Haim, Y., & Ferrier, S. (2009). How much compensation is enough? A framework for incorporating uncertainty and time discounting when calculating offset ratios for impacted habitat. *Restoration Ecology*, *17*, 470–478. https://doi.org/10.1111/j.1526-100X.2008.00382.x
Morelli, F., Pruscini, F., Santolini, R., Perna, P., Benedetti, Y., & Sisti, D. (2013). Landscape heterogeneity metrics as indicators of bird diversity: Determining the optimal spatial scales in different landscapes. *Ecological Indicators*, *34*, 372–379. https://doi.org/10.1016/j.ecolind.2013.05.021
Musgrove, A., Aebischer, N., Eaton, M., Hearn, R., Newson, S., Noble, D., Parsons, M., Risely, K., & Stroud, D. (2013). Population estimates of birds in Great Britain and the UK. *British Birds*, *106*, 231–232.
Natural England. (2010). *Higher level stewardship farm environment plan (FEP) Manual*.
Oksanen, J., Simpson, G., Blanchet, F., Kindt, R., Legendre, P., Minchin, P., O'Hara, R., Solymos, P., Stevens, M., Szoecs, E., Wagner, H., Barbour, M., Bedward, M., Bolker, B., Borcard, D., Carvalho, G., Chirico, M., De Caceres, M., Durand, S., … Weedon, J. (2022). *vegan: Community ecology package*. R package version 2.6–2. 10, 631–637. ISBN 0-387-95457-0.
Panks, S., White, N., Newsome, A., Nash, M., Potter, J., Heydon, M., Mayhew, E., Alvarez, M., Russell, T., Cashon, C., Goddard, F., Scott, S. J., Heaver, M., Scott, S. H., Treweek, J., Butcher, B., & Stone, D. (2022). *Biodiversity metric 3.1: Auditing and accounting for biodiversity—User guide*. Natural England.
Parkes, D., Newell, G., & Cheal, D. (2003). Assessing the quality of native vegetation: The "habitat hectares" approach. *Ecological Management and Restoration*, *4*, 29–38. https://doi.org/10.1046/j.1442-8903.4.s.4.x
Pedley, S. M., & Dolman, P. M. (2020). Arthropod traits and assemblages differ between core patches, transient stepping-stones and landscape corridors. *Landscape Ecology*, *35*, 937–952. https://doi.org/10.1007/s10980-020-00991-0
Pilgrim, J. D., Brownlie, S., Ekstrom, J. M. M., Gardner, T. A., von Hase, A., ten Kate, K., Kate, K., Savy, C. E., Stephens, R. T. T., Temple, H. J., Treweek, J., Ussher, G. T., & Ward, G. (2013). A process for assessing the offsetability of biodiversity impacts. *Conservation Letters*, *6*, 376–384. https://doi.org/10.1111/conl.12002
Pinheiro, J., & Bates, D. (2022). *nlme: Linear and nonlinear mixed effects models*. R package version 3.1-160.
Quétier, F., & Lavorel, S. (2011). Assessing ecological equivalence in biodiversity offset schemes: Key issues and solutions. *Biological Conservation*, *144*, 2991–2999. https://doi.org/10.1016/j.biocon.2011.09.002
Quétier, F., Regnery, B., & Levrel, H. (2014). No net loss of biodiversity or paper offsets? A critical review of the French no net loss policy. *Environmental Science & Policy*, *38*, 120–131. https://doi.org/10.1016/j.envsci.2013.11.009
R Core Team. (2022). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing.
Robertson, M. (2021). The state of no net loss/net gain and biodiversity offsetting policy in English local planning authorities: Full report.
Simmonds, J. S., von Hase, A., Quétier, F., Brownlie, S., Maron, M., Possingham, H. P., Souquet, M., zu Ermgassen, S. O. S. E., ten Kate, K., Costa, H. M., & Sonter, L. J. (2022). Aligning ecological compensation policies with the Post-2020 global biodiversity framework to achieve real net gain in biodiversity. *Conservation Science and Practice*, *4*, 1–9. https://doi.org/10.1111/csp2.12634
Tew, N. E., Memmott, J., Vaughan, I. P., Bird, S., Stone, G. N., Potts, S. G., & Baldock, K. C. R. (2021). Quantifying nectar production by flowering plants in urban and rural landscapes. *Journal of Ecology*, *109*, 1747–1757. https://doi.org/10.1111/1365-2745.13598
Treweek, J. (2009). *Scoping study for the design and use of biodiversy offset in an English context*. Final Report to Defra.
Treweek, J., Butcher, B., & Temple, H. (2010). *Biodiversity offests: Possible methods for measuring biodiversity losses and gains for use in the UK* (Vol. *69*, pp. 29–32). Pract Bull Inst Ecol Environ Manag.
UKHab Ltd. (2023). *UK Habitat Classification Version 2.0*. https://ukhab.org/eula-v2-03-2023/
Wende, W., Herberg, A., & Herzberg, A. (2005). Mitigation banking and compensation pools: Improving the effectiveness of impact mitigation regulation in project planning procedures. *Impact Assessment and Project Appraisal*, *23*, 101–111. https://doi.org/10.3152/147154605781765652

Wilson, R. (2021). *Are we delivering biodiversity net gain? Do broad habitat metrics mask biodiversity net loss and can a focus on invertebrates help?* (Vol. *113*, pp. 36–41). Bull Chart Inst Ecol Environ Manag.

Zu Ermgassen, S. O. S. E., Baker, J., Griffiths, R. A., Strange, N., Struebig, M. J., & Bull, J. W. (2019). The ecological outcomes of biodiversity offsets under "no net loss" policies: A global review. *Conservation Letters*, *12*, 1–17. https://doi.org/10.1111/conl.12664

Zu Ermgassen, S. O. S. E., Marsh, S., Ryland, K., Church, E., Marsh, R., & Bull, J. W. (2021). Exploring the ecological outcomes of mandatory biodiversity net gain using evidence from early-adopter jurisdictions in England. *Conservation Letters*, *14*, 1–10. https://doi.org/10.1111/conl.12820

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Table S1. Habitats, survey dates and citations for the site datasets included in analysis.

How to cite this article: Marshall, C. A. M., Wade, K., Kendall, I. S., Porcher, H., Poffley, J., Bladon, A. J., Dicks, L. V., & Treweek, J. (2024). England's statutory biodiversity metric enhances plant, but not bird nor butterfly, biodiversity. *Journal of Applied Ecology*, *00*, 1–14. https://doi.org/10.1111/1365-2664.14697