Received: 29 August 2023 | Accepted: 7 February 2024

DOI: 10.1002/pan3.10626

PEOPLE AND NATURE BRITISH ECOLOGICAL SOCIETY

RESEARCH ARTICLE

Valuing the economic benefits of species recovery programmes

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Handling Editor: Masashi Soga

Abstract

1. Accounting for the values placed on nature by the public is key to successful policies in reversing ongoing biodiversity declines. However, biodiversity values are rarely included in policy decisions, resulting in poorer outcomes for people and nature.
2. Our paper addresses an important evidence gap related to the non-availability of values for appraising large-scale policies and investment programmes for species recovery and habitat improvement at the national level.
3. We use a stated preference choice modelling approach to estimate household preferences and Willingness to Pay for species recovery and habitat improvement over a wide range of habitats in England.
4. The framing of our stated preference study is crucial to the evidence we develop. Within the study, we define species recovery as incremental improvements to habitat quality and present respondents with choices between conservation policy options that improve different habitat types. We then use the response data to estimate values for habitat quality improvements, and the associated improvements to species presence and abundance. We are thus able to estimate economic benefits for 'wild species recovery' simultaneously across a wide range of habitat types.
5. Willingness to pay values for habitat improvement was found to be highest for improvements from 'moderate' to 'full' species recovery by 2042; and for habitat types which have relatively low current extents in England, such as lowland fens.
6. *Policy Implications*: biodiversity policy designers can make use of stated preference methods to guide decisions over which aspects of biodiversity targets to focus more resources on, since this enables policy to reflect public preferences, and thus engages higher public support for conservation. In our specific data and context, this implies prioritising the restoration of species recovery to high levels and focussing resources on scarcer rather than more abundant habitat types.

KEYWORDS
biodiversity values, choice experiments, conservation policy, stated preferences

1 | INTRODUCTION

Despite ongoing conservation efforts from local to global scales, biodiversity continues to decline at an accelerating rate, severely impacting on people's well-being and livelihoods (IPBES, 2019). The causes of the current global biodiversity crisis, and the opportunities to address it, have been linked to the way in which nature is valued in political and economic decisions (IPBES, 2022). Numerous global initiatives argue that a greater recognition of nature's values in policy decisions can help attain more sustainable outcomes that benefit both people and nature (Dasgupta, 2021; IPBES, 2022; TEEB, 2010). At the international level, the Convention on Biological Diversity recommends that biodiversity values be fully integrated into policies at all levels of government (UNEP, 2022). However, evidence from the IPBES (2022) 'Values Assessment' suggests that less than 5% of published valuation studies report uptake in policy decisions. It thus appears that the CBD recommendation is far from being realised.

In the United Kingdom, the most recent State of Nature Report indicates that between 1970 and 2016 there was a 13% decline in average species abundance, based on a composite indicator for almost 700 terrestrial and freshwater species (Hayhow et al., 2019). In addition to this net loss, some 15% of almost 80,000 UK species are classified as threatened and at risk of extinction when assessed against Red List criteria. In the absence of new policy interventions (and effective implementation of the Kumming-Montreal agreement: UNEP, 2022), it is likely that the overall trend in declining species abundance will continue, and extinction risk will rise for more species, driven by the multiple pressures from anthropogenic climate change, intensification of land use, pollution, urbanisation, hydrological change and invasive species. In response to these threats, the UK Environment Act 2021 created targets to halt the decline in species abundance by 2030; increase species abundance by at least 10% by 2042 relative to 2030 levels; improve the Red List Index for species extinction risk by 2042 and create or restore in excess of 500,000 hectares of a range of wildlife-rich habitats outside protected sites by 2042. However, these ambitious conservation targets are likely to be challenging to achieve, particularly for conservation agencies that have increasingly tight budgets under the current financial crisis. The implication of this is that these agencies will need to: (i) increasingly justify their expenditures on conservation actions and (ii) make difficult choices relating to how resources are allocated between different conservation actions. Economic valuation methods provide one means to address these twin policy needs, first by enabling the analyst to show how conservation can contribute to higher social welfare through generating economic values; and second by evaluating the preferences of the general public for *how* conservation is undertaken. This provides a means of targeting public spending at those aspects of biodiversity most valued by taxpayers, likely leading to wider acceptability by stakeholders of conservation policy choices. In this paper, we report a new study to estimate the preferences of the English general public for alternative designs of biodiversity policies. Alternative designs are described using a set of attributes which we argue are general to biodiversity conservation policy choices in many spatial settings worldwide.

There is already a large body of stated preference research on the benefits of biodiversity conservation (see the review in Hanley & Perrings, 2019). Christie et al. (2006) were one of the first to attempt to measure the direct economic value of the characteristics of biodiversity. They showed that members of the UK general public had a willingness to pay for conservation policy which depended on (i) whether rare or common species were protected, (ii) whether these species were well-known or unfamiliar to most people, (iii) whether the policy would slow down current rates of loss rather than stopping or reversing this trend and (iv) whether the policies were aimed at habitat restoration versus habitat creation. Studies have since shown that individuals are willing to pay for changes in community-level measures of biodiversity such as species richness (Boeri et al., 2020; Czajkowski et al., 2009; Martínez-Jauregui et al., 2021). Stated preference studies have also shown a willingness to pay to depend on the identities of individual species (Morse-Jones et al., 2012; Richardson & Loomis, 2009); and their conservation status (e.g. scarce versus abundant, or a declining or increasing population trend: Lundhede et al., 2014; for Danish birds; Yao et al., 2019; for endangered native species in New Zealand).

Individuals may also care about the habitats in which species are found, for example having preferences for specific types of forest relative to other forest types. Moreover, there is evidence that the marginal value of protecting additional species declines as more species are protected (Jacobsen et al., 2008), whilst values for native salmon recovery programmes in the Pacific Northwest have been found to depend on the extent of avoided loss, and how quickly avoided losses are realised (Lewis et al., 2019). Work has shown that distance in which people live from the habitat in focus can impact on the values of moving away from a 'locally extinct' to a 'locally conserved' status (Danley et al., 2021). Studies also show that people care about how biodiversity conservation objectives are achieved (the policy choice), irrespective of the outcome. For example, Hanley et al. (2003) found that peoples' willingness to pay for wild geese conservation in Scotland varied according to whether the shooting was used as part of a management policy. People's values for policies which aim to prevent reductions in biodiversity can also be specific to which sectors are negatively impacted by control measures (e.g. Aaneson et al., 2015 for cold water corals).

However, stated preference research that estimates economic values for improvements across a wide range of habitats or across a whole range of species is rare. This is the evidence gap we seek to fill in the current paper, since policymakers often face choices in allocating conservation resources across a wide range of competing demands. Taking England as a case study, we investigate how the values of biodiversity conservation to people vary across the type of management action (such as the extent of habitat restoration) and across a selection of 11 habitats. Using a stated preference approach, we focus on the direct benefits of biodiversity conservation to people (the utility they potentially gain from enhanced conservation status), rather than on the indirect effects of biodiversity on well-being as mediated through its role in ecosystem functioning. We estimate transferable economic values for species recovery and habitat improvement at the national level in the specific context of biodiversity conservation policy in England.

A key requirement for our analysis was to be able to estimate direct use and non-use values of biodiversity conservation measures undertaken in England over the next 20 years. Since specific UK policy implementation details are yet to be formulated, and since these are unlikely to be prescriptive in nature, it was necessary to frame these economic values of biodiversity as dependent on generalisable characteristics (attributes) of policy implementation. We argue that these characteristics—which habitats to target, how big an improvement to aim for in each habitat, and how large an area of each habitat type to target—may also characterise the key strategic policy choices for many biodiversity policymakers globally.

2 | METHODS

2.1 | Species and habitat recovery framework

We design a stated preference discrete choice experiment (DCE: Hanley & Czajkowski, 2019) that reflects a number of context-sensitive aspects of species recovery outcomes, providing a basis to account for factors such as habitat type (e.g. woodland and farmland), the extent of species recovery over a 20-year time period and the potential scale of action (e.g. number or proportion of hectares restored).

The framing of a DCE is typically critical to the evidence developed. Here, species recovery is defined as incremental improvements to habitat quality (Westwood et al., 2014). Species and habitats are two sides of the same coin: a more intact habitat will (typically) have greater species presence, overall species abundance and biodiversity. Through this framing, we present respondents with choices between habitat improvement options (i.e. choices across habitat types) and then use the response data to estimate relative values for these habitat improvements. Such habitat values are useful by themselves, as national-level species policies will most likely result in, or be accomplished through, changes to habitat quality, but they can also be translated into values for the associated improvements to species presence and abundance.

Using this approach, we are able to estimate economic benefits for 'wild species recovery' simultaneously across a wide range of habitat types, in the context of habitat quality improvement and area conserved. Table 1 shows the habitat types used in the experimental design, along with their approximate current extent in the United Kingdom. These habitats were chosen as a representative selection of the 'broad habitat' types that appear in the UK Biodiversity Action Plan Habitats (JNCC, 2019). The habitats are therefore indicative of the range of habitats that might be improved in future UK biodiversity policies.

Initial survey development considered different ways of defining species recovery, including separately defining species abundance, species diversity and extinction risk as independent outcomes. However, using this set of outcomes as attributes within the choice experiment would be difficult since, from an ecological perspective, it is not credible to specify general scenarios in which abundance and diversity move independently in opposite directions. Instead, the approach taken defined species recovery in relation to a gradient of habitat intactness, which reflects species presence, richness and abundance along a spectrum from a degraded or converted habitat (e.g. intensive agricultural land use such as arable production) to

TABLE 1 Habitat types used in the experimental design, their approximate current extents within England and the relevant 'Broad habitat'.

Habitat type	Approx. Extent (ha) (source)	Broad habitat (defined in JNCC, 2019)
Wood pasture and parkland	172,000 ha (Natural England, 2021)	Broadleaved, mixed and yew woodland
Native mixed deciduous woodland	750,000 ha (Forestry Commission, 2020)	Broadleaved, mixed and yew woodland
Upland oakwood	44,000 ha (Forestry Commission, 2020)	Broadleaved, mixed and yew woodland
Arable land	500,000 ha (Defra, 2021)	Arable land
Lowland hay meadows	23,000 ha (Natural England, 2022)	Neutral grassland
Semi-natural grassland	550,000 ha (UK CEH, 2015)	Calcareous grassland and Acid grassland
Heathland	270,000 ha (UK CEH, 2015)	Dwarf shrub and heath
Lowland fens	20,000 ha (Natural England, 2022)	Fen marsh and swamp
Blanket bog	235,000 ha (Natural England, 2022)	Bogs
Rivers	29,000 ha (Countryside Survey, 2007)	Rivers and streams
Coastal sand dunes	20,000 ha (UK CEH, 2015)	Supralittoral sediment

partially intact and then fully intact. Species conservation or habitat restoration outcomes are therefore defined within our DCE by a positive movement along the gradient towards a more intact habitat. Figure 1 shows the information provided to respondents on 'wild species' recovery along this gradient.

The profile of species for each habitat at each level of recovery was specified based on a review of available evidence, including an index of ecological status developed specifically for Great Britain based on plant and animal observations on a 10 km^2 grid (Dyer et al., 2017). These observations feature species occurrence records from national invertebrate, bird and vascular plant monitoring schemes, and include threatened and non-threatened species. These species are considered good indicators of ecological change and the data from many of these surveys are used to inform UK biodiversity policy targets (JNCC, 2019). Using this ecological status index and the locations of selected intact habitats (Rowland et al., 2017) the impact of degradation of intact natural and semi-natural habitats was described in terms of species presence.

2.2 | Choice experiment design

The resultant attributes to be included within the DCE reflected the species recovery context explained to respondents. Table 2 shows these attributes and their levels:

The attributes describe:

- The exten of 'species recovery' along the gradient set out in Figure 1.
- The 'area of the habitat' over which species recovery would occur relative to the current extent of that habitat in England (this was also shown to respondents in terms of total hectares).
- The size of 'sites targeted' for recovery (from small to large): this attribute was conveyed as implying a potential trade-off between 'large sites' with more ecological integrity which might be further away from the respondent's home, and a greater number of 'small sites' which were more likely to be near to where the respondent lives.
- The 'cost' per UK household of each policy option over the period 2023–2042, specified as an increase in the cost of living attributable to implementing the specific recovery plan shown on that choice card for that specific habitat. A budget reminder was shown to respondents before they made their choices.

A three-way choice card was used which included two 'new policy' options at a positive cost to the respondent, and a business-as-usual option with no change in species recovery actions and no additional cost to the respondent (it would not be desirable to force people into paying for conservation if they do not want that; Johnston et al., 2017). Respondents were reminded, however, that this business-as-usual option would, if followed, lead to the continued decline in species. Each respondent made choices over four biodiversity conservation policy options (i.e. completed four choice cards) for each of three different habitat types which were randomly selected for each person from the set of 11 possible habitat types (Table 1). Thus, overall, each respondent completed 12 choice cards. Extensive focus group testing and one-on-one interviews showed that respondents could understand the information provided for three habitat types, but that increasing the number of habitat types above this created problems of comprehension and understanding. Figure 2 shows an example choice card.

The choice tasks exercise was followed by a comprehensive set of questions that probed respondent's reasons for their choices and for potential issues related to their understanding of these choice tasks. These follow-up questions included: the ease/difficulty of the choice questions; the importance of each attribute to the choice made, to determine if any aspect of the choice scenario was typically ignored; motivations for choices in terms of use and/or non-use values (Colombo et al., 2013); reasons for a serial selection of the 'no change' option—to identify potential protest responses versus genuine reasons for not wanting to pay for species recovery outcomes; and the perceived policy consequentiality (that responses would influence actions taken to protect and improve nature and wildlife) and payment consequentiality (that responses would result in changes to what households pay for species recovery actions).

2.3 | Survey testing and data collection

We tested the wording of the survey using an extensive programme of qualitative survey instrument testing, in 30 one-to-one interviews and focus groups over 6 waves and iterations of survey development. A pilot sample of 250 respondents was collected in February 2022 to allow improvements in the statistical efficiency of the final survey choice cards. The main survey was undertaken in early summer 2022 and was implemented online. This approach to collecting DCE data is now well-accepted in the literature (Johnston et al., 2017). The main sampling requirements were a nationally representative sample of households in England based on age, gender and socio-economic group (SEG). Sampling quotas were specified according to national statistics. A large overall sample size was specified (4659 respondents) to accommodate a survey design with multiple habitats. Online panels were used to recruit respondents, but potential participants were not shown the topic, client or any introduction to survey specifics before agreeing to participate. Responses collection was facilitated by Watermelon, a registered Market Research Society (MRS) member. All data collection adhered to the MRS Code of Conduct, which specifies rules and good practices for ethical research, including data protection. All responses were given with informed consent to the purposes for which they would be used, and individuals in the sample all provided written consent to participate in the survey.

2.4 | Econometric analysis

We seek to understand (i) what values people place on the different aspects (attributes) of biodiversity policy in England reflected in the

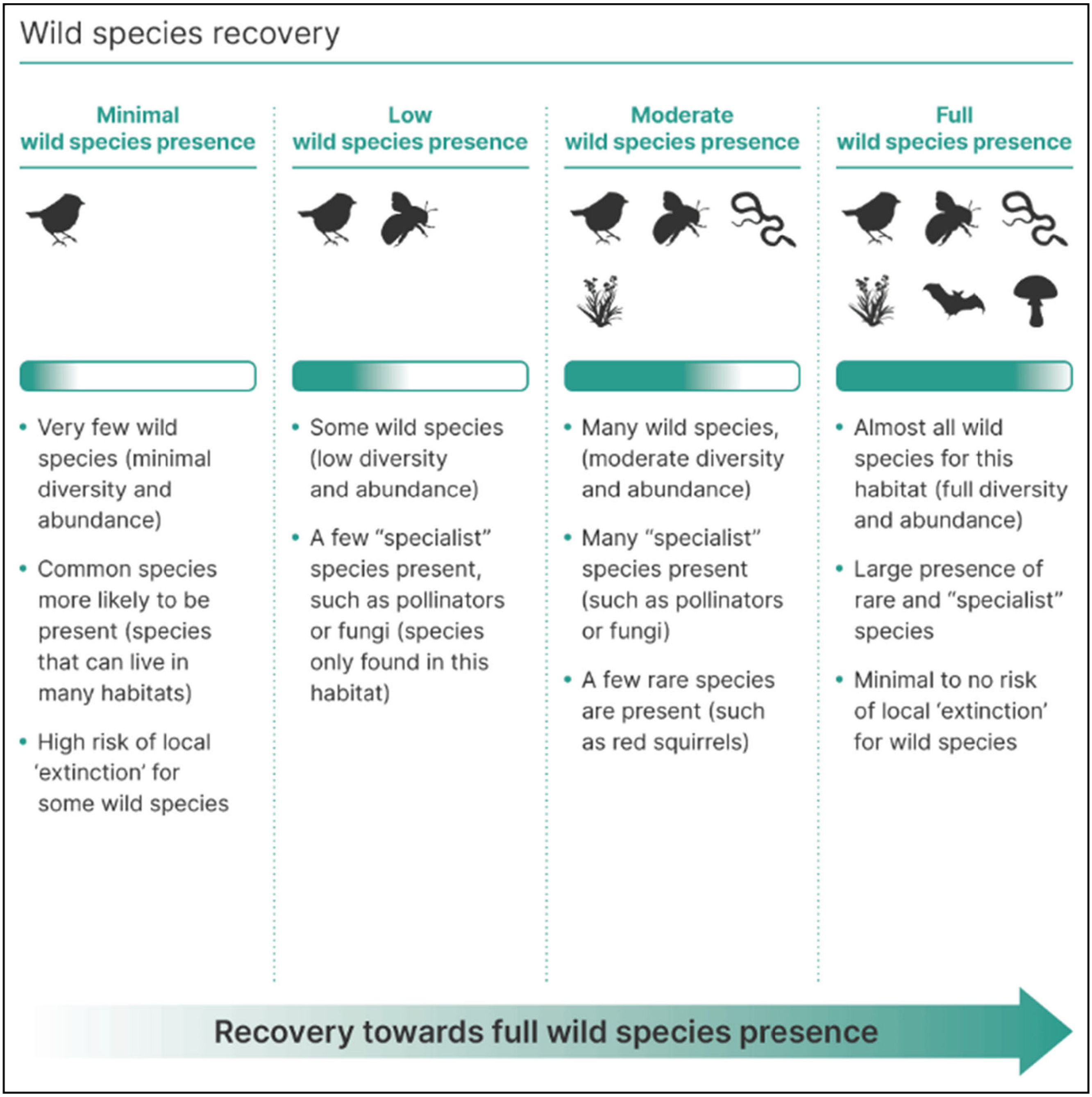


FIGURE 1 Wild species recovery spectrum. This graphic demonstrates how wild species recovery in the general case was presented to respondents. On one end of the spectrum, minimal wild species presence would result in narrow diversity of mostly common species being present and high risks of local extinction for some wild species, while on the other full wild species presence would result in a wide diversity of wild species and low risk of local extinction.

choice cards (ii) what this implies about their willingness to pay for changes in any of these attributes (e.g. from a 'low' to 'moderate' level of wildlife presence) and (iii) how these values compare across the habitat types included in the design. Obtaining this information requires us to estimate a probabilistic choice model which predicts the likelihood that an individual will choose a specific policy option, as a function of the attribute levels within this option relative to all other attributes and levels. The parameters recovered from such a choice model can be interpreted as showing both relative values (how the recovery of one habitat type is valued by people compared to another habitat type) and people's willingness to pay for this preferred option.

For modelling choices and to recover estimates of willingness to pay for changes in each attribute, we base our approach on random utility theory (McFadden, 1974). In this model, the utility of the individual i resulting from choosing alternative j in situation t can be expressed as:

$$V_{ijt} = ac_{ijt} + \mathbf{bX}_{ijt} + e_{ijt}, \quad (1)$$

TABLE 2 Choice task attributes and levels used in the experimental design of choice cards.

Attribute	Description	Levels
Species recovery	Species recovery by 2042	• From Minimal to low wild species presence • From Low to moderate wild species presence • From Moderate to full wild species presence
Area of habitat	Amount of habitat improved	• 2.5% of total in England • 5% of total in England • 10% (or 12.5%) of total in England • 15% (or 25%) of total in England • 20% (or 37.5%) of total in England • 25% (or 50%) of total in England
Sites targeted	Sites targeted by recovery actions	• Small sites • Medium sites • Large sites
Cost	Increase in household expenditure due to recovery actions (amount from 2023 to 2042)	• £5 per year • £10 per year • £25 per year • £50 per year • £75 per year • £100 per year • £150 per year • £250 per year • £500 per year

Note: Native mixed deciduous woodland, Semi-natural grassland and Arable land (organic) habitats used a scale of 2.5%–25%. All others used a scale of 2.5% to 50%. The table shows the attributes used in the choice experiment and the levels each attribute could take in the choice cards.

where the utility expression is assumed additively separable in the cost of the alternative, c_{ijt}, and other attributes, $\boldsymbol{X}_{\text{ijt}}$; a and $\boldsymbol{b}$ denote the corresponding parameters; and e_{ijt} is a stochastic component allowing for factors not observed by the econometrician to affect individuals' utility and choices. Whilst the researcher does not observe e_{ijt}, they are able make assumptions about its distribution. Depending on this assumption, the model can be transformed into different classes of choice models. Assuming that the stochastic component e_{ijt} follows an independent and identically distributed extreme value (type I) distribution, this leads to the logit probability specification, used in simple conditional logistic regressions, with a probability of choosing alternative j from a set of J available alternatives:

$$P_{\text{ijt}}(a, \boldsymbol{b}) = \frac{\exp\left(ac_{\text{ijt}} + \mathbf{bX}_{\text{ijt}}\right)}{\sum_{k=1}^{J} \exp\left(ac_{\text{ikt}} + \mathbf{bX}_{\text{ikt}}\right)}. \tag{2}$$

Given that we are interested in deriving willingness to pay values from choices, based on respondents willingness to trade off increases in any of the biodiversity attributes against increases in the monetary attribute c_{ijt}, it is convenient to introduce the following modification of (1), which is equivalent to using a money-metric utility function (in our case, it means estimating the parameters in WTP space; Train & Weeks, 2005, Scarpa et al., 2008):

$$U_{\text{ijt}} = \sigma a\left(c_{\text{ijt}} + \sigma b\boldsymbol{X}_{\text{ijt}}\right) + \varepsilon_{\text{ijt}} = \lambda\left(c_{\text{ijt}} + \beta\boldsymbol{X}_{\text{ijt}}\right) + \varepsilon_{\text{ijt}} \tag{3}$$

In this specification, by rescaling the utility function, the vector of parameters, $\beta = b / a$ can be directly interpreted as a vector of the implicit prices (marginal WTPs) for the non-monetary attributes, $\boldsymbol{X}_{\text{ijt}}$, facilitating an interpretation of the results.

An inconvenient assumption of this simple (multinomial logit) model is the independence and identical distribution of the error term for all of the alternatives and respondents, as well as identical preferences of different respondents—the same coefficients λ and $\boldsymbol{\beta}$ enter the utility function for all individuals. One way of relaxing this assumption—that is, allowing for some level of unobserved preference heterogeneity and, possibly, correlations between the alternatives and choice tasks—is to include consumer-specific parameters, λ_i, $\boldsymbol{\beta}_i$, which leads to a Mixed Logit Model (MXL). A commonly used approach is to make mixing distributions continuous. If individual parameters are assumed continuously distributed following a parametric distribution specified a priori by a modeller, $\left[\lambda_i, \boldsymbol{\beta}_i\right] \sim f\left(\overline{\boldsymbol{\beta}}, \boldsymbol{\Sigma}\right)$, with means, $\overline{\boldsymbol{\beta}}$, and variance–covariance matrix, $\boldsymbol{\Sigma}$, the random parameters mixed logit model is formed (RP-MXL, Hensher & Greene, 2003). In MXL, the probability of making given choices in a set of T situations, is a weighted average of standard logit probabilities and it can be presented as:

$$P_i(\theta) = \int\left(\prod_t \sum_j I_{\text{ijt}} P_{\text{ijt}}([\lambda, \boldsymbol{\beta}])\right) f([\lambda, \boldsymbol{\beta}]|\theta) d[\lambda, \boldsymbol{\beta}], \tag{4}$$

where I_{ijt} equals 1 if individual i has chosen alternative j, and it equals 0 otherwise. The utility function for respondents is analogical to an MNL model, except for the fact that the vector of the parameters $\left[\lambda_i, \boldsymbol{\beta}_i\right]$ can vary for different respondents.

The model is estimated using the maximum likelihood method for the utility function parameters, conditional on individuals' observed choices and attribute levels associated with choice alternatives. Estimating the MXL model requires the use of simulation

FIGURE 2 Example choice card. This Figure shows one of the sequence of choices presented to respondents, in this instance for potential changes to upland oakwood habitats, with indications of how varying levels of recovery in ecological quality would result in changes to the number of wild species present. Respondents were asked to choose one of the three policy options in each choice card.

methods because the integral in (4) does not have a closed form. We can thus apply a simulation procedure in which $[\lambda_r, \beta_r]$ is drawn from $f([\lambda_r, \beta_r] \mid \theta)$ and, for each $[\lambda_r, \beta_r]$the logit formula is calculated. The simulated probability is given by the average over R draws:

$$\hat{P}_i(\theta) = \frac{1}{R} \sum_{r=1}^{R} \left(\prod_t \sum_j I_{ijt} P_{ijt}([\lambda_r, \beta_r]) \right) \quad (5)$$

$\hat{P}_i(\theta)$ is an unbiased estimator of $P_i(\theta)$ by construction. The simulated probabilities can then be used in a log-likelihood function (McFadden & Train, 2000). In the simulation, we used 10,000 scrambled Sobol draws (Czajkowski & Budziński, 2019).

We assumed that all attribute parameters within the choice model varied across respondents and were normally distributed, with the exception of the (negative) cost coefficient which was assumed to be log-normally distributed to ensure that people were not assumed to prefer more expensive conservation options to less expensive options with equal outcomes. Sample weights were applied to address the over/under-representation of socio-economic groups in the sample. Since we use a WTP-space model, the coefficients for each attribute/level can be interpreted as representing annual WTP per household per year (in GBP).

3 | RESULTS

We found that a majority (59%) of respondents agreed with the statement that 'taking action to restore nature and improve areas for wildlife should be one of the highest priorities for Government'. In response to a question on the preferred actions to achieve species recovery outcomes, 31% of respondents felt that restoring natural habitats for wildlife or improving degraded habitats should be the highest priority. Respondents' preferences for species recovery policies are presented in Table 3. This is our baseline model for comparing general preferences and WTP for policies: (1) improving sites with minimal, low or moderate wild species presence, represented by three dummy variables with the status quo/no change used as a reference, (2) targeting small, medium or large sites (two dummy variables, with small sites used as a reference level) and (3) the importance of the percentagewise amount of habitats improved (continuous variable). The model is estimated using all choice observations. It does not take habitat interactions into account.

As may be seen, mean willingness to pay is £458/household/year for improving sites with 'moderate' wildlife presence to 'full' wildlife presence, £327 for improving 'low' wildlife presence sites to 'moderate' levels and £169 for improving sites with 'minimal' wildlife presence to 'low' levels. Overall, these results stress the higher value placed by respondents on the 'full recovery' of habitats, resulting in the restoration of species abundance to the most ecologically-intact states. It must be noted, however, that the estimates of the standard deviation parameters (final column of Table 3) show that there is very considerable heterogeneity in peoples' preferences towards species recovery. In other words, respondents varied greatly in the value they placed on each change in species recovery specified in the design.

Table 3 also shows that respondents preferred actions to be targeted at 'large' sites rather than 'medium' size sites. The information provided in the survey indicated that large sites may result in better overall species recovery outcomes than small sites, and therefore this finding may reflect respondents considering the potential efficacy of the proposed species recovery options presented. It must be noted, however, that the WTP values associated with the change from small to medium sites (£6.62/household/year) or to large sites (£15.01/year), are an order of magnitude lower than WTP associated with specific recovery targets. Finally, we find that respondents prefer that the recovery occurs on larger areas of each habitat: on average, recovering each additional 10% of a habitat was an equivalent of an additional WTP of £19.54/household/year.

Also of interest is how WTP varies for the same policy management variable across the 11 different habitat types used in the experimental design. To do this, we estimated a second model, in which the means of all random parameters are interacted with 10 habitat types (with arable land used as the baseline). The model is estimated using all choice observations. Table 4 presents the results, and shows WTP for attribute changes for all policy attributes to be habitat-specific (calculated as a sum of the main effect and interaction effect for each habitat). Given our interest in WTP for specific areas of each habitat recovered, the 'amount of habitat improved' attribute is now expressed in 1000 ha; otherwise, the specification of the model is the same as of our baseline model. Figure 3 shows how mean WTP for an area of 1000 ha recovered varies across these different habitat types. It can be seen that the highest mean WTP values for recovery are placed on those habitats with currently have the lowest extent in England (lowland fens, lowland hay meadows, upland oakwood, rivers and coastal sand dunes), compared with willingness to pay for improving habitat types which are more abundant. This result is possibly indicative of the effect of relative scarcity on people's values. Finally, Figure 4 shows how total WTP for an indicative habitat changes as the percentage of the current extent of that habitat type is improved to good status: as may be seen, marginal WTP is falling as the amount improved increases.

Figure 3 shows values in £ per household per year on the vertical axis. Height of solid bars shows mean WTP for each habitat type of an additional 1000 ha. of national recovery. In this study, 95% confidence intervals are also shown for these mean values.

4 | DISCUSSION

4.1 | Contributions of the paper

Our study investigates a set of policy questions which are important for any policy designer worldwide concerned with biodiversity conservation. These are (1) which habitats to prioritise for action; (2) what degree of species recovery to aim for; (3) how best to plan

TABLE 3 Willingness to pay (£/HH/year) for environmental recovery attributes—the results of the random parameters mixed logit (MXL) model.

Attribute	Level/measure	Mean WTP (£/HH/year) (st. er.)	St. dev. (st. er.)
Species recovery by 2042	Minimal to low	169.12*** (5.36)	202.51*** (5.34)
	Low to moderate	327.37*** (5.71)	207.73*** (4.09)
	Moderate to full	458.33*** (6.82)	299.95*** (5.05)
Sites targeted by recovery actions	Small sites (reference)	—	—
	Medium sites	6.62*** (2.47)	6.82 (4.57)
	Large sites	15.01*** (3.04)	74.93*** (4.79)
Amount of habitat improved	Per 1%	1.95*** (0.09)	3.34*** (0.15)
-Cost	£/year	−33.37*** (2.45)	113.27*** (2.89)
Model fit	LL at convergence	−44703.58	
	LL at constant(s) only	−55442.18	
	McFadden's pseudo-R^2	0.1937	
	Ben-Akiva-Lerman's pseudo-R^2	0.4777	
	AIC/n	1.5997	
	BIC/n	1.6019	
	n (observations)	55,908	
	r (respondents)	4659	
	k (parameters)	14	

Note: *, **, *** represent statistical significance at 0.1, 0.05 and 0.01 level, respectively. Standard errors in parentheses. For log-normally distributed parameter (-Cost) the mean and standard deviation of the underlying normal distribution are provided.

TABLE 4 Mean willingness to pay for policy attributes (habitat-specific results in 100 GBP/year).

Attribute	Arable land	Blanket bog	Heathland	Lowland fens	Lowland hay meadows	Native mixed deciduous woodland	Semi-natural grassland	Upland oakwood	Wood pasture and parkland	Rivers	Coastal sand dunes
Minimal to low	1.64*** (0.14)	1.34*** (0.24)	1.60*** (0.24)	1.84*** (0.24)	1.72*** (0.24)	1.35*** (0.25)	1.29*** (0.24)	1.46*** (0.23)	1.22*** (0.24)	1.51*** (0.24)	1.02*** (0.26)
Low to moderate	2.95*** (0.13)	2.72*** (0.21)	3.15*** (0.21)	3.18*** (0.20)	3.17*** (0.21)	3.13*** (0.22)	2.97*** (0.21)	2.95*** (0.20)	3.34*** (0.22)	3.05*** (0.21)	2.78*** (0.22)
Moderate to full	4.19*** (0.13)	4.11*** (0.21)	4.20*** (0.21)	4.49*** (0.20)	4.34*** (0.20)	4.22*** (0.21)	4.21*** (0.21)	3.97*** (0.20)	4.10*** (0.21)	4.46*** (0.22)	4.14*** (0.22)
Medium sites (vs. small)	0.04 (0.07)	0.28** (0.13)	−0.02 (0.11)	−0.08 (0.12)	0.00 (0.11)	−0.07 (0.11)	0.13 (0.11)	−0.08 (0.10)	−0.05 (0.12)	0.56*** (0.15)	0.78*** (0.17)
Large sites (vs. small)	0.11 (0.09)	0.51*** (0.16)	−0.17 (0.13)	−0.09 (0.13)	−0.09 (0.12)	−0.09 (0.13)	−0.01 (0.13)	−0.18 (0.12)	−0.12 (0.13)	0.62*** (0.17)	0.76*** (0.18)
Habitat improved (1000 ha)	0.00*** (0.00)	0.00*** (0.00)	0.00*** (0.00)	0.05*** (0.01)	0.05*** (0.01)	0.00*** (0.00)	0.01*** (0.00)	0.04*** (0.00)	0.01*** (0.00)	0.04*** (0.01)	0.06*** (0.01)

Note: *, **, *** represent statistical significance at 0.1, 0.05 and 0.01 level, respectively, for a test of the null hypothesis that the parameter equals zero, for every attribute level/habitat combination. Standard errors in parentheses.

these improvements in terms of many small versus few large conserved areas and (4) what scale of ambition to set for the overall policy. This set of policy attributes has not been analysed jointly before in terms of the preferences of people living within the geographic area impacted by policy choice; yet national policymakers often need to make such trade-offs between policy attributes, especially when faced with the twin pressures of increased need for actions to protect biodiversity and constrained budgets. Whilst the willingness to pay values we presented are specific to the population of England and the specific conservation scenarios employed in the choice sets, the approach taken is generalisable to other spatial scales and/or locations. This is particularly important where there is a desire and/or need to take public preferences into consideration in setting conservation targets; or, more specifically, if conservation policies are subject to benefit–cost analysis prior to regulatory approval. We thus argue that our paper has general interest.

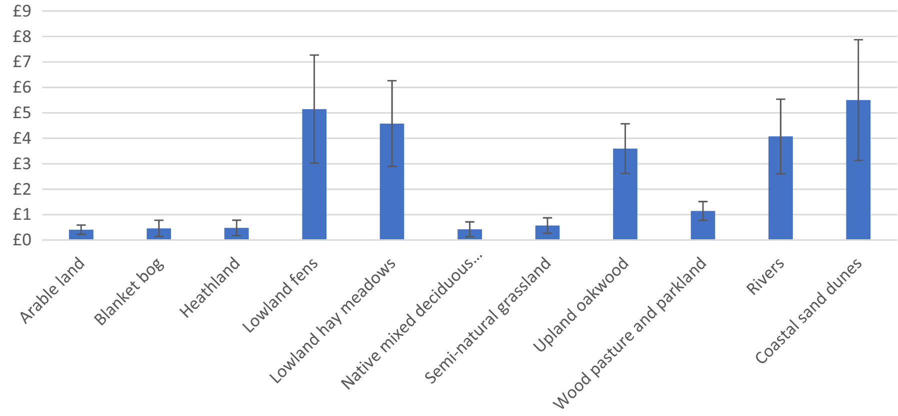
4.2 | Main results

We found that English households placed the highest values on improving habitats to a greater level of intactness and species presence. Across all habitat types, recovery from 'moderate' to 'full' recovery of wild species was valued much more highly than recovery from 'minimal' to 'low' recovery. In terms of relative values across habitats (recall that each respondent made conservation policy choices for a random selection of 3 habitat types out of 11 possible types), we found the highest marginal WTP (per 1000 ha additional area of recovery in 'wild species' presence) was greatest in relatively scarce UK habitat types such as lowland fens, lowland hay meadows and coastal sand dune systems; and was lowest in more abundant habitat types such as arable land and lowland mixed deciduous woodlands. The policy implications are that effort would (a) be better spent in focused efforts on less-common habitats rather than more common habitats and (b) on moving the status of sites from 'moderate' to 'full' wild species recovery rather than from 'minimal' to 'low', so long as the costs of the former are not disproportionately greater than the costs of the latter.

Such public support for costly conservation actions is encouraging in the light of UK commitment to the Kumming-Montreal GBD targets on preventing further decline and then achieving recovery in '..areas of high biodiversity importance, including ecosystems of high ecological integrity' by 2030 and 2050 respectively. However, it is likely that the costs of conservation action will also vary according to variation in the policy attributes we include in the design: thus, a project focussing on improving habitat condition from 'moderate' to 'full' species presence might fail a cost–benefit test, or be less preferable to a project improving quality from 'low' to 'moderate' if the marginal costs of improving habitat condition are increasing sufficiently quickly.

Given that no published valuation study which we are aware of used the same set of conservation attributes as that used in the present study, it is hard to make comparisons between the results reported here and those already available in the literature. However, two comments can be made. First, unlike Jacobsen et al. (2008) we find no

FIGURE 3 Marginal willingness to pay (WTP) for environmental recovery of additional 1000 ha of specific habitat types.



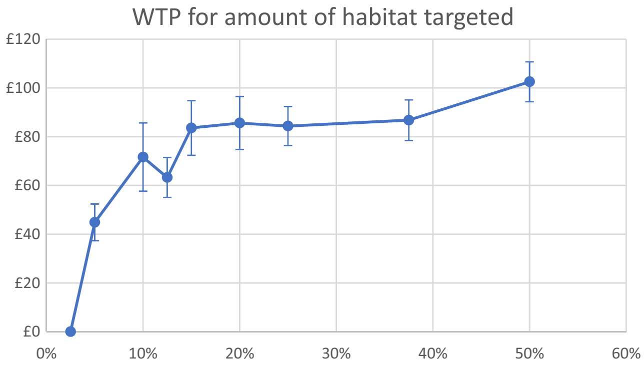


FIGURE 4 Benefit of species recovery (£/household/year). Note: chart is for a single indicative habitat. As the amount of that habitat improved increases, marginal willingness to pay for further improvements in that habitat declines.

evidence of declining marginal values for species recovery (expressed as the number of species present in a specific habitat): values obtained here showed on the contrary that people's WTP was greater for a move from 'moderate' to 'full' recovery than for recovery from 'minimal' to 'low'. Second, evidence was found of marginal values declining as the number of hectares of a given habitat type improved increases. Thus, declining marginal values seem to characterise the amount of habitat protected, rather than the number of species conserved.

4.3 | Challenges for economic valuation of biodiversity using stated preference methods

Robust applications of stated preference methods to biodiversity conservation requires effective communication of the 'valuation problem' which respondent's choices relate to. Clearly, making choices over the design of biodiversity policy is not something which most people will be familiar with. Our study tries to achieve meaningful responses and thus obtain robust estimates of willingness to pay through five features of our experimental design and data collection procedures. First, we used a set of 'warm-up' questions to prompt respondents' into thinking about species decline, and provided key explanatory information about species recovery that supports the discrete choice experiment. Respondents were presented with a concise set of information describing the habitats they would be presented with along with the species recovery outcomes and scale of potential improvements. The format, understandability and context of this information was thoroughly tested before the main survey was implemented. Second, information on the long-term decline in wild species was introduced using excerpts from the most recent State of Nature Report (Hayhow et al., 2019) as an independent evidence source. Respondents were asked if they were aware of these long-term trends and then provided with a summary of the main causes of decline in wild species.

Thirdly, we introduced the concept of species recovery to respondents in terms of the types of actions and measures that could be implemented to reverse this decline. These included changing farming practices, making space for nature in urban areas, habitat restoration or enhancement, reducing pollution, species reintroductions and enforcement of persecution laws that protect wild species. Fourth, we explained to respondents that species recovery actions would be costly to people due to the impacts on household expenditure through higher prices (such as food products), bills (e.g. investments by utility companies contributing to species recovery outcomes) and/or higher taxes (e.g. public funding for public goods). Finally, respondents were told that the purpose of the survey was to understand people's views about the restoration of nature and wildlife to '…help inform decisions by Government, Local Authorities, and other organisations on how they can protect and improve the environment in England over the next 15 to 20 years'. This stressed the outcome consequentiality of the survey, which previous research has shown to be key to respondents being able and willing to reveal their true maximum willingness to pay (Johnston et al., 2017). Whilst it is certainly true that the WTP estimates presented here may well be subject to hypothetical market bias (Johnston et al., 2017), it is the *relative* WTP values between conservation policy attributes (e.g. between habitat types; between different levels of recovery) which are of most interest in this paper. It is not obvious why such relative values should be subject to hypothetical bias, in contrast to absolute WTP values.

5 | CONCLUSIONS

This paper shows how economic values for 'wild species recovery' can be simultaneously estimated across a wide range of habitat types in the context of habitat quality improvement and area conserved. Indeed, a generalisable contribution of the paper is to conceptualise ecological restoration from the perspective of the economic valuation of biodiversity in terms of wild species recovery, whereby higher levels of recovery above a current degraded baseline lead to higher diversity and richness, and a greater presence of rare and/or specialist species. Results from this study confirm both the overall economic importance of biodiversity recovery to the English population, with average households willing to commit significant financial resources to biodiversity recovery; and show which habitats should be made the focus of recovery actions when budgets are tight and when policymakers wish to take account of citizen preferences in making conservation policy choices.

Taking England as a case study, we investigate how WTP varies across type of management action and across a selection of habitats—showing that, for example, the highest marginal WTP values are found for improving the condition of those habitats which are the least abundant in England. We find that people can both understand the habitat improvement scale used and can meaningfully differentiate between habitats to provide considered responses to the choice task. This is reflected in the results which show that respondents place different values on the recovery of different types of habitats, prefer changes that result in more intact habitats (all else equal), and generally do not make choices based on whether recovery occurs close to home or far away. This finding, coupled with responses that probe on household motivations, indicates that a significant portion of the value placed on species recovery comes from non-use motivations, such as bequest or existence values. This result also agrees with previous research (e.g. Christie et al., 2006; Colombo et al., 2013) using stated preference methods for biodiversity valuation in the United Kingdom. As non-use values are impossible to measure using methods other than stated preferences, this provides a strong argument for including stated preference research as part of the evidence base for species recovery policies.

From a wider, international perspective, the paper illustrates a method for setting public-preference-based priorities for biodiversity conservation policy. The findings that the value people place of habitat restoration varies with the extent of restoration from degraded to 'full wild species presence', but also with both the current abundance of the habitat relative to other habitats and with how much of each habitat is improved, may have similarities with other conservation contexts. If this is so, it implies that taxpayer support for biodiversity conservation varies along multiple dimensions of potential improvement.

AUTHOR CONTRIBUTIONS

Study design: Ella Browning, Mike Christie, Mikołaj Czajkowski, Ali Chalak, Russell Drummond, Nick Hanley, Kate Jones, Jake Kuyer, Allan Provins. Project management: Russell Drummond, Jake Kuyer, Allan Provins. Econometric analysis: Mikolaj Czajkowski, Ali Chalak. Paper writing: Nick Hanley, Mikolaj Czajkowski, Mike Christie, Ellie Browning, Russell Drummond, Allan Provins, Kate Jones. Paper revisions: Nick Hanley, Mikolaj Czajkowski, Allan Provins.

ACKNOWLEDGEMENTS

We thank three referees for their comments. Mikołaj Czajkowski gratefully acknowledges the support of the National Science Centre of Poland (Sonata Bis, 2018/30/E/HS4/00388).

FUNDING INFORMATION

This work was funded by the UK Department of Environment, Food and Rural Affairs under the contract 'Valuing the Benefits of Species Recovery in England'.

CONFLICT OF INTEREST STATEMENT

No author reports a conflict of interest.

DATA AVAILABILITY STATEMENT

Data and software codes are available as an Open Science Framework project at https://osf.io/en4d5/; DOI: https://doi.org/10.17605/OSF.IO/EN4D5.

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SUPPORTING INFORMATION

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

Supporting Information S1. Defra—Valuing the Benefits of Species Recovery.

TABLE S1. Characteristics of sample, and equivalent national population statistics.

How to cite this article: Browning, E., Christie, M., Czajkowski, M., Chalak, A., Drummond, R., Hanley, N., Jones, K. E., Kuyer, J., & Provins, A. (2024). Valuing the economic benefits of species recovery programmes. *People and Nature*, *6*, 894–905. https://doi.org/10.1002/pan3.10626