Resource and Energy Economics 85 (2026) 101539

Contents lists available at ScienceDirect

Resource and Energy Economics

journal homepage: www.elsevier.com/locate/ree

The effects of spatial framing and attribute range on the measurement of nonuse values of biodiversity improvements

Kennet Christian Uggeldahl [a,*], Thomas Lundhede [a,b], Jette Bredahl Jacobsen [a], Søren Bøye Olsen [a]

[a] Department of Food and Resource Economics, University of Copenhagen, Rolighedsvej 23, 1958 Frb. C, Denmark
[b] Department of Agricultural Economics, Extension and Rural Development, University of Pretoria, Hatfield, South Africa

ARTICLE INFO

JEL Classification:
Q51
Q57

Keywords:
Biodiversity
Choice experiment
Contextual embedding
Scope sensitivity
Ecological indicator
Stated preferences
Validity

ABSTRACT

Assessing the value of changes in environmental conditions using stated preference valuation studies requires accurate quantification and communication of outcomes that affect human welfare. Using a stated choice experiment to estimate primarily nonuse value of changes in biodiversity per se, i.e., as an inherent characteristic of an ecosystem, we employ a composite metric known as the Biodiversity Intactness Index to capture and communicate the multifaceted nature of biodiversity. However, using complex ecological indices to value abstract concepts might make respondents more susceptible to effects related to the framing of the choice context, thereby raising concerns about validity. Employing a split sample design, we find that value estimates depend on the spatial context in which biodiversity improvements are presented: the larger the spatial scale, the smaller the value. Varying the range of the biodiversity improvement attribute in additional split samples, we find that in two out of the three tested spatial framings, the results are insensitive to the presented attribute range. Respondents thus appear to react to the absolute, rather than the relative, size of the improvements presented. The results from these two spatial framings also exhibit sensitivity to scope, supported by both internal and external scope tests. These findings might alleviate some of the validity concerns associated with employing abstract ecological indices in stated preference valuation studies.

1. Introduction

Biodiversity is a complex and multifaceted concept, encompassing not only the variety of all living organisms but also the intricate interrelationships among them (UN, 1992). Hence, coming up with measures that encompass all its aspects is difficult (Mace, 2005), and a diversity of measures have been suggested and used in natural science (Marshall et al., 2020). Within the environmental valuation literature, simple indicators have most often been used (Strange et al., 2024). It has been noted that these indicators are not measuring biodiversity *per se*, but rather the biological resources that are a manifestation of biodiversity, i.e., the individual species or habitats that are used as indicators (Laurila-Pant et al., 2015; Pearce, 2001). This has been one of the critiques of the derived value estimates (Bartkowski et al., 2015). Recent studies also show that people hold a holistic perception of biodiversity (Bakhtiari et al., 2014; Uggeldahl et al., 2025), further highlighting that the simplistic indicators used in past stated preference studies are unlikely to be appropriate indicators linking the biophysical change to human welfare (see, e.g., Boyd et al. 2016, Johnston et al. 2012 and Schultz

* Corresponding author.
E-mail address: kcu@ifro.ku.dk (K.C. Uggeldahl).

https://doi.org/10.1016/j.reseneeco.2025.101539
Received 21 October 2024; Received in revised form 31 October 2025; Accepted 16 November 2025
Available online 19 November 2025
0928-7655/

et al. 2012 for work on linking indicators), especially when the focus is on the nonuse value of biodiversity (Boyd et al., 2023). Hence, composite, more comprehensive, biodiversity indices need to be used that better capture the underlying value of biodiversity as a good in itself, rather than as an intermediary good that underpins the value of other ecosystem services.

Apart from better representing the underlying good, an additional benefit of using composite ecological indices in valuation is that some of these are, by design, generalizable and transferable across space, which in theory increases the usability of the value estimate for, e.g., benefit transfer. However, in practice the use of composite indices of biodiversity might increase the cognitive burden on the respondents, which might be why past studies have relied on simpler indicators. The increased cognitive burden is likely to make respondents more susceptible to researcher induced effects (Scott, 2002), such as effects related to the framing of the choice context. Framing effects are understood to arise when the presentation format of substantively equivalent information about a choice scenario leads to systematically different choice outcomes (Kahneman, 2002). Such effects have been documented in different fields related to human judgment (Scheufele and Iyengar, 2014). In stated preference research, the term “framing” has sometimes been used to also cover effects related to information provision and changes to the decision context. Faccioli and Glenk (2022) provide a classification of this literature and note that relatively little research exists on how the framing of objectively equivalent descriptions affects stated preferences and welfare analysis.

We explore two effects related to the framing and the scope of biodiversity improvements in stated choice experiments (SCE): the framing of the spatial scale of the good – which we refer to as a “spatial framing effect” – and the effect of attribute scope, or quantity of the good – which we refer to as “attribute range effects”. These two effects are particularly relevant for the generalizability of value estimates obtained using spatially explicit, scalable, indices.

A spatial framing effect might occur when the size of the area over which an index is calculated, and subsequently presented to respondents, affects the elicited values of otherwise equivalent changes in biodiversity. While spatial scope effects, which occur due to changes in the geographic extent or location of the good, have received substantial attention in the literature (e.g., Spencer-Cotton et al., 2018), spatial framing effects have to our knowledge not previously been investigated.

Attribute range effects are observed when value estimates are affected by the range of the attribute levels presented, e.g., when differences in the overall range of biodiversity improvements presented across choice sets lead to systematic differences in elicited values. While attribute range effects have received some attention in the past (Börger et al., 2025; Glenk et al., 2019; Luisetti et al., 2011; Meyerhoff et al., 2015), it has not previously been tested when using complex, composite ecological indices which might be particularly susceptible to such effects. Furthermore, varying attribute range between split samples can also function as an external scope test, which is considered an important part of validity testing (Johnston et al., 2017). We use the difference in attribute range between splits to provide an overall validity check of the estimated values elicited using a composite ecological index, the Biodiversity Intactness Index (BII), to convey information about biodiversity improvements to respondents in our SCE. To our knowledge, this has not been formally done before. We thus also contribute to the relatively scarce literature on biodiversity valuation using composite indices (e.g., Weller and Elsasser 2018), and more generally to the literature on the use of ecological indices in stated preference research (e.g., Vossler et al. 2023, Johnston et al. 2011).

Our results provide compelling evidence for the existence of a spatial framing effect. Presenting changes in the index calculated over a smaller area (making the index change bigger in relative terms) results in a larger willingness to pay (WTP) compared to presenting the equivalent improvement as a change in the index calculated over a larger (national) area. This provides novel insights into the ongoing discourse on the importance of spatial information framing in stated preference surveys, as outlined in, e.g., Glenk et al. (2020). We also find attribute range effects, however only when the biodiversity index is calculated over a larger (national) area. Across the two treatments with only this larger spatial framing, we get different WTPs for improvements of the same size. The results indicate that respondents subjected to this spatial framing react more to the relative, rather than absolute, size of the presented improvements, i.e., that the range of the attribute affects results. This suggests that presenting spatially explicit ecological indices at a smaller geographical scale better captures scope effects, as our results show no attribute range effects here.

The rest of the paper is structured as follows: The next chapter briefly reviews the relevant literature on framing effects related to the scope and scale of the good. Chapter 3 presents the empirical case used in the investigation, Chapter 4 presents the results, and Chapter 5 discusses and concludes.

2. Background

Alternative biodiversity indices can be thought of as different ways of framing specific ecological improvements, where the choice of index might impact how the ecological outcome is valued. Existing literature already provides evidence of framing effects in biodiversity valuation. For instance, Jacobsen et al. (2008) find large differences in biodiversity value estimates when describing the impact in terms of protected named species compared to a quantitative number of protected unnamed species. Respondents in environmental stated preference studies may be more susceptible to such researcher-induced effects, e.g., related to the framing of the choice context, due to their typically limited prior knowledge and experience with the good being valued and the valuation scenario (Glenk et al., 2019; Scott, 2002).

This paper extends previous research on framing effects across different indices or indicators, by investigating potential framing effects related to using the Biodiversity Intactness Index. Specifically, we focus on framing effects caused by differences in the spatial scale, i.e., the size of the area over which the index is calculated. In the case of a biodiversity index, the impact of a proposed new project to increase protection in a specific nature area could be calculated at various spatial scales. For instance, the resulting change in the biodiversity index could be calculated at either (a) the specific nature area considered, (b) the region in which the nature area is located, or (c) the country in which the nature area is located. It follows that the calculated change in the index will be relatively larger when using a smaller base area, i.e., in scenario (a), and result in a relatively higher numerical change in the index than if using option (c) – even though the actual project and the change in biodiversity is the same. The potential effect of spatial framing can thus arise because of the change in the spatial context, the corresponding change in the numerical presentation of the improvement this implies, or as a combination of both. This spatial framing effect is related to spatial scope effects as discussed by Spencer-Cotton et al. (2018). Spatial scope effects occur due to actual changes in the geographical context of the good, such as different regions, or changes in the geographic extent of the good, such as local, regional or national. Spatial scope effects are often confounded with "attribute scope" effects, i.e., changes in the quality/quantity of the good, as "*increasing the geographic scale may also increase the amount of the good*" (Spencer-Cotton et al., 2018, p. 835). The spatial framing effect considered in this paper can be thought of as a spatial scope effect, but without the confounding change in geographic extent or location of the underlying good. That is, we define spatial framing effects as the impact of *presenting an identical change* in biodiversity at *different geographical scales*. While spatial scope effects have been extensively explored (Logar and Brouwer, 2018; Rolfe and Windle, 2008; Spencer-Cotton et al., 2018; Van Bueren and Bennett, 2004; Vossler et al., 2023), the potential effects of spatial framing remain underexplored in the economic valuation literature. In theory, elicited values in a stated preference study should be unaffected by the spatial scale used to communicate the changes to respondents. However, inspired by, e.g., Van Bueren and Bennett (2004), we hypothesize that a larger geographical scale will make respondents more aware of a larger array of environmental issues and substitutes and, hence, result in a smaller WTP for the biodiversity improvement offered.

In addition to investigating the framing effect related to spatial scale, we also investigate the effects of attribute scope, namely attribute range effects. In the literature, particular attention has been given to the range of the cost vector (e.g., Börger et al. 2025 and Glenk et al. 2019) with less attention to the effect for non-price attributes (Meyerhoff et al., 2015; Luisetti et al., 2011). Generally, the findings regarding the cost vector indicate that changes in the range affect WTP (Glenk et al., 2019), but that this effect can be mitigated by using cheap talk and opt-out reminders (Börger et al., 2025). For non-cost attributes the results are more mixed. Luisetti et al. (2011) find evidence for attribute range effects, i.e., that respondents react to the relative, rather than absolute, levels of the attribute, violating a fundamental assumption underpinning SCEs. However, Meyerhof et al. (2015) find no effect of attribute range on the estimated preference parameters. Varying the attribute range can also be used for testing scope sensitivity, i.e., whether respondents' stated values are increasing (or at least not decreasing) with increasing improvements in the quantity or quality of the good. Scope (in)sensitivity has long been at the center of the validity debate concerning stated preference research (Carson, 2012; Hausman, 2012; Kling et al., 2012), and scope tests are thus considered an important part of validity testing (Johnston et al., 2017). While we do not expect that spatial framing effects will be affected by attribute range effects, or vice versa, we use a split sample experimental setup that combines variations in attribute range with variations in spatial frames. This enables us to test for scope sensitivity across different spatial frames, which we use to further evaluate the validity of value estimates obtained under different spatial framings.

3. Methods

3.1. The Biodiversity Intactness Index

It has been argued that the full extent of the concept of biodiversity cannot accurately be described with a single metric, and in practice, the design and use of biodiversity indices face data constraints (Mace, 2014; Pereira et al., 2013). A diversity of metrics have been suggested and used (Marshall et al., 2020), but one that stands out is the Biodiversity Intactness Index (BII), proposed by Scholes and Biggs (2005). The BII is endorsed by the Group on Earth Observations of the Biodiversity Observation Network and adopted by the Intergovernmental Platform on Biodiversity and Ecosystem Services as a 'core' indicator of progress toward the Convention on Biological Diversity's Aichi Targets (Martin et al., 2019). The BII is defined as "the average abundance of a large and diverse set of organisms in a given geographical area, relative to their reference populations" (Scholes and Biggs, 2005). Here, "reference populations" refer to an unaltered, pre-industrial state, for which the current conditions in protected areas can be used as a surrogate measure. The BII has been estimated worldwide by, e.g., Newbold et al. (2016), Purvis et al. (2018) and Gassert et al. (2022), using observations of species and their abundances collected in the PREDICTS database (Hudson et al., 2017). The data is used to estimate a relationship between human activities – specifically land use change and intensification – and biodiversity. As such, the BII does not measure the absolute level of biodiversity, but the condition, naturalness and intactness of the area, implied by the current level of biodiversity (as

Table 1
Spatial framing and attribute range of the biodiversity improvement attribute across the six different versions of the survey. The change in biodiversity is described as numerical changes in the average BII.

	Spatial framing:		
Attribute range:	"Small"	"Large"	"Small & Large"
Narrow	4 – 19	0.010 – 0.045	4 – 19; 0.010 – 0.045
Wide	4 – 38	0.010 – 0.091	4 – 38; 0.010 – 0.091

Note that the range of the attribute on the same row represents equivalent changes in biodiversity, but presented at either a small (100 km^2) or a large (42.000 km^2, i.e., national) spatial scale, or a combination of both.

predicted by the land use) relative to a theoretical intact baseline. However, it provides a scientifically sound, transparent, sensitive, dis-aggregable indicator of broad-sense biodiversity that can be compared with a baseline, applied at any spatial scale and readily interpreted (Purvis et al., 2018). Furthermore, it aligns well with people's perceptions of biodiversity (Uggeldahl et al., 2025) and fulfils the criteria of a biophysical indicator of nonuse value suggested in Boyd et al. (2023).

3.2. Study design

Our study uses a three (spatial framing) by two (attribute range) version study design, resulting in six different versions of the survey, outlined in Table 1. Respondents were randomly allocated to one of the six versions. The three different spatial framings used to describe the biodiversity improvement were:

- *"Small"*, where the improvement in biodiversity was described as an increase in the average BII for the specific area (100 km^2) in which the change would occur.
- *"Large"*, where the same improvement in biodiversity was instead described in terms of an increase in the national average BII arising from improving biodiversity in the specific area (100 km^2).
- *"Small & Large"*, where the improvement in biodiversity was described both in terms of the increase in the average BII for the specific area (100 km^2), as well as the corresponding increase in the national average BII.

It is important to note that the different spatial framings all describe exactly the same physical improvement in biodiversity in a 100 km^2 area. The only difference between the versions is the spatial scale at which this improvement is presented to respondents. For instance, a 4-point increase in the average BII in a 100 km^2 area is equivalent to a 0.01-point increase in the national average BII.[1] Furthermore, as detailed in the following section, the information provided to respondents in the survey did not give any indication of exactly where the specific area was located, apart from stating that it was far from the respondent's place of residence. It is also worth noting that the naming of the different spatial framings ("Small", "Large" and "Small & Large") was not used in the survey and is only used here to distinguish between the framings.

The spatial framing effect can be investigated by comparing estimates across the different spatial framing versions, given the same attribute range. Fig. 1 provides an example of a choice set, i.e., a policy alternative, showing the presentation of an equivalent change in biodiversity in the three different spatial framings. Table 1 outlines the resulting numeric differences in the improvements in BII used to convey the biodiversity improvement attribute to respondents.

Table 1 also displays that to test for attribute range effects and to conduct external scope tests, we employed two different ranges for the biodiversity improvement attribute. One version featured a (relatively) narrow attribute range, while the other used an extended – and thus wider and partly overlapping – attribute range. The specific levels of the biodiversity improvement attribute are shown in Table 2.

Insensitivity to attribute range requires that the marginal willingness to pay (MWTP) estimated over the roughly overlapping range (4−21) across the two versions, is equal. This implies that the WTP for the same improvement (e.g., 9 units) is the same across versions, i.e., not impacted by the presented overall range of improvements. Conditional on the insensitivity to attribute range, sensitivity to scope requires that the MWTP for improvements over 21 units in the wide range versions are positive, implying that the WTP for the larger biodiversity improvement levels (29 and 38 units) presented in the wide range should be higher than the largest improvement presented in the narrow range versions (19 units).

3.3. The survey

A long-standing critique of preference-based valuation of biodiversity has revolved around the public's low knowledge and limited understanding of the concept of biodiversity (Hanley et al., 1995; Bartkowski et al., 2015; Farnsworth et al., 2015). To address this concern, the development of the survey used for collecting data in our SCE included investigating the public's perception and understanding of biodiversity and its value employing a separate Q-methodology study. The results, reported in Uggeldahl et al. (2025), are in line with previous findings (e.g., Bakhtiari et al., 2014), indicating that lay people's perceptions of biodiversity to a large degree resonate with the academic and institutional definitions. This suggests that lay people may indeed be able to understand relatively comprehensive descriptions of biodiversity.

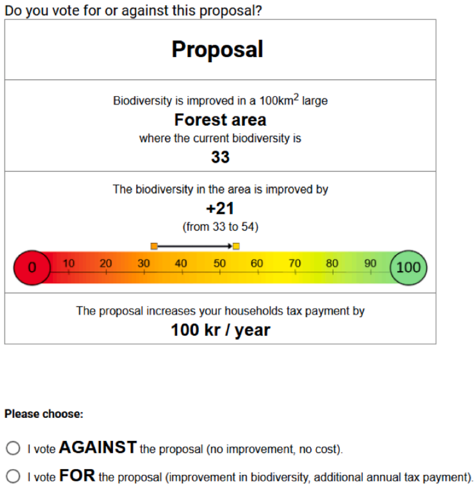
A draft of the survey was developed based on the findings from Uggeldahl et al. (2025), including inputs from the focus group discussions conducted therein. The survey consisted of a set of warm up questions, after which a general description of biodiversity was presented to respondents. This was followed by a description of how the BII measures biodiversity, and the status of biodiversity in Denmark using the BII was outlined (national average: 55.1). This included a description of how current land use types score on the BII.[2] Following these descriptions, the valuation scenario and the attributes were explained to respondents. A full version of the survey, translated from Danish, is available in the online supplementary material.

The scenario in the choice experiment was framed as a referendum concerning a proposed policy to improve biodiversity in a

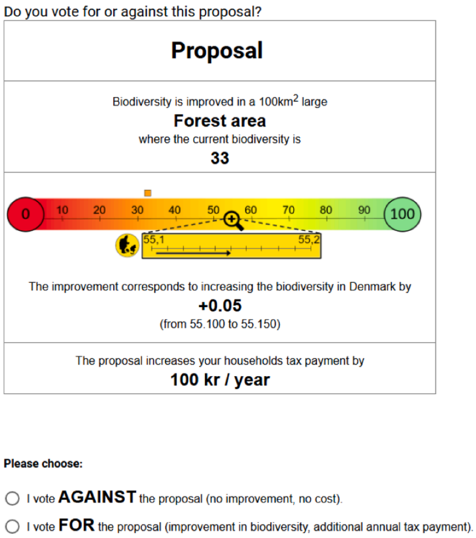
[1] As the surface area of Denmark is roughly 42 000 km^2, and 4 * 100 km^2 / 42 000 km^2 = 0.0095

[2] The BII values for Denmark, a raster map with cell size 100x100m, were obtained from Gassert et al. (2022), and land use data for Denmark, a raster map with cell size 10x10m cells, were obtained from Levin et al. (2017).

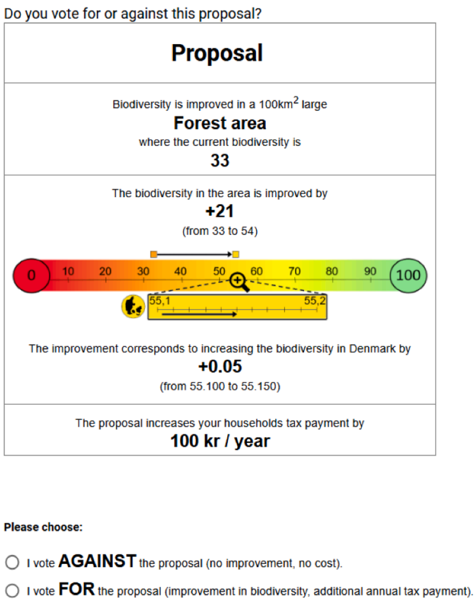
Small spatial scale: Improvement presented as the change in the average BII for the considered 100km^2 area and illustrated as the improvement from the current average BII in the area to the new average BII.



Large spatial scale: Improvement presented as the change in national average BII resulting from the improvement in a 100km^2 area and illustrated as the improvement from the current national average BII to the new national average.



Small & Large spatial scale: Improvement presented as both the change in the average BII for the considered 100km^2 area, as well as the change in national average BII resulting from the improvement in the area. Illustrated as the improvement from the current average BII in the area to the new average BII, as well as the improvement from the current national average BII to the new national average.



(caption on next page)

Fig. 1. Example of a choice set, showing an equivalent increase in biodiversity, for each of the spatial framings used in the paper (translated from Danish).

Table 2
Description of attributes and levels used in the stated choice experiment.

Attribute	Description shown to respondents (translated from Danish)	Levels*
The primary nature type in the area (*for*)	*Can be forest or open nature (e.g., meadows and heaths). The naturally occurring level of biodiversity will typically be higher in forests than in meadows and heaths.*	Forest or open nature
Current biodiversity in area (*cur*)	*Described by the biodiversity index, which can range from 33 to 57.*	33, 42 or 57
The size of the improvement in biodiversity (*imp*)	"Small" spatial framing version: *Measured as the increase in the average biodiversity index in the area. The increase can be between 4 and 38 and will occur over the next 10–20 years.*	4, 9, 13, 21, 29, 38
	"Large" spatial framing version: *Measured as the increase in the average biodiversity index in Denmark. Compared to the current national average of 55.1, the improvement can increase the average by between 0.010 and 0.091. The improvement will occur over the next 10–20 years.*	0.010, 0.021, 0.031, 0.05, 0.069, 0.091
	"Small & Large" spatial framing version: *Measured as the increase in the biodiversity index in the area. The increase can be between 4 and 38. This corresponds to an increase in the average biodiversity index in Denmark between 0.010 and 0.091, compared to the current 55.1. The improvement will occur over the next 10–20 years.*	4, 9, 13, 21, 29, 38 and 0.010, 0.021, 0.031, 0.05, 0.069, 0.091
Cost of the policy (*cost*)	*The cost is financed through an increase in the annual income tax. It will be used exclusively to pay for the implementation of the policy. The size of the payment depends on the household's income and is therefore not the same for everyone. You should consider whether your household is willing to pay the amount for the described policy.***	100, 250, 600, 1000, 1500, 2400 DKK/household /year

* For the narrow attribute range versions, the range of the biodiversity improvement attribute was approximately halved. The levels used in these versions were 4, 6, 9, 11, 15, 19 (small spatial framing) and 0.010, 0.014, 0.021, 0.026, 0.036, 0.045 (large spatial framing)

** All respondents were shown the same price levels. The last two lines in the description were included to avoid respondents engaging in approximations of aggregate costs of the proposed alternatives, something a few respondents indicated doing in the focus groups, and only to consider whether the individual cost of the proposal outweighs the value they hold for it.

100 km^2 area in Denmark. As our focus was on the nonuse value of biodiversity *per se*, the exact location that would be affected by the policy was deliberately not specified, nor were the specific actions that would be commenced to deliver the improvement. This was done to avoid the specific location affecting the estimates, and to reduce potential effects that the actions for improving biodiversity in themselves may have on the respondents' preferences for the policy (Czajkowski et al., 2009; Hanley et al., 2003). To ensure that the values expressed by the respondents were, as far as possible, reflecting only non-use values, the scenario explained that despite the precise location not having been decided on yet, the candidate areas were all located far from the respondent's place of residence. This was implemented so that when the respondent's postal code indicated they lived on the east (west) side of the Great Belt strait that divides Denmark, the candidate areas were said to be located on Jutland (Zealand) which is on the west (east) side of the Great Belt strait. While the strait can be crossed easily via an 18 km toll bridge, the toll is expensive, and it thus serves as a substantial barrier in the landscape (Olsen et al., 2020). Furthermore, the only information respondents had regarding the area was that it was somewhere on the other side of the Great Belt strait, compared to where they lived, i.e., either on Jutland (roughly 30 000 km^2) or Zealand (roughly 10 000 km^2).

The policy proposed in the referendum was described using four attributes (Table 2). The primary attribute of interest was the size of the improvement in biodiversity. Two other attributes were included to evaluate whether respondents perceived the improvement in biodiversity differently depending on the characteristics of the area where the change occurred. These attributes described the primary type of nature in the area, and the current level of biodiversity in the area. The fourth attribute described the cost of the policy to the respondents, using annual income tax as payment vehicle. After the attributes and the valuation scenario had been described to respondents, they were presented with six choice sets, each with two options: a "status quo" and a "proposed policy", presented as a referendum to vote for or against the proposed policy. Examples of choice sets presented using the different spatial framings are shown in Fig. 1.

In terms of the experimental design of the choice experiment, a full factorial design consisting of 216 combinations was used. Using a full factorial design avoids potential issues related to misspecification of the priors in efficient designs (Walker et al., 2018), and implies that interaction effects, which are of specific interest for this study, are independently estimable (Holmes et al., 2017). The full factorial design was blocked into 36 blocks of six choice sets each. The blocking procedure secured attribute level balance, i.e., that each respondent saw the levels of each attribute equally many times. Respondents were randomly allocated to one of the blocks, and the order of the choice sets within blocks was randomized. The same experimental design was used across all six survey versions.

Following the choice set sequence, a series of follow up questions were asked about the respondents' choices and perceptions of the valuation scenario.

The questionnaire was tested in a focus group interview with six respondents from the Copenhagen area, in addition to the preceding qualitative work conducted in Uggeldahl et al. (2025), on which the initial versions of the survey were based. The survey was adjusted based on the received feedback,[3] after which seven individual, cognitive interviews were conducted online with respondents from across the country, who answered an online version of the survey before the interview. Before the main data collection, a pilot survey with 195 completed responses, roughly distributed equally between the three different spatial framing versions, was conducted, after which final adjustments to the survey were made. The focus group and cognitive interviews, as well as the pilot study, indicated that the description of biodiversity generally matched the respondents' perceptions of biodiversity, although some respondents indicated that the focus on nonuse values made the choice sets "*challenging, but not impossible, to answer*".

3.4. Econometric approach

Responses to the discrete choice experiment are modelled based on the principles of Lancaster's characteristics theory of value and random utility theory (McFadden, 1973). In each choice set, *k*, a respondent, *n*, is assumed to choose the alternative, *i*, that maximizes their utility, where the utility is derived from the attributes of the good, and not from the goods *per se*. The utility of an alternative can thus be expressed as:

$$U_{nik} = \alpha p_{nik} + \beta x_{nik} + \quad \varepsilon_{nik} \tag{1}$$

where *p* denotes the cost attribute, *x* the vector of non-cost attributes, and α and β are the corresponding coefficients to be estimated. The random term, ε, is added as the analyst never directly observes utility. While the utility is often specified to be linear in parameters as in Eq. 1, the attributes themselves can have non-linear effects, e.g., when expressed in natural logarithmic form, or through interactions with other attributes or socio-economic characteristics.

By assuming that the unobservable parts of utility are independently and identically distributed extreme value, the probability of respondent *n*'s sequence of choices can be represented by the conditional logit model:

$$P(y_n|p_n x_n) = \prod_{k=1}^{K} \frac{\exp(\alpha p_{nik} + \beta x_{nik})}{\sum_{j=1}^{J} \exp(\alpha p_{njk} + \beta x_{njk})} \tag{2}$$

where y_n is the sequence of choices observed over the K choice tasks. However, more flexible models such as the random parameters logit (RPL) model, have become standard practice within choice modelling. RPL models allow the preference parameters, β_n, to vary within the population, according to a distribution specified by the researcher. Usually, a normal distribution is assumed for many preference parameters when there is no strong *a priori* or theoretical expectation regarding the parameter's sign. The choice probabilities are approximated through simulation by taking draws from the assumed distribution. For more information see Train (2009).

The coefficients estimated through models such as the one specified in Eq. 2, indicate the effect of each observed variable relative to the variance of the unobserved factors (the scale parameter). Because the scale factor can vary between samples, the coefficients from different samples are not directly comparable. However, what is usually of interest in environmental valuation is not the coefficients themselves, but the ratio of the non-cost coefficients to the cost coefficient, i.e., the MWTP for changes in the non-cost attributes. Taking the ratio of the coefficient cancels out the effect of scale, and hence, WTP estimates between samples are directly comparable. To bypass the extra step of first estimating a model, and then calculating the ratios of the estimated parameters and the standard errors of these ratios, the model in Eq. 2 can be specified in WTP-space, as opposed to preference space (Train and Weeks, 2005). Denoting the WTP as $w_n = \beta_n/\alpha_n$ the probability of the choice sequence represented by the RPL model becomes[4]:

$$P(y_n|p_n, x_n, \theta) = \int \prod_{k=1}^{K} \frac{\exp(\alpha_n p_{nik} + (\alpha_n \quad w_n) x_{nik})}{\sum_{j=1}^{J} \exp\left(\alpha_n p_{njk} + (\alpha_n \quad w_n) x_{njk}\right)} f(\quad w_n|\theta) \quad d(\beta_n) \tag{3}$$

where $f(w_n|\theta)$ represents the distribution of potentially random parameters, and θ the parameters of the distribution. In our case the non-cost parameters are assumed to follow a normal distribution, with mean *b* and standard deviation σ, and the cost parameter is assumed to be log-normally distributed, following standard practice. While the estimated WTP is scale invariant and can be compared across samples, the cost coefficient, α, is not, as it is still estimated relative to the scale parameter (Train and Weeks, 2005).

In the analysis, utility of an alternative is generally defined as (dropping the subscripts for brevity):

$$U = \alpha_{cost}(-cost + imp(\beta_{imp} + \beta_{cur*imp} current + \beta_{for*imp} forest) + \beta_{SQ} SQ) \tag{4}$$

indicating that the attributes describing the characteristics of the area, i.e., the current level of biodiversity (*current*) and the primary

[3] The respondents indicated that the BII provided an intuitive way of measuring biodiversity, but lacked references points indicating what was a "good", or "good enough" level, subsequently making the choice scenarios hard to answer. It was thus decided to provide some reference points for the BII, and descriptions of the average BII for different land use types in Denmark was included in the survey, available in the supplementary material. Respondent also found the survey quite text heavy, which resulted in a slight rewriting and shortening of the description, and some of the non-essential text being placed behind clickable pop-ups in the online version of the survey.

[4] Note that in estimation, the values for the cost attribute are entered as negative, given the theoretically assumed disutility for cost.

nature type in the area (entered as dummy variable *forest,* compared to open nature), only enter as interactions with the improvement attribute (*imp*). These attributes are characteristics of the area that are not affected by the proposed project, and therefore they do not vary within a choice set. Hence, they are not identifiable in themselves, but only through interaction with attributes that vary between the alternatives. We chose the interaction with the improvement attribute, as this yields intuitive interpretation and policy relevance, i.e., answering how the MWTP for biodiversity improvements is affected by the characteristics of the improved area. An alternative specific constant (*SQ*) is included for the status quo alternative, i.e., voting against the policy, and captures the effects on choices not captured by the attributes. The increased tax payment associated with the policy is described by *cost.*

In Eq. 4, the marginal effect of *imp* is assumed to be constant across the range of the improvement attribute, i.e., same for larger and smaller improvements. Note, that given the interaction with the attributes describing the characteristics, the marginal effect of *imp* is not constant, but depends on the levels of these characteristics. However, initial analysis based on a dummy coding of the improvement attribute, i.e., without making any assumptions regarding the functional form, indicated that the marginal effect of *imp* was not constant, but declining in the size of the improvement (see S.2 in the supplementary material). Given this dependence of the marginal effect of *imp* on the magnitude of presented change in biodiversity, a piecewise linear specification was used, ensuring that the MWTP is estimated over a roughly equal range for the wide and narrow attribute range versions. This enables a direct comparison of the MWTP estimates between the versions, and thus a test of the effects of attribute range. The utility for these models is specified as:

$$U = \alpha_{cost}(cost + \quad d(\text{imp} \leq 21)\beta_{imp\leq 21}\text{imp} + (1 - d(\text{imp} \leq 21))(21 * \beta_{imp\leq 21} + \beta_{imp>21}(\text{imp} - 21)) + imp(\beta_{cur*imp}current + \beta_{for*imp}forest) + \beta_{SQ}SQ) \tag{5}$$

where $d(\text{imp} \leq 21)$ is an indicator taking the value 1 if the biodiversity improvement level for the alternative is above or equal to 21, and 0 otherwise. This specification allows for a "kink" at 21 units, which was chosen because it most closely equalizes the ranges between the wide version (improvement levels: 4, 9, 13, 21, 29, 38) and the narrow versions (improvement levels: 4, 6, 9, 11, 15, 19). It should be noted that for the narrow attribute range versions, Eq. 5 reduces to Eq. 4. To simplify model estimation and presentation, only a linear effect is estimated for the interactions of improvement with the current level of biodiversity and the nature type.

The effect of spatial framing is evaluated based on the estimated MWTP for biodiversity improvements, across the different spatial framings for the versions with an identical attribute range. The effects of attribute range will be evaluated by comparing the MWTP estimated over the same range ($\leq$21 units of improvement), between the versions with the same spatial framing, but where the range of the improvement attribute differed. Assuming no attribute range effects are found, i.e., identical MWTP for biodiversity improvements $\leq$ 21 units, an external scope sensitivity test can be conducted by evaluating whether the MWTP for biodiversity improvements over 21 units are positive (or non-negative in the weakest form of the test), indicating that respondents are willing to pay more for the larger improvements presented in the wide range version compared to the narrow versions.

External scope sensitivity can also be evaluated based on the results from a model with dummy coding of the *imp* attribute (S.2 in the supplementary material). Note however, the slightly different model specification used, as the attributes relating to the characteristics of the area are interacted with the alternative specific constant for the proposed policy alternative. With this specification, scope sensitivity requires that the WTP for the larger improvement levels shown in the wider attribute range split (21, 29, 38) are larger, or at least equal to, the WTP for the largest level shown in the narrow range split (19) within the same spatial framing.

4. Results

4.1. Description of the sample

The data was collected by extracting a representative random sample of 34,874 Danish citizens aged 18–75 from the Danish Central Person Register. The selected citizens were sent an invitation to participate in the survey via their digital mailbox, the main communication channel between Danish public authorities and citizens, which all citizens in Denmark aged 15 and older are required to have. This avoids the potential coverage errors associated with using online panel providers (Bonnichsen and Olsen, 2016). The survey could be completed on mobile phone, tablet or computer, which was done by 53 %, 45 % and 2 %, respectively.

The invitations to participate in the survey were sent on 14 December 2023. On 4 January 2024 reminders were sent to respondents who had not yet opened the questionnaire. The data collection closed on 21 January 2024. In total 6357 respondents completed the survey, resulting in a response rate of 18.2 %, varying between 17.5 – 18.7 % across the different versions. The median response time was 12.4 min (5 % trimmed mean: 14.1 min).[5] For the final sample used in the subsequent analysis, protesters and strategic respondents were removed. These were identified based on follow-up questions targeting respondents who either chose the "status quo" (potential protesters) or the "proposed policy" (potential strategic respondents) option consistently across all six choice sets. If the follow-up questions for these respondents indicated that they were not revealing their true preferences, they were removed from the sample (see statements used to exclude respondents due to protesting or strategic behavior in Tables S.4.1 and S.4.2 in the supplementary material). The rates of protesters and strategic respondents are shown in Table S.1 in the supplementary material, along with the demographic characteristics of the final sample. Protest rates were found to be statistically higher in the "Large" spatial framing, compared to the "Small" spatial framing. No statistically significant differences in the amounts of strategic respondents, or

[5] The median (5 % trimmed mean) time spent on the introduction and warm up questions, consisting of 6 screens, was 3 min (3.4 min), the description of biodiversity and choice scenario, consisting of 8 screens, 3.4 min (3.9 min), and the first choice set 40 s (45 s).

sociodemographic variables were found between versions when respondents who did not want to disclose income are excluded. Removing protest- and strategic respondents resulted in an effective sample of between 732 – 785 respondents across the different versions.

When asked about the description of biodiversity presented in the survey, 89 % of all respondents (85–91 % across versions) indicated that the description corresponded well or very well with their prior perception of biodiversity. No significant differences in views regarding the description were found across the six split samples.

4.2. Perceptions of the scenario

If respondents embed different things into biodiversity improvements depending on the spatial framing, this might affect how they perceive the magnitude and consequences of the improvements presented in the scenarios. After completing the six choice sets, respondents were asked about their perceptions of the size of the improvement in biodiversity offered in the first choice set. Specifically, respondents were asked to indicate on a scale ranging from "very small" to "very large", how they perceived the magnitude of the improvement in biodiversity both for the 100 km^2 area in itself, as well as for the overall biodiversity in Denmark. The attribute levels in the first choice set were shown again to respondents when answering these follow up questions.

Fig. 2 illustrates the shares of respondents that perceived the size of the improvement as small/very small or large/very large. Panels A and B in Fig. 2 display respondents' perceptions of the size of the improvement with respect to the biodiversity in the 100 km^2 area. Respondents in the "Large" spatial framing (both in the wide and narrow attribute range versions) perceive the improvement to be smaller than when the other spatial framings are used, as confirmed by a Chi-square tests (p-value <0.05). Note that respondents in the "Large" spatial framing were not explicitly provided with any information regarding the size of the change in biodiversity in the 100 km^2 area itself, but only on the implication for the overall biodiversity in Denmark (see Fig. 1). No significant difference in the perceptions between respondents in the "Small" and the "Small & Large" spatial framing (both when comparing the wide and narrow attribute range versions) are found.

Displayed in panel C and D in Fig. 2, the perceptions regarding the size of the improvement in terms of the biodiversity in Denmark show a similar tendency in the survey versions using a wide attribute range (panel C), but here significant differences are found between all three spatial framings base on chi-squared tests. In the survey versions using a narrow attribute range (panel D), however, respondents in the "Large" and the "Small & Large" spatial framings perceive the improvements for biodiversity in Denmark to be relatively smaller than respondents subjected to the "Small" spatial framing.

While not directly evident from Fig. 2, respondents in the narrow attribute range versions (panels B and D) perceive the improvement in biodiversity to be smaller than respondents in the wide versions (panels A and C). The difference in perceptions between the wide and narrow attribute range versions are significant for the "Small" and the "Small & Large" spatial framings, irrespective of whether the perception of the size of the improvement concerns the specific area (panels A vs B) or the whole of Denmark (panels C vs D). This is a first indication that respondents in these versions are scope sensitive, i.e., react to the size of the presented improvements, as larger improvements have been shown in the wide range attribute range versions. The difference between the wide and narrow versions was not significant in the "Large" spatial framing.

4.3. Estimation results

Table 3 presents estimation results for the RPL models for the six versions of the survey, with utility specified according to Eq. (5), i. e., where the effect of *imp* for the wide attribute range versions is estimated with a piecewise linear specification. In the model specifications, utility is specified in WTP-space and all non-cost attributes as well as the alternative-specific constant for the status quo are modelled as normally distributed random parameters with a mean parameter and a standard deviation parameter being estimated for each attribute. The cost attribute is also modelled as a random parameter, though assumed to follow a log-normal distribution. All models are estimated using the Apollo software in R (Hess and Palma, 2019) with 2000 MLHS draws (Hess et al., 2006).

The results for the piecewise linear specification applied for biodiversity improvements indicate that improvements up to 21 units on the BII ($\beta_{imp\leq21}$), which is roughly the range for which the wide and the narrow attribute range versions overlap, have a significantly positive impact on the mean WTP in all split samples except for the wide attribute range version for the "Large" spatial framing. The estimates for the standard deviation ($\sigma_{imp\leq21}$) indicate significant heterogeneity regarding the effect of improvements in all survey versions. For the wide attribute range versions, considering the main effect of larger improvements (over 21 units on the BII), the parameter estimates for the mean ($\beta_{imp>21}$) are insignificant for the "Small" and the "Small & Large" spatial framings, while they are significant for the "Large" spatial framing. The insignificant estimate for $\beta_{imp>21}$ in the "Small" and the "Small & Large" implies that respondents do not derive additional value from improvements over 21 units. However, the effect of *imp* also depends on the interaction terms. The aggregate effect on MWTP is shown in Table 4. That $\beta_{imp\leq21} > \beta_{imp>21}$, adheres to the standard assumption of diminishing marginal utility. However, it should be noted that we only find decreasing marginal utility in biodiversity improvements, not in the overall level of biodiversity, as the mean estimates for $\beta_{cur*imp}$ in the "Small" and the "Small & Large" spatial framings are insignificant. The contrary results in the "Large" spatial framing (wide attribute rage version), i.e., $\beta_{imp>21} > \beta_{imp\leq21}$, implies that respondents derive more value from the final units of larger biodiversity improvements (as also evident from the dummy coded model presented in S.2 of the online supplementary material). Only in the "Large" spatial framing and narrow attribute range version is the current level of biodiversity ($\beta_{cur*imp}$) found to have a negative and significant effect on the MWTP of biodiversity improvements, which would imply a diminishing marginal utility of biodiversity, as could be expected based on economic theory. Many of the associated

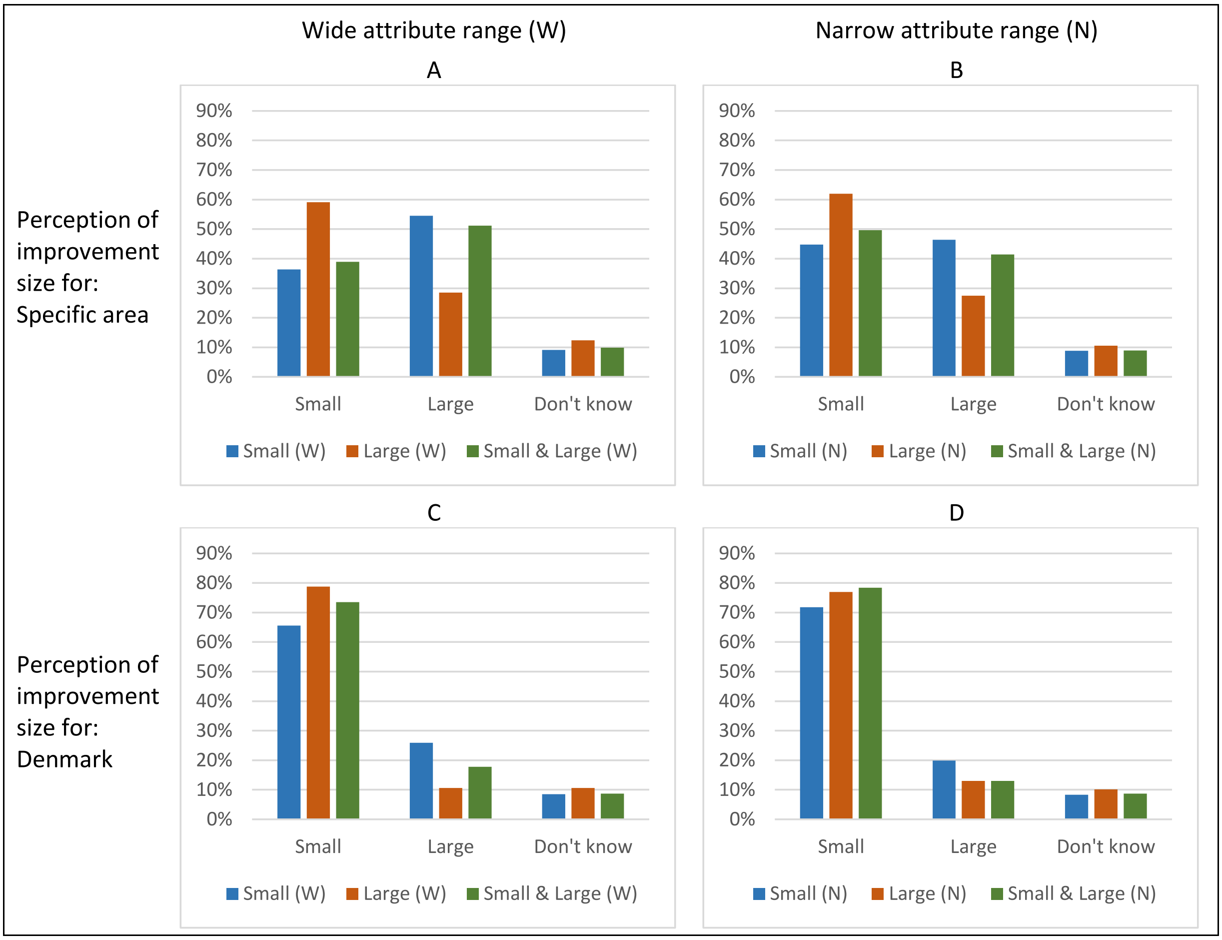


Fig. 2. Shares of respondents perceiving the size of the improvement as large or small. Respondents were asked to indicate their perception of the improvement in the first choice set displayed to them. As choice set order was randomized, all improvement levels were presented roughly the same number of times, and the perceptions illustrated in the figure thus reflect averages over the different improvement levels shown in the first choice set.

estimates for the standard deviation ($\sigma_{cur*imp}$) indicate significant heterogeneity among respondents.

Similarly to the current biodiversity in the area, the mean effects of the primary nature type ($\beta_{for*imp}$) are insignificant at the 5 % level for all but one version. In the "Small" spatial framing with a narrow range, respondents are willing to pay about 10 DKK more for every unit the BII is improved in forested areas compared to open nature areas. There is considerable heterogeneity in preferences for the nature type, though not in the wide attribute range version of the "Small" spatial framing and in the narrow attribute range version of the "Large" spatial framing.

The estimates for the mean effect of the SQ (β_{SQ}) are negative and significant for all split samples. This indicates that respondents across versions generally prefer the proposed policies over the current situation regardless of the attributes and levels of said policies, though also here subject to considerable heterogeneity across respondents. While the estimates for the improvement attribute represents preferences for biodiversity improvement on the intensive margin, the SQ can be thought of as capturing preferences on the extensive margin (i.e., the WTP for any policy change, which can also be interpreted as a dissatisfaction of current policy). The SQ coefficients are significantly different between spatial framings (p-value < 0.04[6]), except between the narrow attribute range versions of the "Small" and the "Small & Large" spatial framing (p-value: 0.11), being largest in absolute terms for the "Large" spatial framing, followed by the "Small & Large" and the "Small" spatial framing.[7] This indicates that the spatial framing also influences the extensive margin of choice, which to some degree can be seen as compensating for the lower MWTP observed for the "Large" spatial framing. Generally, the relatively large estimates of the SQ coefficients in all versions indicate a strong preference for any biodiversity improving project, irrespective of the attribute levels. This could also indicate issues with the identification of RPL models as discussed

[6] Standard errors for the difference between parameter estimates based on Paternoster et al. (1998).
[7] The differences in SQ coefficients within the same spatial framing were not significant (p-value > 0.13).

Table 3
Results from RPL models in WTP space for all versions of the survey, with utility specified as in Eq. (5) – piecewise linear effect of biodiversity improvements. Parameter estimates for all non-cost attributes indicate marginal WTP in DKK/household/year. 100 DKK = 13.4 EUR.

Spatial framing (attribute range):	Small (wide)	Small (narrow)	Large (wide)	Large (narrow)	Small & Large (wide)	Small & Large (narrow)
Attribute:						
Means of random parameters						
α_{cost}	0.78 ***	0.93 ***	1.32 ***	1.30 ***	0.87 ***	1.37 ***
s.e.	0.08	0.11	0.21	0.14	0.12	0.15
$\beta_{imp\leq21}$	50.64 ***	53.85 ***	7.84	28.03 ***	35.37 ***	46.27 ***
s.e.	7.83	10.77	11.48	2.12	6.47	10.69
$\beta_{imp>21}$	9.52		8.89 **		-2.47	
s.e.	7.24		3.72		7.08	
$\beta_{cur*imp}$	0.14	0.25	-0.11	-0.40 ***	0.18 ,	-0.07
s.e.	0.12	0.21	0.07	0.02	0.10	0.15
$\beta_{for*imp}$	3.75	10.26 *	5.21 ,	-2.35	4.40 ,	6.20
s.e.	2.33	4.76	3.02	2.35	2.30	4.87
β_{SQ}	-409 ***	-548 ***	-1060 ***	-979 ***	-622 ***	-728 ***
s.e.	66.83	63.31	91.38	39.75	77.39	93.04
Standard deviations of random parameters						
σ_{cost}	0.55 ***	0.88 ***	1.50 ***	1.57 ***	0.97 ***	1.23 ***
s.e.	0.12	0.16	0.28	0.22	0.18	0.20
$\sigma_{imp\leq21}$	40.63 ***	55.29 ***	25.26 **	11.95 ***	39.70 ***	56.33 ***
s.e.	6.94	6.29	9.85	1.07	3.47	5.96
$\sigma_{imp>21}$	23.89 ,		0.18		18.74	
s.e.	14.24		1.79		14.28	
$\sigma_{cur*imp}$	0.27 ,	0.23	0.11 **	0.05 ***	0.08 **	0.07 **
s.e.	0.14	0.26	0.04	0.01	0.03	0.03
$\sigma_{for*imp}$	3.99	33.78 ***	5.93 *	1.12	10.38 *	37.94 ***
s.e.	4.07	9.77	2.85	0.84	4.47	6.08
σ_{SQ}	769 ***	868 ***	1038 ***	1100 ***	987 ***	940 ***
s.e.	65.70	51.63	65.87	43.61	55.31	69.37
# respondents	756	758	734	732	778	785
Final LL	2296	2353	2246	2242	2434	2341
Rho-squared	0.270	0.253	0.264	0.264	0.248	0.283

Note: Simulations done using 2000 MLHS draws, standard errors constructed using the robust sandwich estimator. The estimates for the non-cost attributes have been rescaled to DKK from 1000 s DKK used in estimation. '***' indicates significance at the 0.001 level; '**' at the 0.01 level; '*' at the 0.05 level and "," at the 0.1 level.

Table 4
MWTP estimates (in DKK/household/year) for biodiversity improvements depending on the level of biodiversity in an area where the primary nature type is open nature. 100 DKK = 13.4 EUR.

Spatial framing (attribute range):	Small (wide)	Small (narrow)	Large (wide)	Large (narrow)	Small & Large (wide)	Small & Large (narrow)
Improvement (≤21)						
MWTP (at BII: 33)	55.3 ***	62.0 ***	4.3	15.0 ***	41.4 ***	44.0 ***
(s.e)	5.7	6.5	10.1	2.9	5.3	6.9
MWTP (at BII: 42)	56.5 ***	64.2 ***	3.3	11.4 ***	43.0 ***	43.4 ***
(s.e)	5.5	6.2	9.8	3.1	5.3	6.2
MWTP (at BII: 57)	58.6 ***	67.9 ***	1.7	5.5	45.7 ***	42.4 ***
(s.e)	5.6	7.0	9.4	3.4	5.6	5.5
Improvement (>21)						
MWTP (at BII: 33)	14.2 **		5.3		3.5	
(s.e)	6.0		3.6		5.3	
MWTP (at BII: 42)	15.4 **		4.4		5.2	
(s.e)	6.1		3.8		5.1	
MWTP (at BII: 57)	17.5 **		2.7		7.9	
(s.e)	6.6		4.3		5.1	

Note: Standard errors for MWTP estimated using the delta method. '***' indicates significance at the 0.001 level; '**' at the 0.01 level; '*' at the 0.05 level and "," at the 0.1 level.

in Walker (2002). However, these model identification issues generally relate to SCEs with more than two alternatives. Furthermore, the estimates from MNL models (see S.4 in the online supplementary material) are of similar magnitude.

Table 4 presents the MWTP of a one unit increase in the BII of a 100 km^2 area, calculated at the levels of the current biodiversity presented in the survey, and assuming the area is primarily open nature. The calculations include the effect of the current level of biodiversity also for the splits where this was found to be insignificant based on Table 3. Comparing the main effects of biodiversity improvements up to 21 units across spatial framings suggest the presence of a spatial framing effect. Mean MWTP for improvements up to 21 units are significantly higher in the "Small" and "Small & Large" spatial framing compared to the "Large" spatial framing, as can also clearly be seen from Fig. 3. The MWTP obtained in the "Small & Large" spatial framing falls in between the other two, though the difference from the "Small" spatial framing is not statistically significant, except when comparing MWTP between the narrow attribute range versions at the current BII levels of 42 (64.2 DKK vs. 43.4 DKK) and 57 (67.9 vs. 42.4 DKK), where the p-value for the difference is 0.02 and 0.004 respectively.

We find that the MWTP estimates are insensitive to the presented attribute range, as a *t*-test fails to reject the equality of the estimated MWTP between the wide and the narrow attribute range versions in the "Small" and the "Small & Large" spatial framings, when evaluated over roughly the same improvement range (0–21 units). This is also clear from a visual inspection of Fig. 3 and eliminates concerns that respondents might be affected by the attribute level range used. No significant differences in MWTP between the attribute range versions in the "Large" spatial framing are found either, but the estimates themselves are also insignificant, except at the lower levels of current biodiversity (33 and 42) in the narrow attribute range version.

Finally, we find that the "Small" and "Small & Large" spatial framings pass external scope tests, i.e., that WTP for the larger improvement levels in the wide attribute range versions are larger than the WTP for the largest improvement in the narrow version. For the "Small" spatial framing, this is indicated by the positive and significant MWTP estimates for improvements over 21 units shown in Table 4. However, more direct tests can be conducted by comparing the results from models with the improvement levels dummy coded (see S.2 in the online supplementary material).[8] These tests compare the highest WTP for an improvement level in the narrow attribute range versions with the WTP for the higher levels in the wide attribute range versions. In the "Small" spatial framing the WTP estimates for 29 and 38 units of improvement in the wide attribute range version are significantly higher than the WTP for a 19 units improvement in the narrow attribute range version (p-values 0.01 and 0.003, respectively). In the "Small & Large" spatial framing the WTP for a 38 units improvement is significantly larger than the highest WTP in the narrow attribute range version estimated for a 15 unit[9] improvement (p-value 0.01). These results imply that both the "Small" and the "Small & Large" spatial framings pass an external scope test. In the "Large" spatial framing, the results from the dummy coded model indicate insensitivity to scope. The findings regarding scope sensitivity align with the observations regarding perceptions of the size of the improvement between the narrow and the wide attribute range versions in Fig. 2. Taken together, this indicates that the spatial framing can also affect validity.

5. Discussion & Conclusion

Estimating the nonuse value of biodiversity requires a metric that captures the aspects of biodiversity affecting utility. Theoretical arguments (Boyd et al., 2023) and practical findings (Uggeldahl et al., 2025) indicate that more holistic metrics than previously used might be preferable for measuring and presenting biodiversity changes in stated preference studies. We use one such metric, the Biodiversity Intactness Index (BII) which is broadly endorsed by ecologists, in a SCE for estimating the nonuse value of biodiversity improvements in Denmark. In light of the concern that respondents might be more susceptible to context and framing effects when abstract concepts like biodiversity are measured using complex ecological indices, we test for potential effects of spatial framing and attribute range on WTP. Spatial framing effects have, to our knowledge, not previously been examined in the literature.

Our results clearly indicate the presence of a spatial framing effect, as the MWTP differs significantly depending on the size of the area over which the improvement is calculated, despite the actual size of the improvement being identical. When improvements are presented using smaller geographic aggregation levels (the "Small" spatial framing), we find that MWTP is higher compared to presenting at a larger geographic aggregation level (the "Larger" spatial framing). This result is similar to findings in the literature related to scope effects (e.g., Van Bueren and Bennett, 2004), where value estimates are found to be lower when the good was framed in a national context, versus a local context (although this framing effect is confounded with differences in scope and contextual aspects).

That the perception of the good is affected by the spatial context in which it is presented is also supported by follow-up questions: respondents' perceptions of the size of the improvement differ depending on the spatial scale at which the BII is presented (Fig. 2). The differences in perceptions are also in line with the estimated MWTPs (Fig. 3).

In practical terms, one explanation for this effect is the difference in the relative size of the improvement compared to the benchmark between the spatial framings. Presented in the "Small" spatial framing, the improvement against the current baseline in the area will be bigger in relative terms, compared to the identical improvement measured against a national baseline. It might be that the bigger relative change in the "Small" spatial framing passes some threshold where it is concrete enough to be considered valuable for the respondent, compared to the more abstract and relatively smaller improvement in the "Larger" spatial framing. This argument is perhaps refuted by the finding that WTP on the extensive margin, i.e., including the negative WTP for the SQ option, to some degree evens out the differences in WTP between treatments. However, such a behavioral interpretation of the SQ involves an explicit assumption that the estimate represents the utility of the SQ alternative relative to the proposal alternative and is not just seen technically as a means of capturing the average effect on utility of any other factors not included in the model (Meyerhoff and Liebe, 2009).

Other potential explanations for the spatial framing effects are that the "Larger" spatial framing might increase the number of substitute or complementary goods that the respondents consider or are indirectly reminded of when evaluating the proposed policies

[8] It should be noted that the dummy coded model differs slightly from the models in Table 2, as the attributes related to the characteristics of the area are interacted with the alternative specific constant for the "proposed policy" alternative.

[9] In the narrow attribute range version of the "Small & Large" spatial framing, the dummy coded model showed the highest WTP for a 15 point improvement, however, this was not significantly higher than a 19 point improvement.

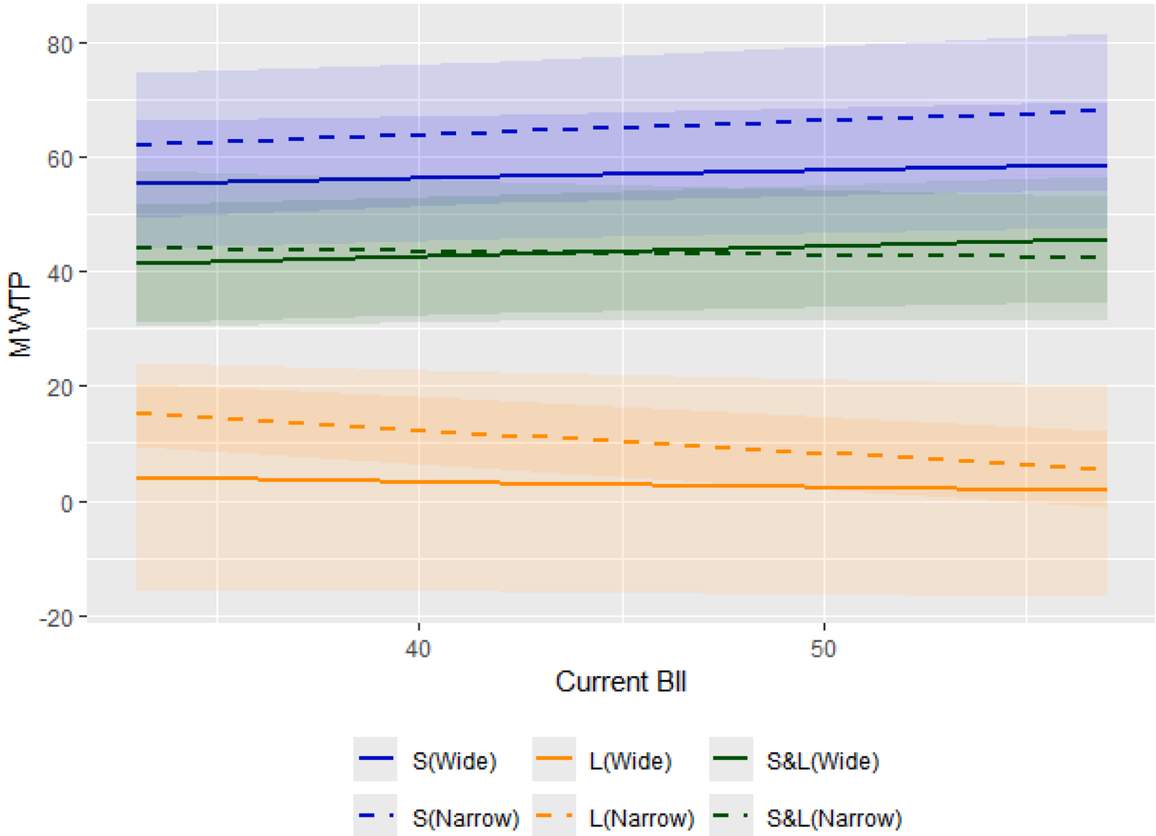


Fig. 3. MWTP for biodiversity improvement (for the first 21 units) depending on current level of biodiversity in the area where the primary nature type is open nature. Bands represent the 95 % confidence interval.

(Van Bueren and Bennett, 2004). Being remined of an increased number of substitutes would likely decrease the value of an individual improvement. This explanation is consistent with the embedding effect, i.e., that respondents are willing to pay more for a good when assessed individually compared to when valued as part of a more inclusive package. It might also be that a "Larger" spatial framing indirectly serves as a reminder of the overall context and scale of the project, in a similar way as concluded in the study by Jacobsen et al. (2011), where WTP was lower when respondents were reminded that the considered nature restoration project only was a small part in the overall context of nature preservation plans.

Finally, the spatial framing might also be interpreted by the respondents as a cue regarding the institutional setting, i.e., the authority that might be in charge of delivering the improvement. If there is a preference for local authorities to manage local projects as found by Konisky (2011), that could affect respondents' preferences for the project proposed in the survey. While we cannot rule out the possibility that respondents in the "Larger" spatial framing version might have misunderstood the fact that the presented improvements referred to the national-equivalent biodiversity improvement of a change taking place in a given area of 100 km^2, this issue was not observed during the pre-testing of the survey, nor indicated by answers to a follow up question concerning the perceived character of the improvement. Furthermore, this fact was stressed both in the information directly preceding the choice sets (see survey in the supplementary material), as well as explicitly stated in the choice sets (see Fig. 1).

Regardless of which of these explanations actually drive the effect, they all seem compatible with the finding that presenting respondents with both pieces of information in the "Small & Large" spatial framing yields welfare estimates that fall in between those obtained when presenting only a "Small" or "Large" spatial framing.

It should also be noted that the spatial framing is not only researcher induced but can also reflect a true underlying policy and value context. The context in which the policy is valued should naturally reflect the characteristics of the policy in question, and the nonuse value linked to that policy. This guidance should be seen along that of Boyd et al. (2023), who argue that the ecological metric should be measured at "*geospatial scale linked to the existence value in question*" (Boyd et al., 2023, p. 9). However, the policy context or spatial scale linked to the value might not always be clear. For instance, a policy can have a national goal, which is reached by local actions. Consider the improvement of biodiversity in an area as part of a policy only concerning this local area, e.g., the establishment of a national park. In this case the improvement in biodiversity should probably be presented using a "Small" spatial framing. However, if the improvement in the area is part of a national biodiversity policy, it supposedly has national importance (see, e.g., Jacobsen and Thorsen, 2010 for discussion of such aspects in the Danish context). In such a setting, informing respondents of both the "small" as well as the "large" effects would seem like a reasonable compromise. This question resembles that of whether indicators should be presented in cardinal or relative terms,[10] where arguments for presenting both have been made on the grounds of increasing interpretability (Johnston et al., 2012).

In any case, our results corroborate the findings of Faccioli and Glenk (2022) in that the framing of objectively equivalent information can affect outcomes in stated preference research, and that practitioners should recognize and be aware of this, particularly during study development, where attribute framing decisions can sometimes be made rather arbitrarily. Our results indicate that the choice of spatial framing is not trivial, and that transferring marginal values across different spatial contexts might thus incur substantial transfer errors. This may in part explain findings in past benefit transfer studies that transferring values across spatial scales and jurisdictional boundaries can have significant implications for transfer validity (e.g., Johnston and Duke, 2009; Van Bueren and Bennett, 2004).

Turning next to the construct validity of our findings, we find that in two of the spatial framings, the "Small" and the "Small & Large", the marginal value estimates are insensitive to the presented attribute range, indicating that respondents react to the absolute, rather than relative, size of the presented improvements, eliminating concerns that respondents might be affected by the attribute level range used. This corroborates the findings in Meyerhof et al. (2015) but differs from Luisetti et al. (2011) and the general findings on cost vector effects (Glenk et al., 2019). However, contrary to past studies on attribute range, we have used a piecewise linear specification and compare the effect of improvements when estimated over the same range, thus controlling for possible effects due to diminishing marginal utility. If the effect was assumed to be linear over the full range, the estimates in the wide versions would be significantly lower than in the narrow versions. Our results using the piecewise linear specification of the improvement attribute imply that the marginal value of an improvement depends on the size of the improvement. This can potentially lead to counterintuitive conclusions, such that the value of two sequential smaller projects in the same area would be higher than the value of a single project with the same combined improvement.

Furthermore, we find that our results are sensitive to the scope of the improvement, which is formally tested in an external scope test, which the "Small" and the "Small & Large" spatial framings pass. The fact that the value estimates from the "Large" spatial framing are sensitive to the presented attribute ranges as well as insensitive to scope, suggests that the spatial framing not only impacts the value estimates, but also compromise their validity.

In terms of further criteria related to construct validity, we conclude that our study passes the face validity test of WTP overshooting suggested by Glenk et al. (2024). Furthermore, income has the expected effect on WTP, with higher income individuals generally being willing to pay more for biodiversity improvements.[11] In relation to the other C's of validity assessment (Bishop and Boyle, 2019), content validity is enhanced by the rigorous qualitative work preceding the survey development (Uggeldahl et al., 2025), leading to the choice of a biophysically valid linking indicator. While convergent and criterion validity are harder to assess given the limited number of similar studies and the inherent lack of any "true" estimates for the nonuse values in question, the estimates largely have the sign which could be expected based on previous literature.

Finally, while our study has focused explicitly on non-use values, there is no theoretical argument for why such effects might not be present for use values. On the contrary, given that there is theoretical and empirical evidence for use values being affected by spatial aspects, we suspect that the spatial framing of the good is likely to also influence use values.

To summarize our findings for practical applications, we find that the use of the BII passes various validity tests, indicating that it can be a way to present changes in biodiversity in SCEs. In terms of how geographically explicit changes should be presented, our results indicate that presenting spatially explicit ecological indices at a smaller geographical scale, compared to only presenting at larger scales, seems to produce results that better adhere to standard validity criteria. However, whether to also include information regarding the effect on a large spatial scale is a question warranting further empirical scrutiny, and probably also depends on the policy question being analyzed. We conjecture that also including information about the effect on a larger spatial scale might improve the evaluability of the information (see Bateman et al. 2009 for a discussion of evaluability), as it should help respondents to better comprehend the magnitude of the proposed change.

CRediT authorship contribution statement

Kennet Christian Uggeldahl: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Thomas Lundhede:** Writing – review & editing, Writing – original draft, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization. **Jette Bredahl Jacobsen:** Writing – review & editing, Writing – original draft, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization. **Søren Bøye Olsen:** Writing – review & editing, Writing – original draft, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization.

[10] Exemplified in Johnston and Zawojska (2020) by the argument that changes (and the difference) in cardinal numbers (e.g., 2000 or 20,000 birds) might seem large in the absence of relative information (e.g., much less than 1 % of the population or less than 1 % of the population)

[11] Tested with models including income dummies for the effects of *imp*. These results are omitted from this paper for the sake of brevity but are available on request.

Acknowledgment

The authors gratefully acknowledge funding from the Independent Research Fund Denmark project "Developing and testing a new measure for including biodiversity in economic modelling" [Grant no: 0219–00010B].

Conflicts of Interest Statement

The authors of the manuscript RESEN-D-24–00205 titled "The effects of spatial framing and attribute range on the measurement of nonuse values of biodiversity improvements" certify that they have NO affiliations with or involvement in any organization or entity with any financial interest (such as honoraria; educational grants; participation in speakers' bureaus; membership, employment, consultancies, stock ownership, or other equity interest; and expert testimony or patent-licensing arrangements), or non-financial interest (such as personal or professional relationships, affiliations, knowledge or beliefs) in the subject matter or materials discussed in this manuscript.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.reseneeco.2025.101539.

Data availability

Data will be made available on request.

References

Bakhtiari, F., Jacobsen, J.B., Strange, N., Helles, F., 2014. Revealing lay people's perceptions of forest biodiversity value components and their application in valuation method. Glob. Ecol. Conserv. 1, 27–42.
Bartkowski, B., Lienhoop, N., Hansjürgens, B., 2015. Capturing the complexity of biodiversity: a critical review of economic valuation studies of biological diversity. Ecol. Econ. 113, 1–14.
Bateman, I.J., Day, B.H., Jones, A.P., Jude, S., 2009. Reducing gain–loss asymmetry: a virtual reality choice experiment valuing land use change. J. Environ. Econ. Manag. 58 (1), 106–118.
Bishop, R.C., Boyle, K.J., 2019. Reliability and validity in nonmarket valuation. Environ. Resour. Econ. 72, 559–582.
Bonnichsen, O., Olsen, S.B., 2016. Correcting for non-response bias in contingent valuation surveys concerning environmental non-market goods: an empirical investigation using an online panel. J. Environ. Plan. Manag. 59 (2), 245–262.
Boyd, J., Ringold, P., Krupnick, A., Johnston, R.J., Weber, M.A., Hall, K., 2016. Ecosystem services indicators: Improving the linkage between biophysical and economic analyses. Int. Rev. Environ. Resour. Econ. 8 (3–4), 359–443.
Börger, T., Glenk, K., Meyerhoff, J., Rehdanz, K., 2025. Mitigating cost-vector effects in stated choice experiments using cheap talk and opt-out reminders. J. Assoc. Environ. Resour. Econ. 12 (3), 701–737.
Boyd, J., Johnston, R.J., Ringold, P., 2023. Biophysical measures to support analysis and communication of existence values. Int. Rev. Environ. Resour. Econ. 17 (2–3), 153.
Carson, R.T., 2012. Contingent valuation: a practical alternative when prices aren't available. J. Econ. Perspect. 26 (4), 27–42.
Czajkowski, M., Buszko-Briggs, M., Hanley, N., 2009. Valuing changes in forest biodiversity. Ecol. Econ. 68 (12), 2910–2917.
Faccioli, M., Glenk, K., 2022. More in good condition or less in bad condition? Valence-based framing effects in environmental valuation. Land Econ. 98 (2), 314–336.
Farnsworth, K.D., Adenuga, A.H., De Groot, R.S., 2015. The complexity of biodiversity: a biological perspective on economic valuation. Ecol. Econ. 120, 350–354.
Gassert, F., Mazzarello, J., Hyde, S. (2022). Global 100m Projections of Biodiversity Intactness for the years 2017-2020. Technical White Paper. Available at: ⟨https://ai4edatasetspublicassets.blob.core.windows.net/assets/pdfs/io-biodiversity/Biodiversity_Intactness_whitepaper.pdf⟩.
Glenk, K., Meyerhoff, J., Akaichi, F., Martin-Ortega, J., 2019. Revisiting cost vector effects in discrete choice experiments. Resour. Energy Econ. 57, 135–155.
Glenk, K., Johnston, R.J., Meyerhoff, J., Sagebiel, J., 2020. Spatial dimensions of stated preference valuation in environmental and resource economics: methods, trends and challenges. Environ. Resour. Econ. 75, 215–242.
Glenk, K., Meyerhoff, J., Colombo, S., Faccioli, M., 2024. Enhancing the face validity of choice experiments: a simple diagnostic check. Ecol. Econ. 221, 108160.
Hanley, N., Spash, C., Walker, L., 1995. Problems in valuing the benefits of biodiversity protection. Environ. Resour. Econ. 5, 249–272.
Hanley, N., MacMillan, D., Patterson, I., Wright, R.E., 2003. Economics and the design of nature conservation policy: a case study of wild goose conservation in Scotland using choice experiments. Anim. Conserv. Forum 6 (2), 123–129.
Hausman, J., 2012. Contingent valuation: from dubious to hopeless. J. Econ. Perspect. 26 (4), 43–56.
Hess, S., Palma, D., 2019. Apollo: A flexible, powerful and customisable freeware package for choice model estimation and application. J. Choice Model. 32, 100170.
Hess, S., Train, K.E., Polak, J.W., 2006. On the use of a Modified Latin Hypercube Sampling (MLHS) method in the estimation of a Mixed Logit model for vehicle choice. Transp. Res. Part B Methodol. 40 (2), 147–163.
Holmes, T.P., Adamowicz, W.L., Carlsson, F., 2017. Choice experiments. A Prim. Nonmarket Valuat. 133–186.
Hudson, L.N., Newbold, T., Contu, S., Hill, S.L.L., Lysenko, I., De Palma, A., Phillips, H.R.P., Alhusseini, T.I., Bedford, F.E., Bennett, D.J., 2017. The database of the PREDICTS (projecting responses of ecological diversity in changing terrestrial systems) project. Ecol. Evol. 7 (1), 145–188.
Jacobsen, J.B., Thorsen, B.J., 2010. Preferences for site and environmental functions when selecting forthcoming national parks. Ecol. Econ. 69 (7), 1532–1544.
Jacobsen, J.B., Boiesen, J.H., Thorsen, B.J., Strange, N., 2008. What's in a name? The use of quantitative measures versus 'Iconised'species when valuing biodiversity. Environ. Resour. Econ. 39, 247–263.
Jacobsen, J.B., Lundhede, T.H., Martinsen, L., Hasler, B., Thorsen, B.J., 2011. Embedding effects in choice experiment valuations of environmental preservation projects. Ecol. Econ. 70 (6), 1170–1177.
Johnston, R.J., Duke, J.M., 2009. Willingness to pay for land preservation across states and jurisdictional scale: implications for benefit transfer. Land Econ. 85 (2), 217–237.
Johnston, R.J., Zawojska, E., 2020. Relative versus absolute commodity measurements in benefit transfer: consequences for validity and reliability. Am. J. Agric. Econ. 102 (4), 1245–1270.

Johnston, R.J., Segerson, K., Schultz, E.T., Besedin, E.Y., Ramachandran, M., 2011. Indices of biotic integrity in stated preference valuation of aquatic ecosystem services. Ecol. Econ. 70 (11), 1946–1956.
Johnston, R.J., Schultz, E.T., Segerson, K., Besedin, E.Y., Ramachandran, M., 2012. Enhancing the content validity of stated preference valuation: the structure and function of ecological indicators. Land Econ. 88 (1), 102–120.
Johnston, R.J., Boyle, K.J., Adamowicz, W., Bennett, J., Brouwer, R., Cameron, T.A., Hanemann, W.M., Hanley, N., Ryan, M., Scarpa, R., 2017. Contemporary guidance for stated preference studies. J. Assoc. Environ. Resour. Econ. 4 (2), 319–405.
Kahneman, D.(2002). .Maps of bounded rationality: A perspective on intuitive judgement and choice.
Kling, C.L., Phaneuf, D.J., Zhao, J., 2012. From Exxon to BP: has some number become better than no number? J. Econ. Perspect. 26 (4), 3–26.
Konisky, D., 2011. Public preferences for environmental policy responsibility. Publius J. Fed. 41 (1), 76–100.
Laurila-Pant, M., Lehikoinen, A., Uusitalo, L., Venesjärvi, R., 2015. How to value biodiversity in environmental management? Ecol. Indic. 55, 1–11.
Levin, G., Iosub, C.-I., Jepsen, M.R.(2017). .Basemap02: Technical documentation of a model for elaboration of a land-use and land-cover map for Denmark.
Logar, I., Brouwer, R., 2018. Substitution effects and spatial preference heterogeneity in single-and multiple-site choice experiments. Land Econ. 94 (2), 302–322.
Luisetti, T., Bateman, I.J., Turner, R.K., 2011. Testing the fundamental assumption of choice experiments: are values absolute or relative? Land Econ. 87 (2), 284–296.
Mace, G.M., 2005. An index of intactness. Nature 434 (7029), 32–33.
Mace, G.M., 2014. Biodiversity: its meanings, roles and status. Nature in the Balance: The Economics of Biodiversity. Oxford University Press, New York, pp. 35–56.
Marshall, E., Wintle, B.A., Southwell, D., Kujala, H., 2020. What are we measuring? A review of metrics used to describe biodiversity in offsets exchanges. Biol. Conserv. 241, 108250.
Martin, P.A., Green, R.E., Balmford, A., 2019. The biodiversity intactness index may underestimate losses. Nat. Ecol. Evol. 3 (6), 862–863.
McFadden, D. (1973). *Conditional logit analysis of qualitative choice behavior*.
Meyerhoff, J., Liebe, U., 2009. Status quo effect in choice experiments: Empirical evidence on attitudes and choice task complexity. Land Econ. 85 (3), 515–528.
Meyerhoff, J., Oehlmann, M., Weller, P., 2015. The influence of design dimensions on stated choices in an environmental context. Environ. Resour. Econ. 61, 385–407.
Newbold, T., Hudson, L.N., Arnell, A.P., Contu, S., De Palma, A., Ferrier, S., Hill, S.L.L., Hoskins, A.J., Lysenko, I., Phillips, H.R.P., 2016. Has land use pushed terrestrial biodiversity beyond the planetary boundary? A global assessment. Science 353 (6296), 288–291.
Olsen, S.B., Jensen, C.U., Panduro, T.E., 2020. Modelling strategies for discontinuous distance decay in willingness to pay for ecosystem services. Environ. Resour. Econ. 75, 351–386.
Paternoster, R., Brame, R., Mazerolle, P., Piquero, A., 1998. Using the correct statistical test for the equality of regression coefficients. Criminology 36 (4), 859–866.
Pearce, D., 2001. Valuing biological diversity: issues and overview. OECD: Valuation of Biodiversity Benefits; Selected Studies. OECD, Paris.
Pereira, H.M., Ferrier, S., Walters, M., Geller, G.N., Jongman, R.H.G., Scholes, R.J., Bruford, M.W., Brummitt, N., Butchart, S.H.M., Cardoso, A.C., 2013. Essential biodiversity variables. Science 339 (6117), 277–278.
Purvis, A., Newbold, T., De Palma, A., Contu, S., Hill, S.L.L., Sanchez-Ortiz, K., Phillips, H.R.P., Hudson, L.N., Lysenko, I., Börger, L., 2018. Modelling and projecting the response of local terrestrial biodiversity worldwide to land use and related pressures: the PREDICTS project. In: Advances in ecological research, 58. Elsevier, pp. 201–241.
Rolfe, J., Windle, J., 2008. Testing for differences in benefit transfer values between state and regional frameworks. Aust. J. Agric. Resour. Econ. 52 (2), 149–168.
Scheufele, D.A., & Iyengar, S. (2014). *The state of framing research: A call for new directions*.
Scholes, R.J., Biggs, R., 2005. A biodiversity intactness index. Nature 434 (7029), 45–49.
Schultz, E.T., Johnston, R.J., Segerson, K., Besedin, E.Y., 2012. Integrating ecology and economics for restoration: using ecological indicators in valuation of ecosystem services. Restor. Ecol. 20 (3), 304–310.
Scott, A., 2002. Identifying and analysing dominant preferences in discrete choice experiments: an application in health care. J. Econ. Psychol. 23 (3), 383–398.
Spencer-Cotton, A., Kragt, M.E., Burton, M., 2018. Spatial and scope effects: valuations of coastal management practices. J. Agric. Econ. 69 (3), 833–851.
Strange, N., Ermgassen, S. zu, Marshall, E., Bull, J.W., Jacobsen, J.B., 2024. Why it matters how biodiversity is measured in environmental valuation studies compared to conservation science. Biol. Conserv. 292, 110546. https://doi.org/10.1016/j.biocon.2024.110546.
Train, K., Weeks, M., 2005. Discrete choice models in preference space and willingness-to-pay space. Applications of simulation methods in environmental and resource economics. Springer, pp. 1–16.
Train, K.E., 2009. Discrete choice methods with simulation. Cambridge university press.
Uggeldahl, K., Olsen, S.B., Lundhede, T., Jacobsen, J.B., 2025. Public Perceptions of Biodiversity and the Value of its Conservation. Ecol. Econ. 236, 108681.
UN (1992). Convention on biological diversity. *Treaty Collection*.
Van Bueren, M., Bennett, J., 2004. Towards the development of a transferable set of value estimates for environmental attributes. Aust. J. Agric. Resour. Econ. 48 (1), 1–32.
Vossler, C.A., Dolph, C.L., Finlay, J.C., Keiser, D.A., Kling, C.L., Phaneuf, D.J., 2023. Valuing improvements in the ecological integrity of local and regional waters using the biological condition gradient. Proc. Natl. Acad. Sci. 120 (18).
Walker, J., 2002. Mixed logit (or logit kernel) model: dispelling misconceptions of identification. Transp. Res. Rec. 1805 (1), 86–98.
Walker, J.L., Wang, Y., Thorhauge, M., Ben-Akiva, M., 2018. D-efficient or deficient? A robustness analysis of stated choice experimental designs. Theory Decis. 84, 215–238.
Weller, P., Elsasser, P., 2018. Preferences for forest structural attributes in Germany–evidence from a choice experiment. For. Policy Econ. 93, 1–9.