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Disentangling distance and country effects on the value of conservation across national borders

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Abstract

Coordination of conservation policies and conservation actions between countries is expected to reduce overall costs and increase effectiveness. It rests on the assumption that, as a global public good, the provision of biodiversity conservation is independent of geographical and political jurisdictions. However, from a welfare economic perspective this assumption requires testing and justification. Indeed, distance may matter, as may the country of provision. This study applies a choice experiment to estimate individuals' marginal willingness to pay for comparable biodiversity conservation measures and outcomes across country borders, and with different distances from their place of residence to conservation locations in Denmark and in Southern Sweden. The case is designed to distinguish the effect of distance from the effect of country of residence versus country of provision. We find a clear and distinguishable effect of both location and country of provision. We find distance-related attributes to reflect bridge tolls and per-kilometre transport costs, and Swedes and Danes to

prefer provision in their own country, over provision in the neighbouring country. The results of this study may be useful in discussing cooperation on regional and even global biodiversity conservation efforts.

Keywords: Beech forests, choice experiment, Denmark, distance, ,environmental valuation, , international policy coordination, Sweden, transboundary valuation.

1. Introduction

The continued loss of biodiversity at the global scale has prompted national and international actions and policies targeting international coordination of efforts (e.g. Natura 2000[1], Rio summit[2], CBD2010[3], MA[4] 2005). Despite such efforts, the rate of biodiversity loss does not appear to be slowing (Butchart et al. 2010). Many countries have not met the targets set by the Convention on Biological Diversity (Convention on Biological Diversity Secretariat 2010, Perrings et al. 2010), and renewed pledges were made at Nagoya, Hyderabad and Pyeongchang (Conference of the parties to the Convention on Biological Diversity 2011, 2012, 2014).

The challenge of migratory species conservation, habitat fragmentation and variation in conservation costs at the continental scale and across countries suggests several advantages of increasing the international coordination of biodiversity conservation. Species conservation efforts at the trans-national scale have in many cases been professed to be more cost effective than independent national planning (Hull et al. 1998; Rodrigues & Gaston, 2002; Strange et al. 2006; Bladt et al. 2009, Moilanen and Arponen 2011). The performance of existing agreements

[1] A network of European protected areas.
[2] Rio de Janeiro, Brazil, 1992
[3] Convention on Biological Diversity‘s (CBD) 2010 Biodiversity Target : http://www.cbd.int/
[4] Millenium ecosystem assessment

in this regard is not clear (Bladt et al. 2009) and the welfare consequences have not previously been addressed.

Several factors have challenged the progress of trans-national agreements, including conflicts with other national priorities, difficulties legitimately incorporating them into national laws (Bennett and Ligthart 2001; Dimitrakopoulos et al. 2004; Paavola 2004; Pinton 2001), and free-riding (Olson 1965; Ostrom 1990).

Global biodiversity conservation may be seen as a public good (Yao et al. 2014), and as such could offer long term benefits at a global scale (Halkos and Perrings 2012) independently of where it is provided. An example pointed out by Perrings and Gadgil (2003) is the option value embedded in the preservation of the global gene pool and its independence of where biodiversity protection is provided. However, the geographical distribution may matter in some situations. Some ecosystem services associated with biodiversity such as recreational benefits or regulatory services have a clear local relevance. Several studies do indeed find values of biodiversity and other environmental goods to be distance dependent (Pate and Loomis 1997; Bateman and Langford 1997, Hanley et al. 2003, Bateman et al. 2006, Schaafsma[5] 2011 Jørgensen et al. 2012; Sutherland and Walsh 1985; Hanley et al. 2003; Loomis 1996; Nielsen et al., 2016).

Often longer distances are associated with provision in other countries than the country of residence of the beneficiaries. This raises the question of whether the values obtained from biodiversity protection depends on the country of provision, or more precisely, whether the

[5]Schaafsma et al. (2013) shows distance decay effects for users and non-users however with mixed results on their relative strength for similar ecosystem services at a local scale.

country of provision is the same as the country of residence of the beneficiaries. Possible explanations for such an effect include sense of ownership or identity (Bateman et al (2002); Hanley et al., 2003) or ethical concerns (Daw et al. 2015) by beneficiaries, e.g. if respondents have a belief system involving an obligation to protect biodiversity conservation in their own country. Further, the respondents could be concerned that access to the good in another country, could be restricted in the future due to limited democratic influence by the individual on policies in other countries. The importance of valuation of environmental goods in other countries than the country of residence of beneficiaries has been investigated (e.g. Horton et al. 2003; Dumalisile et al. 2005; Hoyos et al. 2009; Ressurreição et al. 2012), but to our knowledge it has not previously been separated from a distance effect.

Against this background, the objective of this paper is to shed light on two empirical research questions: Does the value of biodiversity conservation depend on the distance to the site of conservation? Does the value of biodiversity conservation depend on whether the respondent resides in the country in which the biodiversity conservation takes place? To this end we carefully selected the location of our case areas, emphasising that the cultural and natural settings of the case areas should be very similar, while allowing us to separate the two effects of distance to site of provision and country of provision. Thus, we designed a Choice Experiment (CE) valuation study focused on habitat and biodiversity conservation measures in beech (*Fagus sylvatica*) dominated broadleaved forests in Southern Scandinavia. We selected three regions, two in Denmark (Funen and Zealand) and one in Sweden (Scania), where conservation measures would provide outcomes of comparable quality. We take advantage of the fact that the distance between Zealand and conservation sites in Funen is similar to the distance between Zealand and conservation sites in southern Sweden. Both Funen and southern Sweden are separated from Zealand by bridged waters and roughly similar distances.

1.1 Literature review

As a background for our research questions, we reviewed the relevant literature, focusing on studies addressing the linkage between stated preferences for environmental goods, spatial dimensions and nationality. Distance decay models have been applied in a number of stated preference studies to estimate spatial heterogeneity. Sutherland and Walsh (1985) was one of the early studies to show that respondents living further from policy areas have lower estimated marginal WTP. Bateman et al. (2006) provided a theoretical justification for distance decay analysis from a use value perspective (recreational demand), where greater travel distances to a natural resource site implies lower net values, ceteris paribus, due to greater costs of reaching the site. Many studies have applied the basic form of the distance decay model to assess spatial welfare heterogeneity (Abildtrup et al., 2013; Adamowicz et al., 1997; Bateman et al., 2002, 2006; Brouwer et al., 2010; Jørgensen et al., 2013; Loomis, 2000; Meyerhoff, 2013; Morrison and Bennett, 2004; Nielsen et al., 2016; Pate and Loomis, 1997; Rolfe and Windle, 2012; Yao et al., 2014). They do so by applying the postal code of a respondent's mailing address (home or origin point) and a geocoded single point that represents the affected area (the destination point).

However, recent studies used patterns other than simple distance to capture spatial welfare heterogeneity. For instance, Campbell et al. (2009) presented a spatial kriging method, and Johnston and Ramachandran (2014) and Meyerhoff (2013) applied hot (or cold) spot analysis using local indicators of spatial association. Johnston and Ramachandran (2014) investigated spatial welfare distributions using geocoded choice experiment data in a river restoration case. They showed that the common distance decay methods could not capture spatial patterns in

WTP estimates for non-market outcomes. Finally, it has been argued that theoretical distance decay justifications may not apply for non-use value (Bateman et al., 2006; Hanley et al., 2003).

While all these studies have addressed the effect of concepts of distance on welfare measures of environmental changes, they did not investigate if distance effects can be separated from nationality effects with respect to the site of provision. This has particular policy relevance when analysing the value of habitat and biodiversity conservation as a public good in an international context.

The effect of nationality of respondents relative to the country of provision for the environmental good has been addressed in various ways. For example, respondents' nationality was found to be a significant element of WTP for users of the whale-watching experience in an Australian marine park (Davis and Tisdell, 1999). Similarly, Samdin et al. (2010) compared Malaysians and international visitors' preferences and found that the respondents' nationality affected significantly their preferences for protection of the Taman Negara National Park. In a study focused on valuing marine species Ressurreição et al. (2012) found respondent nationality and the degree of attachment to the study site as the main driver of WTP. A study by Carlsson et al. (2012) also showed the effect of respondents' nationality on WTP for a climate change mitigation programme. A somewhat different take is that of Yao et al. (2014), who found a significantly higher WTP for conservation of national symbolic species (Brown Kiwi in New Zealand). Dallimer et al. (2014) showed that people in three different countries (Denmark, Estonia and Poland) were willing to pay significantly more for locally delivered services than for similar types of goods delivered in the two other countries, but did not account for differences in distance between the sites of provision and the respondents' locations. Possible explanations for such effects include sense of ownership or identity (Bateman et al., 2002; Hanley et al., 2003; Dallimer et al., 2014; Dallimer and Strange, 2015; van Houtum and van Naerssen, 2002), ethical concerns (Daw et al., 2015) by beneficiaries, notably if respondents have a belief system involving an obligation to protect biodiversity conservation in their own country, or strict border crossing constraints and differences in welfare (Valasiuk et al, 2017). In general, these studies addressed the nationality effects associated with the countries of provision, when these are far from each other and from the respondents' country of residence and/or have different culture, rules, environment etc.

The contribution of the present paper is to investigate the role of nationality on WTP for biodiversity. The case is two neighbouring countries, sharing a similar environment and easy access between the two countries, allowing for control of distance.

1.2 Hypothesis formulation

Based on the above literature and taking advantage of the spatial layout of our experimental case, we formulate the following null hypotheses:

H1: Distance to the site of biodiversity conservation does not matter for people's WTP for a given policy alternative.

H2: Country of biodiversity conservation provision does not matter for people's WTP for a given policy alternative.

We will test these hypotheses in a model using the pooled sample from all three regions, as well as in models using specific regional sub-samples. Details of the hypothesis test procedure are unfolded along with the econometric model specifications below.

2. Material and methods

2.1 Study area

Respondents were sampled from three locations in Funen, Zealand (both in Denmark) and Scania (in Sweden), and we described how a conservation policy could be implemented in broadleaved forests in each of these regions. Conservation measures included setting forests aside for habitat and biodiversity conservation, and measures enhancing the number of old, dying and dead trees in the forest.

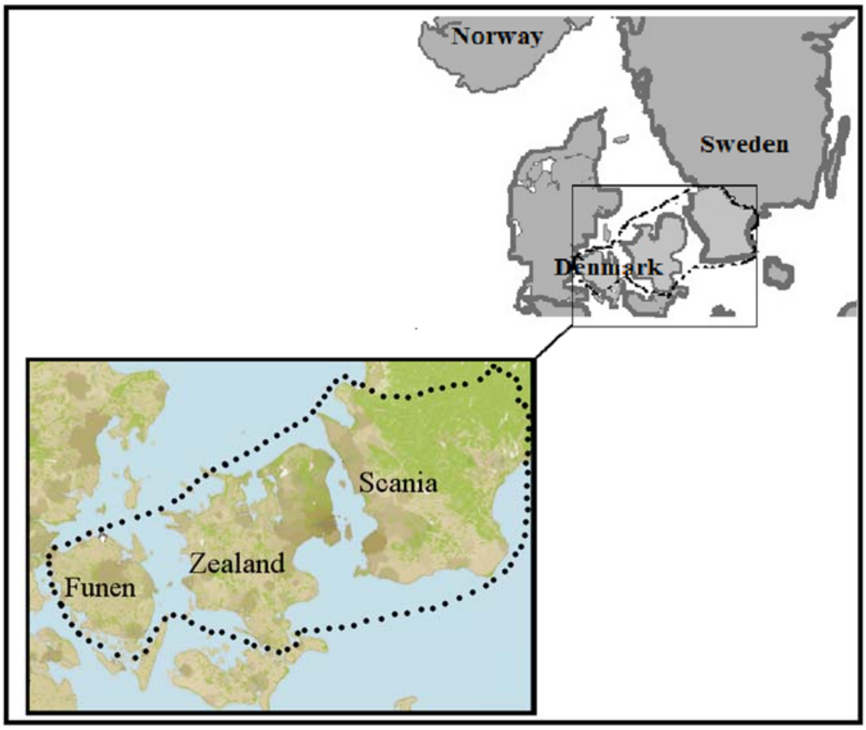

Figure 1: A map of the study area. (The green and brown colours indicate the forest cover and urban areas, respectively. The dashed line indicates the border lines of the three study regions) (source: Kempeneers et al., 2011; Paivinen et al., 2001; Schuck et al., 2002)

Travel distances between Funen and Zealand (within national boundaries), and Zealand and Scania (across national boundaries) are quite similar in range, whereas the distance between Scania and Funen is about double. This design allowed us to separate distance and nationality effects, and take travel costs into consideration. This includes the cost of toll-bridges over the Great Belt (between Funen and Zealand) and the Oresund (between Zealand and Scania), where tolls are similar in magnitude. The broadleaved forests in all three locations have similar conservation potentials and are dominated by beech (*Fagus sylvatica*), but also oak (*Quercus robur*), ash (*Fraxinus excelsior*) and birch (*Betula pendula*).

2.2 Data collection and survey design

Data were collected through an internet-based questionnaire managed by the survey institute 'Analyse Denmark' during July-August 2012. We received 600 completed questionnaires (a 20% response rate) for each of the three locations (1800 in total). Survey respondents were informed that the hypothetical conservation policy presented in the questionnaire would improve biological diversity as well as enhance the natural dynamics of the forests. The proxies used to describe these two attributes were 1) keeping old trees in the forest to age, die and turn into deadwood in the forest through natural decay and 2) increasing the number of species in the areas in focus by improving the living conditions for animals, plants and other organisms.

The selection of these attributes was based on eight different focus groups and several individual interviews, involving more than 50 persons in total across all three regions. We asked participants in the focus groups to think about biodiversity and evaluate various representations of the way they thought about it. The result of this process was that 'Forest species number' and 'Presence of natural dynamics' captured well how people perceived biodiversity. Further details can be found in Bakhtiari et al. (2014). We note that while such measures do not reflect the full complexity of biodiversity, the selected attributes can be considered reasonable approximations of the respondents' perception of the good (cf. guidelines by Johnston et al., 2017).

Respondents were informed that, across the broadleaved forests in the three regions, one could find around 10,000 species in total. Based on the literature (Petersen et al., 2016, 2012) and data from the global biodiversity facility (GBIF: The Global Biodiversity Information Facility, 2017a) on species diversity and conservation the number of species in Denmark was assumed to be approximately 35,000[6]. Of these, around 65 per cent can be found in broadleaf dominated forests, which are the climax ecosystem in much of the regions area. Since the broadleaf dominated forests account for approximately 41percent of the Danish forest area (Johannsen et al., 2013), we assumed that 10.000 species would be a conservative estimate. The number includes vascular plants and vertebrates, although a substantial part of the species are insects, non-vertebrates and fungi. We assume the numbers of species in the case areas to be similar, since the forest ecosystems on which we focus are quite similar in Denmark and Southern Sweden. We should note that we expect the total number of species in the entire Sweden to be higher, as the country reaches into the boreal zone[7]. However, on any given forest area, much fewer species would in general be present and based on the above references and a few as others (e.g. Lawesson et al., 1998), respondents were informed that around 1,000 species would currently be common and abundant in the forest area subject to the

[6] A national report by Ejrnæs et al. (2014) represents an even more extensive data set and confirmed that the total number of species in Denmark is between 30.000-40.000 species.
[7] Estimates of the total number of species in Sweden and their distribution to different groups can be found at the Global Biodiversity Facility (2017b).

conservation policy. Respondents were then presented with alternative attribute levels which would increase the number of abundant species to 1,500 or 2,000 in the forest conservation area. The number thus comprised common, rare, and potentially endangered species including both larger and smaller species groups. Thus, the approach was to look at the diversity of abundant species at the conservation sites regardless of how rare or endangered the species were prior to the conservation effort[8]. This differs from many other studies emphasizing conservation of endangered species (for two Danish studies see Campbell et al., 2014; Jacobsen et al., 2008).

An important additional attribute was the location of policy implementation, which was presented on a four-level scale: The status quo of current forest management in all regions and policy implementation in Funen, Zealand and Scania, respectively. Finally, we included a tax attribute in the form of an increase in the annual income tax caused by the selected policy. Table 1 shows the attributes and attribute levels. The current management' attribute is equal to the lowest level of each of the attributes (bold).

Table 1

Attributes and levels presented to respondents in choice tasks. The current situation is represented by the lower level and is shown in **bold**

Attribute variable	Attribute level
Location of policy area	**(i) no new policy** (ii) Funen (iii) Zealand (iv) Scania

[8] A poorly phrased sentence in the questionnaires' introductory text to the attribute explanation erroneously suggested that the species not included in the 1000 currently common and abundant are all endangered. That is not the case. Some of these are, but others may simply be less abundant or rare. However, the further description of the species attribute described it in terms of increasing the number of common and abundant species in the conservation areas. Furthermore, as the questionnaires all included this same text, it is not expected to affect the core analyses and contribution of this study: The disentangling of distance to and country of provision site.

Number of abundant forest species in area	**(i) 1000** (ii) 1500 (iii) 2000
Presence of natural dynamics in area	(i) **occasionally leaving trees to age, die and decay** (ii) Leaving 7 trees/ hectare (iii) Leaving 15 trees/ hectare
Annual income tax (DKK*/ year)	**0** ,250, 500, 750, 1000, 1250

*1DKK=0.18 USD$ and 0.13 Euro. In Scania, SEK were used and the exchange rate was around 0.85 SEK/DKK.

A previous qualitative study in the same regions (Bakhtiari et al., 2014) showed that income tax was considered an acceptable way of financing biodiversity conservation policies among most Danish and Swedish citizens. The questionnaire emphasised that, to avoid free riding, and as a result of the coercive payment vehicle, all tax payers in both countries would contribute. In addition, the survey made it clear that this amount would be additional to current tax payments. Finally, we inserted a reminder about the respondents' budget constraints before the choice tasks. Additionally, the questionnaire included questions about the respondents' visiting frequency to different sites, distance to forests visited, and various questions on forest activities.

After the data were collected, we constructed additional variables , which vary over alternatives and individuals. These included the distance from each respondent's mid-point postal code area to the attribute policy site in each of the three regions, a dummy variable for the number of bridges between each respondent's location and each policy site, and a dummy variable for whether or not the policy site was in the respondent's own country. Note that these derived variables are not a part of the factorial experimental design, but rather an addition to the socio-demographic variables of the individuals. We note that the correlation between the bridge dummies (which are 0, 1 or 2) and the distance measures is very low due to respondents being scattered widely within the three regions.

We applied a fractional factorial design and optimising d-efficiency[9] for a multinomial logit model (Scarpa and Rose, 2008), with zero priors and assuming preferences for all attribute levels except price to be randomly distributed. The design included 24 alternatives, which were divided into four blocks. Some of the choice sets included dominated alternatives implying that respondents would not do any real trade-off and these were changed manually. The questionnaires were translated into Danish and Swedish. They were tested through focus groups and a pilot study, and benefited from participants' feedback regarding the wording of the questions.

2.3 Econometric specifications

We tested our hypotheses by estimating a utility function for the conservation improvement as perceived by the respondents in our pooled dataset, and each of the study locations. The utility function for our pooled dataset was described as:

$$U_{ij} = (\mathrm{ASC}_j + \beta_{1i} Foreign_{ij} + \beta_{2i} Distance_{ij} + \beta_{3i} Bridge_{ij} + \beta_{4i} Biodiversity1500_j + \beta_{5i} Biodiversity2000_j + \beta_{6i} Leaving7trees/ha_j + \beta_{7i} \mathrm{Leaving15trees/ha}_j + \beta_{8i} Tax_j + \eta_j + \varepsilon_{ij} \qquad [1]$$

Where *i* =individual and *j* =alternative. The deterministic part of the utility is captured by the *β*'s (the parameters for the attributes) and the related attribute and variable levels. An

[9] The software Ngene was used (ChoiceMetrics, 2012)

Alternative Specific Constant (ASC) was specified for the status quo alternative to capture the systematic component of a potential status quo effect (Scarpa et al., 2005). The Gumbel-distributed random error term of the random utility function is denoted ε_{ij}. An error component η_j was added to the model, and we assumed this component to be present only for the status quo alternative. Consequently, the utility for the status quo alternative was simply the ASC, the error component, and the standard error term (see also Greene and Hensher, 2007; Ferrini and Scarpa, 2007; Scarpa et al., 2005). *Foreign* was a dummy variable (coded as 1 if the provision in alternative *j wa*s not located in respondent *i*'s country). *Distance* measured the logarithm of the shortest distance from respondent *i*'s residence (midpoint of the postal code) to the nearest entrance point (bridge or ferry) to the region of biodiversity conservation present in the alternative *j*. *Bridge* was a variable for how many toll bridges must be crossed to get from respondent *i's* residence to the region *j*. Finally, *Biodiversity1500, Biodiversity2000, Leaving7* and *Leaving15 trees/ha* captured the respective attribute levels of the alternatives. *Tax* referred to the payment vehicle, which was annual income tax. In equation 1, our hypothesis H1 would be rejected if the parameters for *Distance* and *Bridge* were significantly different from zero, and H2 would be rejected if *Foreign* was significantly different from zero.

In the case of respondents from Zealand, the utility of respondent *i* from Zealand in case of policy alternative *j* was:

$$U_{Zealand,ij} = ASC_j + \beta_{Zealand,2i}(Funen)_j + \beta_{Zealand,3i}(Scania)_j + \beta_{Zealand,4i}(Biodiversity1500)_j + \beta_{Zealand,5i}(Biodiversity2000)_j + \beta_{Zealand,6i}(Leaving7\ trees/ha)_j + \beta_{Zealand,7i}(\mathrm{Leaving\ 15\ trees/ha})_j + \beta_{Zealand,8i}(Tax)_j + \eta_j + \varepsilon_{ij} \quad [2]$$

, where the locations *(Funen)$_j$* and *(Scania)$_j$* addressed the utility of a resident in Zealand for implementing forest protection policy *j* in Funen or Scania, respectively, as opposed to implementation in Zealand. In equation 2, H1 was rejected if $\beta_{Zealand,2i}$ and $\beta_{Zealand,3i}$ was significantly different from zero. H2 was rejected if $\beta_{Zealand,3i}$ was significantly different from $\beta_{Zealand,2i}$. Similarly, when respondents were from Scania, H1 was rejected if the parameters for Zealand and Funen were significantly different from zero, and larger for Zealand than for Funen. Finally, H2 could not be tested for Scania and Funen as distance and country of provision could not be separated.

The preference models were estimated using a random parameter error component logit model (RPL) (Ben-Akiva et al., 2001; Brownstone and Train, 1998; Revelt and Train, 1998; Scarpa et al., 2005). In this model the utility of a good was described as a function of its attributes, and people chose among composite goods by evaluating their attributes. According to Train (2003), the mixed logit probabilities could be described as integrals of the standard conditional logit function evaluated at different β's, with a density function as the mixing distribution. Thus, while the utility coefficients varied from one individual to another , they were constant over the *N* choice occasions for each individual, and we accounted for this panel structure. The probability density was specified to be normal and the unconditional probability of choosing a sequence of alternatives k was defined as:

$$\Pr(ik) = \int \left(\prod_{n=1}^{N} \left[\frac{\exp^{\lambda_{ikn}\beta_i' x_{ik_n n}}}{\sum_{j}^{J} \exp^{\lambda_{ikn}\beta_j' x_{ijn}}} \right] \right) \phi(\beta \mid b, W)\, d\beta \qquad [3]$$

The ASC and error terms from eq. [1] were left out for simplicity. β' was a vector of all betas, and the distribution function for β was $\phi(\beta \mid b, W)$, with mean *b* and covariance *W*. The analyst chooses the appropriate distribution for each parameter in *β*.

The model allowed us to estimate the parameters up to a scale factor, λ, which is inversely related to the error variance. Note that λ may differ between subsamples, and was estimated using scale tests (for example, see Bierlaire, 2003).

3. Results

We first report in Table 2 a comparison of the socio-demographics of the three samples with those of the corresponding regions. The three sub-samples slightly underrepresented respondents in high and low income groups, and generally had higher education level, compared with the population in the respective regions. With respect to age and gender the samples were representative for their respective populations.

Table 2
Comparison of sociodemographic variables in the three samples with those of the corresponding regional populations using data from the Swedish Statistical office (2012) and Statistics Denmark (2012). Stars indicate significance of Chi-Square tests for differences in the distribution between the sample and the corresponding population,

		Funen sample %	**Funen Population**	**Zealand sample**	**Zealand population**	**Scania sample**	**Scania population**
Population share	Test of same distribution	NS		NS		NS	
	Female	50	50	50	51	48	50
Age	Test of same distribution	NS		NS		NS	

	Average	44	43	42	41	44	45
Highest achieved education	Test of same distribution	***		***		***	
	Primary	15	20	15	18	17	24
	Vocal-secondary school	18	19	16	15	40	42
	Graduated-higher (university degree)	17	11	18	17	43	34
Household Income	Test of same distribution	***		***		***	
	<200,000	20	22	24	27	21	41
	200,00-399,999	36	37	29	31	40	42
	400,00-599,999	18	17	18	15	12	11
	600,00-799,999	14	13	13	12	18	3
	800,000-999,999	5	4	7	4	6	1
	>1million	6	7	10	11	3	2

***P<0.001.

Table 3

The number of respondents who checked the different visit frequency alternatives for the three different locations.

	Respondents in Scania			Respondents in Zealand			Respondents in Funen		
	Forest in Funen	**Forest in Zealand**	**Forest in Scania**	**Forest in Funen**	**Forest in Zealand**	**Forest in Scania**	**Forest in Funen**	**Forest in Zealand**	**Forest in Scania**
More than 3 times a week	0	0	72	0	9	0	0	0	0
1-3 times a week	2	18	88	0	54	3	78	0	0
1-3 times a month	0	0	111	0	30	2	0	0	0
1-12 times a year	1	21	231	8	384	28	307	50	5
Once a year	2	21	39	51	49	147	135	57	14
Less than once a year	67	90	41	391	55	120	70	199	97

Table 4

The number of respondents in each population who never visited a forest in one of the different locations and mean and standard deviation of distance to the nearest forest in own region as well as average distance to the entry points of other regions.

Residents :	Forest areas in Funen	Distance to the nearest forest in Funen	Forest areas in Zealand	Distance to the nearest forest in Zealand	Forest areas in Scania	Distance to the nearest forest in Scania

Funen residents whom never visited forest	10	2.5 *s.d (1.3)*	294*	123 *s.d (27.5)*	484	230 *s.d (37.3)*
Zealand residents whom never visited forest	150	94 *s.d (19)*	19	3 *s.d (1.5)*	300	101 *s.d (33.8)*
Scania residents whom never visited forest	528	225 *s.d (20)*	450	62 *s.d (30)*	18	3 *s.d(1.6)*

* For example 249 residents of Funen said they never visited a forest in Zealand

The descriptive statistics reported in Table 3 and 4 shows a clear distance effect. Specifically, the least frequent types of visits are those of respondents living in Scania who visit Funen and vice versa, which represents the longest possible trips.

3.1 Estimating utility parameters

When asking three different geographic sub-populations about their preferences for policy actions in three different regions, there was a risk that respondents would not relate to the different choices involving the three regions with the same degree of confidence and precision. Therefore, we tested for difference in scale between the three subsamples (that is, Funen, Zealand and Scania) following the approach suggested by Swait and Louviere (1993). We found respondents in Scania had a statistically smaller scale (0.33), and corrected for this accordingly (see Ben-Akiva and Lerman, 1985; Hensher and Greene, 2003; Louviere et al., 2000; Train, 2003). Based on the log likelihood, pseudo-R^2 and AIC, the Random Parameter Logit model including error component (RPL+EC) was best supported by the data. For distance we assumed a log-normal distribution as we expected a non-positive preference for all people. For all other random parameters, we used a normal distribution. We find these assumptions to provide the best model fit. The estimated parameters and derived WTP based on the pooled data set are shown in Table 5. The standard errors of the WTP were estimated using the Delta method (Hole, 2007). The environmental attributes and the error components were significant and with expected positive signs. The tax coefficient was negative as expected. The alternative specific constant (ASC) was positive and significant. The attributes '*Distance*' and '*Foreign*' were both significant and negative, the implication being that *H1* and *H2* were both rejected.

Table 5.

Parameters and WTP estimates in an RPL+EC model using the pooled data set.

Attributes	**Parameters (standard error)**	**WTP (DKK[1]/year) (95% confidence interval)**

Foreign location	β	-0.27*** (0.08)	-152 (-252.93 ; -51.72)
	σ	0.97*** (0.12)	-
Distance[2]	β	-0.004*** (0.001)	-2 (-3.0 ; -1.2)
	σ	0.01*** (0.006)	-
Bridge	β	-0.70*** (0.06)	-397 (-462 ; -332)
	σ	0.99*** (0.07)	
1500 species	β	0.88*** (0.09)	493 (390;597)
	σ	1.02*** (0.14)	
2000 species	β	0.81*** (0.06)	452 (383; 522)
	σ	0.7*** (0.08)	-
Natural dynamic (leaving 7 deadwood/ha)	β	0.25*** (0.09)	142 (43;243)
	σ	0.09 (0.9)	-
Natural dynamic (leaving 15 deadwood/ha)	β	0.35*** (0.07)	201 (119;284)
	σ	1.02 (0.07)	-
Tax	β	- 0.002*** (>0.001)	-
ASC	2.24*** (0.19)		
AIC/N	1.47		
ρ 2	0.34		
LL	-7710.86		
η (error component)	4.42***		

[1]DKK =0.18 USD, [2]Distance is a logarithmic variable which is measured in kilometres, ***statistically significant at the 1‰ level, ** at the 1% level and * at the 5% level. .

Table 6 shows the results for the subsamples of respondents in Zealand and Scania. We found that residents in Zealand had the largest WTP for a policy implementation in their own location (Zealand) compared with other locations, thus rejecting H1. The WTP for people living in Zealand (Denmark) for a policy implemented in Funen was higher than for implementing a similar alternative in Scania (Sweden), thereby also rejecting H2. Since the value of the dummy variable 'Bridge' was 1 for both alternative locations, it cannot be included in this model.

We note that the WTP for 2000 species is not significantly higher than WTP for the 1500 species level, which suggests a weak scope-sensitivity. We discuss this later. Respondents in Scania also preferred biodiversity conservation in Scania over the two other locations, and they preferred Zealand to Funen, which implies rejection of H1. We were not able to test H2 explicitly for this sub-sample, because of confounding factors, as explained above: Zealand is both further away and a different country. For respondents from Funen, we also could not separate distance from country effect as they share country with Zealand. Therefore, these results are not shown. However, we did find that people in Funen also value local provision more than provision at other sites.

Table 6

Parameter estimates in two location samples using RPL+EC. Parameters in **bold** format relate to hypothesis 1 and 2, respectively. The location attribute levels are estimated relative to the omitted location level identical to the respondent's own location.

Geographical locations: / Attributes		Zealand		Scania	
		Parameters (Standard error)	Marginal WTP (DKK/year) (95% confidence interval)	Parameters (Standard error)	WTP (DKK/year) (95% confidence interval)
Location(Scania)	β	-1.87*** (0.12)	-943 [a] (-1101.10 ; -885.64)	-	-
	σ	1.1*** (0.17)	-	-	-
Location(Funen)	β	-1.40*** *(0.09)*	-706 [a] (-889 ; -598)	-1.54*** (0.10)	-1155[a] (-1290 ;-1022)
	σ	0.9 *** (0.90)	-	0.71*** (0.13)	-
Location(Zealand)	β	-	-	-1.27*** (0.10)	-953[a] (-1017 ; -838)
	σ	-	-	0.51*** (0.19)	-
1500 species	β	0.88 *** (0.14)	457 (311 ;604)	0.69*** (0.08)	543 (313 ;773)
	σ	0.62 *(0.37)*	-	0.52*** (0.14)	-
2000 species	β	0.80*** (0.10)	400 (287 ;512)	0.72*** (0.14)	514 (384 ;643)
	σ	0.9*** (0.13)	-	0.94*** (0.24)	-
Natural dynamic(leaving 7 deadwood/ha)	β	0.19* (0.16)	99 (-65 ;265)	0.05* (0.12)	41 (-140 ;222)
	σ	0.03 (1.9)	-	0.14*** (0.83)	
Natural dynamic(leaving 15 deadwood/ha)	β	0.38*** (0.13)	190 (61 ;318)	0.36*** (0.09)	270 (130 ;409)
	σ	0.93*** (0.14)	-	0.66*** (0.12)	
ASC [b]	β	2.64*** (0.31)	1254 (950; 1558)	2.87*** (0.34)	2145.91 (1675; 2617)
Tax	β	-0.0019*** (0.0009)		-0.001*** (0.0008)	
η (error component)	β	4.43 ***		4.92***	
ρ 2		0.35		0.32	
LL		-2577.6		-2145.8	
AIC/N		1.43		1.49	

***Statistically significant at the 1‰ level, ** at the 1% level and * at the 5% level.

a =WTP amount for policy implementation in Scania and Funen as opposed to Zealand. b =ASC is the utility from status quo (doing nothing) as opposed to policy implementation in respondents own location.

To test if the location was more important for some attributes than for others, we created an interaction variable of the location attributes with the biodiversity and natural dynamic variables. The interaction turned out not to be statistically significant and hence we do not show these results. We have chosen an RPL model, which allow us to capture preference heterogeneity in a flexible way, while maintaining a clear focus on the parameters of interest for our research questions. For the same reason we have not presented models with numerous interaction terms involving socio-demographic variables. However, we did test it and the above results were not sensitive to including such interaction terms.

4. Discussion

The objective of this paper is to shed light on two empirical research questions: Does the value of biodiversity conservation depend on distance to the site of conservation? Does the value of biodiversity conservation depend on whether the respondent resides in the country in which the biodiversity conservation takes place? To answer these questions we designed and implemented a choice experiment, where the population in two countries evaluated comparable measures in both of these countries. This allows us to assess and separate the 'travel distance effect' from the 'country of provision' effect. The results show a significant 'travel distance effect', measured as distance from the respondents' residence vis-à-vis the policy site, as well as a 'country of provision effect'. In addition, respondents have a positive and larger utility for biodiversity improvements in their own country and region. Thus, we can reject the null hypothesis that distance to the site and the country of provision do not matter for welfare measures. With regard to the credibility and external validity of the result, we note that the WTP estimates for distance (estimated at approximately 2 DKK per km) are quite consistent with the travel cost per km in Denmark and Sweden, as assessed by the tax authorities[10], which are (is?) in the range of 2-4 DKK per km. In addition, the WTP for 'Bridge' toll corresponds reasonably well with the real cost of a return ticket, which drivers pay to cross the bridge. Thus, the travel cost-related parameters are consistent with the cost of visiting the forests in the other regions typically once per year, a frequency which is well in accordance with the observed frequencies in the samples of respondents (cf. Table 2). It is worth noting that these variables are likely linked to the respondents' expected direct use values of biodiversity conservation in the different policy sites.

For the Zealand subsample we also tested whether respondents preferred Funen or Scania for forest protection implementation. Based on the marginal effects of location attributes, we again concluded that both distance and nationality of country of provision matter. The fact that the country of provision has a separate effect, once distance effects have been corrected for, suggests that also non-use values derived from biodiversity conservation may be sensitive to the country of provision. For the second subsample, respondents in Scania assessed forest protection policy in Funen and Zealand (these localities only differed in terms of travel distance to the policy location). We found that respondents in Scania had a larger marginal WTP for implementing a forest protection policy in Zealand, compared with Funen. It is worth noting that the majority of the respondents from Scania stated that they never had visited a forest in Funen or Zealand (87% and 75%, respectively).

With respect to the main attributes of the experiment, the WTP measures and patterns are as expected. This includes the levels of WTP for enhanced number of abundant species. Compared

[10] See for Denmark: http://www.skat.dk/SKAT.aspx?oId=2064181, and for Sweden http://www.skatteverket.se/privat/svarpavanligafragor/beloppprocentsatser/privatbeloppfaq/bilavdraghurstortardet.5.10010ec103545f243e8000220.html

to related studies that specifically target threatened species (e.g. Jacobsen et al., 2008; Campbell et al., 2014) we find the WTP per species to be lower and with a weak scope sensitivity. Jacobsen et al. (2012) evaluated Danes' WTP for enhancing populations of common or rare species relative to their WTP for saving threatened species. They showed that people have a lower WTP, potentially even decreasing with scope, for enhancing the populations of common species, compared to saving threatened species. Our case study here addresses policies that make more species abundant, but not specifically and only threatened species. Thus our results seem well in line with comparable results in the literature, namely that WTP per species is somewhat lower because we are valuing common and not endangered species.

4.1 Possible reasons and consequences

Other studies have found that nationality and degree of attachment to locations affect preferences (e.g., Brouwer et al., 2010; Carlsson et al., 2012; Dallimer et al., 2014; Davis and Tisdell, 1998; Hanley et al., 2003; Ressurreição et al., 2012; Samdin et al., 2010; Yau et al., 2014; Valasiuk et al., 2017). Conversly Dumalisile et al. (2005), Horton et al. (2003), and Jin et al. (2010) did not find any significant effect associated with the degree of attachment to the location of environmental improvements. However, none of these studies was designed to separate the effect of distance to a site of provision from the effect of the site being in another country. Our results show that, in the current case, biodiversity conservation benefits are not independent of geographical and national jurisdiction (Dallimer and Strange, 2015; Valasiuk et al., 2017). We find that even respondents from Scania who state they have never visited forests in Denmark, have a WTP for biodiversity conservation that decreases with distance to locations in Denmark. The specific effect of the location of provision – when controlling for distance and other travel cost variables – suggests that non-use values may be sensitive to the geographical location of provision. This result adds to findings by Brock and Xepapadeas (2003) and Hanley et al. (2003), who found biodiversity conservation to benefit people at different spatial and temporal scales. A large number of respondents in our study (Table 4) replied that they never or very rarely had visited forest areas in any of the two other locations. This suggests that recreational benefits (direct use values) are not the main reason for the WTP differences across different locations that we find.

Thus, our core result is that values related to biodiversity conservation may be sensitive to country of provision beyond what can be explained by distance as a cost driving element of use. We argue that obtaining separate estimates of use and non-use values from conservation actions may in fact not be valid. An argument that we, and other authors (e.g., Tacconi, 2000) have made is that non-use values may not be independent of place of provision. If they are not, they are also much harder to separate from use values. Simply including interactions with e.g. recreational habits, distance from home to policy site, etc., will not allow for such a separation if people derive higher non-use based utility from knowing that a pristine natural area is closer to them – even if they never intend to use it.

Our findings may have important policy implications for biodiversity conservation across borders and the funding of these. Notably, our findings suggest that it may be more difficult to gather political support for cross-country biodiversity conservation actions even if such coordination could be more cost-efficient. We may speculate as to the reasons why the country of provision matters beyond the distance to site of provision itself. People may find it more acceptable to invest more in conservation in their home country for a number of reasons as discussed by Bateman et al. (2002), Dallimer et al. (2014), and Dallimer and Strange (2015), Hanley et al. (2003), and van Houtum and van Naerssen (2002), which suggest that ownership or spatial identity may be important for some environmental assets, even for non-use value. Indeed the finding here may carry over to other international environmental investment issues like, for example, climate change management measures. specific studies could address this.

Thus, our results add further to the findings and discussions of Perrings and Halkos (2012), who suggested that the optimal level of biodiversity conservation might be expected to vary depending on the spatial scale at which the problem is analysed, and depending on which (national) groups are involved in conservation decisions. We do not engage in specific cost benefit analyses here, but note that previous studies have shown that the opportunity cost of setting aside forest for biodiversity protection (using capital budgeting approaches) is in the range of 200-400€ per ha and year (Jacobsen et al., 2013; Petersen et al., 2016). Another Danish study (Danish Economic Councils, 2012) found that the cost of protecting Danish forest habitats is less than € 7 million annually, or less than €3 per household per year, and hence significantly lower than the WTP measures estimated in the current study, as well as in similar studies (Jacobsen et al., 2008).

4.2 Caveats and further work

Differences in factors such as national income, species richness, pressures on biodiversity, and conservation infrastructure are all likely to be associated with differences in national conservation efforts (Dallimer et al., 2014; Perrings and Halkos, 2012). In our study, all of these factors were assumed comparable at the sub-sample level. Future studies would probably benefit from investigating these issues across a wider range of cases, even if this may imply difficulties in finding a comparable public good to evaluate across cases.

Our study did not consider factors such as trust and power within and across countries. Yet, we acknowledge that they may play a role in public preferences with regard to coordinating conservation efforts across borders (Boarini et al., 2009). In our case, one could speculate, for example, that Swedes would trust their own country (rules, laws, compliance, governance) more than they would trust Denmark (and vice versa), when it comes to deliver on conservation policies. Following Hanley et al. (2003), they may feel more in control of the implementation. Thus, lack of mutual trust among residents from different countries and regions, in relation to designing and implementing a joint coordination programme, could be a reason for the differences observed (Dallimer et al., 2014; Zak and Knack, 2001). In a similar vein, during focus group interviews we found that participants were not willing to pay as much if efforts were to be implemented by an international agency, as they would if their own government engaged in coordinating protection programmes across borders. This suggests a preference for implementation at local scale, which is aligned with what Hanley et al. (2003) and Dallimer et al. (2014) showed. Thus, trust and control issues may warrant further investigation , and may help explain possible individual variation in preferences for local (national) provision.

We argue that the nationality effects may be related to the individuals' willingness to cooperate with people from other nations. The research field on human cooperation is large and beyond the focus of this study, but of relevance are papers discussing trust, reciprocity and cooperation in a cultural perspective (e.g., Boyd and Richerson, 2009). Individuals who perceive themselves as belonging to the same group or social network may be more likely to cooperate (Henrich and Henrich, 2007). National borders may separate cultural and national identity despite the many socio-demographic similarities between Swedes and Danes. In a recent study

Dorrough and Glöckner (2016) found evidence that, in cross-societal cooperation games, knowledge about the other player's nationality matters.

5. Concluding remarks

We believe that the current study of the value of biodiversity conservation successfully distinguished the effect of the distance to site of provision from the effect of the country of provision, with regard to preferences for conservation outcomes. This is novel to the literature. We found distance-related attributes to reflect bridge tolls and per-kilometre transport costs, and found Swedes and Danes to prefer provision in their own country, over provision in the neighbouring country. Denmark and Sweden are neighbouring countries with similar languages, history and cultures. The magnitude of the nationality effect found in our study may therefore be larger, if future studies address the same issue for countries further apart, and countries less similar to each other than Denmark and Sweden. For example, Dallimer et al. (2014) showed that the nationality affect was a significant factor of WTP for residents of Estonia, Denmark and Poland and respondents had higher preference for biodiversity conservation in their own country relative to in other countries. However, their findings did not separate country effects from distance effects. The overall results of this study have relevant policy implications for regional and even global biodiversity conservation efforts. The underlying assumption in most conservation management models is that the benefit of biodiversity conservation is independent of spatial scale, and culture or nationality. Several studies demonstrate the magnitude of cost-efficiency gains of internationally coordinated conservation policies (Bladt et al., 2009; Dallimer and Strange, 2015; Hull et al., 1998; Kark et al., 2009; Moilanen and Arponen, 2011; Rodrigues and Gaston, 2002; Strange et al., 2006). This study stresses that a mere cost-effectiveness focus may disregard important aspects of the allocation of social benefits, and result in loss of significant welfare economic gains. This is of importance for the design of trans-national conservation policies, as not only effectiveness and efficiency concerns need to be considered, but also considerations about welfare distribution across borders. Neglecting these issues may create a mismatch in policy design across borders, where due attention is needed for both the distribution of costs, as well as benefits. Policy proposals may fail to gain wide support if welfare gains are mainly harvested by the population of a specific region.

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