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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 20001, Rio summit2, CBD20103, MA4 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

48 in this regard is not clear (Bladt et al. 2009) and the welfare consequences have not previously
49 been addressed.
50
51 Several factors have challenged the progress of trans-national agreements, including conflicts
52 with other national priorities, difficulties legitimately incorporating them into national laws
53 (Bennett and Ligthart 2001; Dimitrakopoulos et al. 2004; Paavola 2004; Pinton 2001), and free-
54 riding (Olson 1965; Ostrom 1990).
55
56 Global biodiversity conservation may be seen as a public good (Yao et al. 2014), and as such
57 could offer long term benefits at a global scale (Halkos and Perrings 2012) independently of
58 where it is provided. An example pointed out by Perrings and Gadgil (2003) is the option value
59 embedded in the preservation of the global gene pool and its independence of where
60 biodiversity protection is provided. However, the geographical distribution may matter in some
61 situations. Some ecosystem services associated with biodiversity such as recreational benefits
62 or regulatory services have a clear local relevance. Several studies do indeed find values of
63 biodiversity and other environmental goods to be distance dependent (Pate and Loomis 1997;
64 Bateman and Langford 1997, Hanley et al. 2003, Bateman et al. 2006, Schaafsma[5] 2011
65 Jørgensen et al. 2012; Sutherland and Walsh 1985; Hanley et al. 2003; Loomis 1996; Nielsen et
66 al., 2016).
67 Often longer distances are associated with provision in other countries than the country of
68 residence of the beneficiaries. This raises the question of whether the values obtained from
69 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.

70 country of provision is the same as the country of residence of the beneficiaries. Possible
71 explanations for such an effect include sense of ownership or identity (Bateman et al (2002);
72 Hanley et al., 2003) or ethical concerns (Daw et al. 2015) by beneficiaries, e.g. if respondents
73 have a belief system involving an obligation to protect biodiversity conservation in their own
74 country. Further, the respondents could be concerned that access to the good in another
75 country, could be restricted in the future due to limited democratic influence by the individual
76 on policies in other countries. The importance of valuation of environmental goods in other
77 countries than the country of residence of beneficiaries has been investigated (e.g. Horton et al.
78 2003; Dumalisile et al. 2005; Hoyos et al. 2009; Ressurreição et al. 2012), but to our knowledge
79 it has not previously been separated from a distance effect.
80 Against this background, the objective of this paper is to shed light on two empirical research
81 questions: Does the value of biodiversity conservation depend on the distance to the site of
82 conservation? Does the value of biodiversity conservation depend on whether the respondent
83 resides in the country in which the biodiversity conservation takes place? To this end we
84 carefully selected the location of our case areas, emphasising that the cultural and natural
85 settings of the case areas should be very similar, while allowing us to separate the two effects
86 of distance to site of provision and country of provision. Thus, we designed a Choice Experiment
87 (CE) valuation study focused on habitat and biodiversity conservation measures in beech
88 (*Fagus sylvatica*) dominated broadleaved forests in Southern Scandinavia. We selected three
89 regions, two in Denmark (Funen and Zealand) and one in Sweden (Scania), where conservation
90 measures would provide outcomes of comparable quality. We take advantage of the fact that
91 the distance between Zealand and conservation sites in Funen is similar to the distance
92 between Zealand and conservation sites in southern Sweden. Both Funen and southern Sweden
93 are separated from Zealand by bridged waters and roughly similar distances.

94

95 **1.1 Literature review**

96 As a background for our research questions, we reviewed the relevant literature, focusing on
97 studies addressing the linkage between stated preferences for environmental goods, spatial
98 dimensions and nationality. Distance decay models have been applied in a number of stated
99 preference studies to estimate spatial heterogeneity. Sutherland and Walsh (1985) was one of
100 the early studies to show that respondents living further from policy areas have lower
101 estimated marginal WTP. Bateman et al. (2006) provided a theoretical justification for distance
102 decay analysis from a use value perspective (recreational demand), where greater travel
103 distances to a natural resource site implies lower net values, ceteris paribus, due to greater
104 costs of reaching the site. Many studies have applied the basic form of the distance decay model
105 to assess spatial welfare heterogeneity (Abildtrup et al., 2013; Adamowicz et al., 1997; Bateman
106 et al., 2002, 2006; Brouwer et al., 2010; Jørgensen et al., 2013; Loomis, 2000; Meyerhoff, 2013;
107 Morrison and Bennett, 2004; Nielsen et al., 2016; Pate and Loomis, 1997; Rolfe and Windle,
108 2012; Yao et al., 2014). They do so by applying the postal code of a respondent's mailing
109 address (home or origin point) and a geocoded single point that represents the affected area
110 (the destination point).

111

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

118 WTP estimates for non-market outcomes. Finally, it has been argued that theoretical distance
119 decay justifications may not apply for non-use value (Bateman et al., 2006; Hanley et al., 2003).
120
121 While all these studies have addressed the effect of concepts of distance on welfare measures
122 of environmental changes, they did not investigate if distance effects can be separated from
123 nationality effects with respect to the site of provision. This has particular policy relevance
124 when analysing the value of habitat and biodiversity conservation as a public good in an
125 international context.
126
127 The effect of nationality of respondents relative to the country of provision for the
128 environmental good has been addressed in various ways. For example, respondents' nationality
129 was found to be a significant element of WTP for users of the whale-watching experience in an
130 Australian marine park (Davis and Tisdell, 1999). Similarly, Samdin et al. (2010) compared
131 Malaysians and international visitors' preferences and found that the respondents' nationality
132 affected significantly their preferences for protection of the Taman Negara National Park. In a
133 study focused on valuing marine species Ressurreição et al. (2012) found respondent
134 nationality and the degree of attachment to the study site as the main driver of WTP. A study
135 by Carlsson et al. (2012) also showed the effect of respondents' nationality on WTP for a climate
136 change mitigation programme. A somewhat different take is that of Yao et al. (2014), who found
137 a significantly higher WTP for conservation of national symbolic species (Brown Kiwi in New
138 Zealand). Dallimer et al. (2014) showed that people in three different countries (Denmark,
139 Estonia and Poland) were willing to pay significantly more for locally delivered services than
140 for similar types of goods delivered in the two other countries, but did not account for
141 differences in distance between the sites of provision and the respondents' locations. Possible

142 explanations for such effects include sense of ownership or identity (Bateman et al., 2002;
143 Hanley et al., 2003; Dallimer et al., 2014; Dallimer and Strange, 2015; van Houtum and van
144 Naerssen, 2002), ethical concerns (Daw et al., 2015) by beneficiaries, notably if respondents
145 have a belief system involving an obligation to protect biodiversity conservation in their own
146 country, or strict border crossing constraints and differences in welfare (Valasiuk et al, 2017).
147 In general, these studies addressed the nationality effects associated with the countries of
148 provision, when these are far from each other and from the respondents' country of residence
149 and/or have different culture, rules, environment etc.
150
151 The contribution of the present paper is to investigate the role of nationality on WTP for
152 biodiversity. The case is two neighbouring countries, sharing a similar environment and easy
153 access between the two countries, allowing for control of distance.
154
155 ***1.2 Hypothesis formulation***
156 Based on the above literature and taking advantage of the spatial layout of our experimental
157 case, we formulate the following null hypotheses:
158
159 *H1: Distance to the site of biodiversity conservation does not matter for people's WTP for a given*
160 *policy alternative.*
161
162 *H2: Country of biodiversity conservation provision does not matter for people's WTP for a given*
163 *policy alternative.*
164

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.

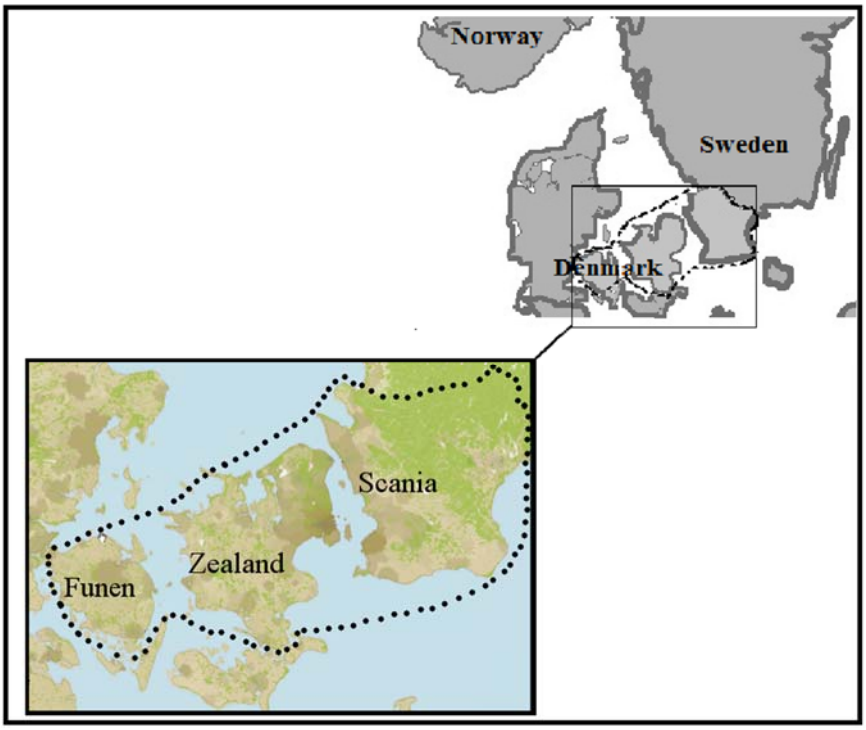

Figure 1: A map of the study area. (The green and brown colours indicate the forest cover and urban areas, respectively. The dashed line indicates the border lines of the three study regions) (source: Kempeneers et al., 2011; Paivinen et al., 2001; Schuck et al., 2002)

182

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

191

192 ***2.2 Data collection and survey design***

193 Data were collected through an internet-based questionnaire managed by the survey institute
194 'Analyse Denmark' during July-August 2012. We received 600 completed questionnaires (a
195 20% response rate) for each of the three locations (1800 in total). Survey respondents were
196 informed that the hypothetical conservation policy presented in the questionnaire would
197 improve biological diversity as well as enhance the natural dynamics of the forests. The proxies
198 used to describe these two attributes were 1) keeping old trees in the forest to age, die and turn
199 into deadwood in the forest through natural decay and 2) increasing the number of species in
200 the areas in focus by improving the living conditions for animals, plants and other organisms.
201 The selection of these attributes was based on eight different focus groups and several
202 individual interviews, involving more than 50 persons in total across all three regions. We
203 asked participants in the focus groups to think about biodiversity and evaluate various
204 representations of the way they thought about it. The result of this process was that 'Forest
205 species number' and 'Presence of natural dynamics' captured well how people perceived

206 biodiversity. Further details can be found in Bakhtiari et al. (2014). We note that while such
207 measures do not reflect the full complexity of biodiversity, the selected attributes can be
208 considered reasonable approximations of the respondents' perception of the good (cf.
209 guidelines by Johnston et al., 2017).
210
211 Respondents were informed that, across the broadleaved forests in the three regions, one
212 could find around 10,000 species in total. Based on the literature (Petersen et al., 2016, 2012)
213 and data from the global biodiversity facility (GBIF: The Global Biodiversity Information
214 Facility, 2017a) on species diversity and conservation the number of species in Denmark was
215 assumed to be approximately 35,000[6]. Of these, around 65 per cent can be found in broadleaf
216 dominated forests, which are the climax ecosystem in much of the regions area. Since the
217 broadleaf dominated forests account for approximately 41percent of the Danish forest area
218 (Johannsen et al., 2013), we assumed that 10.000 species would be a conservative estimate.
219 The number includes vascular plants and vertebrates, although a substantial part of the species
220 are insects, non-vertebrates and fungi. We assume the numbers of species in the case areas to
221 be similar, since the forest ecosystems on which we focus are quite similar in Denmark and
222 Southern Sweden. We should note that we expect the total number of species in the entire
223 Sweden to be higher, as the country reaches into the boreal zone[7]. However, on any given forest
224 area, much fewer species would in general be present and based on the above references and
225 a few as others (e.g. Lawesson et al., 1998), respondents were informed that around 1,000
226 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).

227 conservation policy. Respondents were then presented with alternative attribute levels which
228 would increase the number of abundant species to 1,500 or 2,000 in the forest conservation
229 area. The number thus comprised common, rare, and potentially endangered species including
230 both larger and smaller species groups. Thus, the approach was to look at the diversity of
231 abundant species at the conservation sites regardless of how rare or endangered the species
232 were prior to the conservation effort[8]. This differs from many other studies emphasizing
233 conservation of endangered species (for two Danish studies see Campbell et al., 2014; Jacobsen
234 et al., 2008).
235
236 An important additional attribute was the location of policy implementation, which was
237 presented on a four-level scale: The status quo of current forest management in all regions and
238 policy implementation in Funen, Zealand and Scania, respectively. Finally, we included a tax
239 attribute in the form of an increase in the annual income tax caused by the selected policy. Table
240 1 shows the attributes and attribute levels. The current management' attribute is equal to the
241 lowest level of each of the attributes (bold).
242
243 **Table 1**
244 Attributes and levels presented to respondents in choice tasks. The current situation is represented by the lower level and is
245 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-

266 demographic variables of the individuals. We note that the correlation between the bridge
267 dummies (which are 0, 1 or 2) and the distance measures is very low due to respondents being
268 scattered widely within the three regions.
269
270 We applied a fractional factorial design and optimising d-efficiency[9] for a multinomial logit
271 model (Scarpa and Rose, 2008), with zero priors and assuming preferences for all attribute
272 levels except price to be randomly distributed. The design included 24 alternatives, which were
273 divided into four blocks. Some of the choice sets included dominated alternatives implying that
274 respondents would not do any real trade-off and these were changed manually. The
275 questionnaires were translated into Danish and Swedish. They were tested through focus
276 groups and a pilot study, and benefited from participants' feedback regarding the wording of
277 the questions.
278
279 **2.3 Econometric specifications**
280 We tested our hypotheses by estimating a utility function for the conservation improvement as
281 perceived by the respondents in our pooled dataset, and each of the study locations. The utility
282 function for our pooled dataset was described as:
283
284 $U_{ij} = (\mathrm{ASC_j} + \beta_{1i} Foreign_{ij} + \beta_{2i} Distance_{ij} + \beta_{3i} Bridge_{ij} + \beta_{4i} Biodiversity1500_j + \beta_{5i} Biodiversity2000_j + \beta_{6i}$
285 $Leaving7trees/ha_j + \beta_{7i} \mathrm{Leaving15trees/ha}_j + \beta_{8i} Tax_j + \eta_j + \varepsilon_{ij}$ [1]
286
287 Where *i* =individual and *j* =alternative. The deterministic part of the utility is captured by the
288 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 was* 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:

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

313 , where the locations *(Funen)*$_j$ and *(Scania)*$_j$ addressed the utility of a resident in Zealand for
314 implementing forest protection policy *j* in Funen or Scania, respectively, as opposed to
315 implementation in Zealand. In equation 2, H1 was rejected if $\beta_{Zealand,2i}$ and $\beta_{Zealand,3i}$ was
316 significantly different from zero. H2 was rejected if $\beta_{Zealand,3i}$ was significantly different from
317 $\beta_{Zealand,2i}$. Similarly, when respondents were from Scania, H1 was rejected if the parameters for
318 Zealand and Funen were significantly different from zero, and larger for Zealand than for Funen.
319 Finally, H2 could not be tested for Scania and Funen as distance and country of provision could
320 not be separated.
321
322 The preference models were estimated using a random parameter error component logit model
323 (RPL) (Ben-Akiva et al., 2001; Brownstone and Train, 1998; Revelt and Train, 1998; Scarpa et
324 al., 2005). In this model the utility of a good was described as a function of its attributes, and
325 people chose among composite goods by evaluating their attributes. According to Train (2003),
326 the mixed logit probabilities could be described as integrals of the standard conditional logit
327 function evaluated at different β's, with a density function as the mixing distribution. Thus,
328 while the utility coefficients varied from one individual to another , they were constant over the
329 *N* choice occasions for each individual, and we accounted for this panel structure. The
330 probability density was specified to be normal and the unconditional probability of choosing a
331 sequence of alternatives k was defined as:

332
$$\Pr(ik) = \int \left(\prod_{n=1}^{N} \left[\frac{\exp^{\lambda_{ikn}\beta_i' x_{ik_n n}}}{\sum_{j}^{J} \exp^{\lambda_{ikn}\beta_j' x_{ijn}}} \right] \right) \phi(\beta \mid b, W) d\beta \qquad [3]$$

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

382 random parameters, we used a normal distribution. We find these assumptions to provide the
383 best model fit. The estimated parameters and derived WTP based on the pooled data set are
384 shown in Table 5. The standard errors of the WTP were estimated using the Delta method (Hole,
385 2007). The environmental attributes and the error components were significant and with
386 expected positive signs. The tax coefficient was negative as expected. The alternative specific
387 constant (ASC) was positive and significant. The attributes '*Distance*' and '*Foreign*' were both
388 significant and negative, the implication being that *H1* and *H2* were both rejected.
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402 **Table 5.**
403 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

435 Parameter estimates in two location samples using RPL+EC. Parameters in **bold** format relate to hypothesis 1 and
436 2, respectively. The location attribute levels are estimated relative to the omitted location level identical to the
437 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	

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

439 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
440 implementation in respondents own location.

441

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

466 authorities[10], which are (is?) in the range of 2-4 DKK per km. In addition, the WTP for 'Bridge'
467 toll corresponds reasonably well with the real cost of a return ticket, which drivers pay to cross
468 the bridge. Thus, the travel cost-related parameters are consistent with the cost of visiting the
469 forests in the other regions typically once per year, a frequency which is well in accordance
470 with the observed frequencies in the samples of respondents (cf. Table 2). It is worth noting
471 that these variables are likely linked to the respondents' expected direct use values of
472 biodiversity conservation in the different policy sites.
473
474 For the Zealand subsample we also tested whether respondents preferred Funen or Scania for
475 forest protection implementation. Based on the marginal effects of location attributes, we again
476 concluded that both distance and nationality of country of provision matter. The fact that the
477 country of provision has a separate effect, once distance effects have been corrected for,
478 suggests that also non-use values derived from biodiversity conservation may be sensitive to
479 the country of provision. For the second subsample, respondents in Scania assessed forest
480 protection policy in Funen and Zealand (these localities only differed in terms of travel distance
481 to the policy location). We found that respondents in Scania had a larger marginal WTP for
482 implementing a forest protection policy in Zealand, compared with Funen. It is worth noting
483 that the majority of the respondents from Scania stated that they never had visited a forest in
484 Funen or Zealand (87% and 75%, respectively).
485
486 With respect to the main attributes of the experiment, the WTP measures and patterns are as
487 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

488 to related studies that specifically target threatened species (e.g. Jacobsen et al., 2008; Campbell
489 et al., 2014) we find the WTP per species to be lower and with a weak scope sensitivity. Jacobsen
490 et al. (2012) evaluated Danes' WTP for enhancing populations of common or rare species
491 relative to their WTP for saving threatened species. They showed that people have a lower WTP,
492 potentially even decreasing with scope, for enhancing the populations of common species,
493 compared to saving threatened species. Our case study here addresses policies that make more
494 species abundant, but not specifically and only threatened species. Thus our results seem well
495 in line with comparable results in the literature, namely that WTP per species is somewhat
496 lower because we are valuing common and not endangered species.
497
498 ***4.1 Possible reasons and consequences***
499 Other studies have found that nationality and degree of attachment to locations affect
500 preferences (e.g., Brouwer et al., 2010; Carlsson et al., 2012; Dallimer et al., 2014; Davis and
501 Tisdell, 1998; Hanley et al., 2003; Ressurreição et al., 2012; Samdin et al., 2010; Yau et al., 2014;
502 Valasiuk et al., 2017). Conversly Dumalisile et al. (2005), Horton et al. (2003), and Jin et al.
503 (2010) did not find any significant effect associated with the degree of attachment to the
504 location of environmental improvements. However, none of these studies was designed to
505 separate the effect of distance to a site of provision from the effect of the site being in another
506 country. Our results show that, in the current case, biodiversity conservation benefits are not
507 independent of geographical and national jurisdiction (Dallimer and Strange, 2015; Valasiuk et
508 al., 2017). We find that even respondents from Scania who state they have never visited forests
509 in Denmark, have a WTP for biodiversity conservation that decreases with distance to locations
510 in Denmark. The specific effect of the location of provision – when controlling for distance and
511 other travel cost variables – suggests that non-use values may be sensitive to the geographical

512 location of provision. This result adds to findings by Brock and Xepapadeas (2003) and Hanley
513 et al. (2003), who found biodiversity conservation to benefit people at different spatial and
514 temporal scales. A large number of respondents in our study (Table 4) replied that they never
515 or very rarely had visited forest areas in any of the two other locations. This suggests that
516 recreational benefits (direct use values) are not the main reason for the WTP differences across
517 different locations that we find.
518
519 Thus, our core result is that values related to biodiversity conservation may be sensitive to
520 country of provision beyond what can be explained by distance as a cost driving element of use.
521 We argue that obtaining separate estimates of use and non-use values from conservation
522 actions may in fact not be valid. An argument that we, and other authors (e.g., Tacconi, 2000)
523 have made is that non-use values may not be independent of place of provision. If they are not,
524 they are also much harder to separate from use values. Simply including interactions with e.g.
525 recreational habits, distance from home to policy site, etc., will not allow for such a separation
526 if people derive higher non-use based utility from knowing that a pristine natural area is closer
527 to them – even if they never intend to use it.
528
529 Our findings may have important policy implications for biodiversity conservation across
530 borders and the funding of these. Notably, our findings suggest that it may be more difficult to
531 gather political support for cross-country biodiversity conservation actions even if such
532 coordination could be more cost-efficient. We may speculate as to the reasons why the country
533 of provision matters beyond the distance to site of provision itself. People may find it more
534 acceptable to invest more in conservation in their home country for a number of reasons as
535 discussed by Bateman et al. (2002), Dallimer et al. (2014), and Dallimer and Strange (2015),

536 Hanley et al. (2003), and van Houtum and van Naerssen (2002), which suggest that ownership
537 or spatial identity may be important for some environmental assets, even for non-use value.
538 Indeed the finding here may carry over to other international environmental investment issues
539 like, for example, climate change management measures. specific studies could address this.
540
541 Thus, our results add further to the findings and discussions of Perrings and Halkos (2012),
542 who suggested that the optimal level of biodiversity conservation might be expected to vary
543 depending on the spatial scale at which the problem is analysed, and depending on which
544 (national) groups are involved in conservation decisions. We do not engage in specific cost
545 benefit analyses here, but note that previous studies have shown that the opportunity cost of
546 setting aside forest for biodiversity protection (using capital budgeting approaches) is in the
547 range of 200-400€ per ha and year (Jacobsen et al., 2013; Petersen et al., 2016). Another Danish
548 study (Danish Economic Councils, 2012) found that the cost of protecting Danish forest habitats
549 is less than € 7 million annually, or less than €3 per household per year, and hence significantly
550 lower than the WTP measures estimated in the current study, as well as in similar studies
551 (Jacobsen et al., 2008).
552
553 ***4.2 Caveats and further work***
554 Differences in factors such as national income, species richness, pressures on biodiversity, and
555 conservation infrastructure are all likely to be associated with differences in national
556 conservation efforts (Dallimer et al., 2014; Perrings and Halkos, 2012). In our study, all of these
557 factors were assumed comparable at the sub-sample level. Future studies would probably
558 benefit from investigating these issues across a wider range of cases, even if this may imply
559 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

584 Dorrough and Glöckner (2016) found evidence that, in cross-societal cooperation games,
585 knowledge about the other player's nationality matters.
586
587 **5. Concluding remarks**
588 We believe that the current study of the value of biodiversity conservation successfully
589 distinguished the effect of the distance to site of provision from the effect of the country of
590 provision, with regard to preferences for conservation outcomes. This is novel to the literature.
591 We found distance-related attributes to reflect bridge tolls and per-kilometre transport costs,
592 and found Swedes and Danes to prefer provision in their own country, over provision in the
593 neighbouring country. Denmark and Sweden are neighbouring countries with similar
594 languages, history and cultures. The magnitude of the nationality effect found in our study may
595 therefore be larger, if future studies address the same issue for countries further apart, and
596 countries less similar to each other than Denmark and Sweden. For example, Dallimer et al.
597 (2014) showed that the nationality affect was a significant factor of WTP for residents of
598 Estonia, Denmark and Poland and respondents had higher preference for biodiversity
599 conservation in their own country relative to in other countries. However, their findings did not
600 separate country effects from distance effects. The overall results of this study have relevant
601 policy implications for regional and even global biodiversity conservation efforts. The
602 underlying assumption in most conservation management models is that the benefit of
603 biodiversity conservation is independent of spatial scale, and culture or nationality. Several
604 studies demonstrate the magnitude of cost-efficiency gains of internationally coordinated
605 conservation policies (Bladt et al., 2009; Dallimer and Strange, 2015; Hull et al., 1998; Kark et
606 al., 2009; Moilanen and Arponen, 2011; Rodrigues and Gaston, 2002; Strange et al., 2006). This
607 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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