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Dallimer, M, Jacobsen, JB, Lundhede, TH et al. (3 more authors) (2015) Patriotic values for public goods: transnational trade-offs for biodiversity and ecosystem services? Bioscience, 65 (1). 33 - 42. ISSN 0006-3568

https://doi.org/10.1093/biosci/biu187

This is a pre-copyedited, author-produced PDF of an article accepted for publication in BioScience following peer review.

The version of record (Dallimer M, Jacobsen JB, Lundhede TH, Takkis K, Giergiczny M, Thorsen BJ. (2015) Patriotic values for public goods: transnational trade-offs for biodiversity and ecosystem services BioScience, 65, 33-42) is available online at: http://bioscience.oxfordjournals.org/content/65/1/33.full?keytype=ref&ijkey=s0C5qhUpYDsmwLy

Patriotic values for public goods: are there transnational trade-offs for biodiversity and ecosystem services?

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Running head: Patriotic values for ecosystem services

Keywords: biodiversity conservation, choice experiment, ecosystem services, non-market valuation, semi-natural grassland, stated preference

Abstract

The natural environment is central to human well-being through its role in ecosystem service (ES) provision. Managing ES often requires coordination across international borders. Although this may deliver greater conservation gains than countries acting alone, we do not know if the public supports such an international approach. Using the same questionnaire in three countries, we quantified public preferences for ES in their home country and across international borders. In all three countries, people were willing-to-pay for ES in general. However, our results show there is a limit to the extent that environmental goods can be considered "global". ES with a use element (habitat conservation, landscape preservation) attracted a "patriotic premium", where people were willing-to-pay significantly more for locally-delivered services. Supra-national management of ES needs to be balanced against the preferences people have for services delivered in their home country.

Introduction

The natural environment is central to human well-being through its role in ecosystem service provision (Sachs et al. 2009). There is therefore considerable interest in how best to manage the natural world to enhance the delivery of a wide range of services (e.g., Kumar 2010, UKNEA 2011). However, the effective preservation and enhancement of biodiversity and ecosystem services can require intervention across varied socio-economic and political borders, not least because ecosystems, the biodiversity they contain and the services they deliver are often shared amongst such contexts. For example, long distance migratory species can be responsible for functional links across distant regions (Bauer and Hoye 2014), and thus require novel approaches to their management (e.g., Semmens et al. 2011), which can include transnational organizations. In sub-Saharan Africa, for example, highly mobile migrant pests move frequently across national borders (Dallimer et al. 2003, Cheke and Tratalos 2007). Multinational agencies (e.g, the “Desert Locust Control Organisation – East Africa”) coordinate management at a regional level to minimize the ecosystem disservices, in the form of crop yield loss, caused by such pests. Elsewhere, supra-national bodies, such as the European Union, determine policies and legislation for species and habitat management that operate across many different nations (European Commission 1979, 1992, 2000). Finally, many water catchments are transnational (Lopez-Hoffman et al. 2010) and are managed as such.

Despite the widespread existence of trans- and supra-national bodies in ecosystem and biodiversity management, we know little about the extent of public support for initiatives which operate at international scales. This is important because with limited resources available for biodiversity conservation and ecosystem management, we require an understanding people's preferences for different aspects of the natural world as one means to prioritize actions for a number of reasons: (i) people have opinions about where to invest in conservation (Jacobsen and Thorsen 2010); (ii) conservation is frequently funded by governments who may wish to respond to the values expressed by the public; and (iii) interventions are more likely to succeed if they align with public preferences. This raises questions as to the extent to which biodiversity, environmental goods and services should be delivered locally, as well as globally. Some services, such as recreation, landscape appreciation or wild species diversity, may have a greater value to nearby populations who are able to experience them and therefore benefit from their use, as well as non-use, values (Atkinson et al. 2012). Others, such as carbon sequestered and storage through vegetation restoration, although often quantified at a local scale, deliver their benefits globally (Bulte et al. 2002).

Here we quantify the values that the public place on biodiversity and ecosystem services delivered across international boundaries, as opposed to within their country of residence. We base our study in the European Union (EU), where many policies pertaining to biodiversity conservation and ecosystem service management (e.g. Birds, Habitats Directives, Common Agricultural Policy and its agri-environment elements, commitments to reduce carbon emissions) are formulated at a supra-national level. Although the available evidence suggests that this approach can be relatively effective at the continental-scale at protecting, for example, avian populations (Donald et al. 2007), there is little understanding of the extent to which the general public in Europe support allocating funds for ecosystem service management internationally as opposed to a more local approach.

Methods

A commonly used approach to assess public preferences for the natural world is to assign monetary values to changes in ecosystems and the services they supply. Although sometimes controversial amongst conservation biologists, monetary valuation facilitates making direct comparison with other costs and benefits in decision-making processes and its use has become widespread (Hanley and Barbier 2009, Kumar 2010). Here we use the stated preference non-market valuation technique of the choice experiment (CE) to ask two questions: do people value ecosystem services and biodiversity across international boundaries and, if so, how do those values vary according to the scale at which the goods themselves deliver benefits? To do this we choose a suite of services that vary in their scale of delivery from global (enhanced carbon capture for climate change mitigation) through both global and local (biodiversity conservation) to mainly local (the preservation of landscapes that are culturally and aesthetically appreciated) (see Survey Design below). We hypothesize: (i) there will be a preference for ecosystem services to be delivered locally, as opposed to across international borders; and (ii) this preference will be weaker for more global public goods.

Choice experiments draw on theories of economic value (Lancaster 1966) and the application of random utility theory to choice (McFadden 1974). The methodology is based on probabilistic choice where individuals are assumed to select a single alternative which maximizes their utility from a set of available alternatives (Supplementary Material Appendix S1). CEs involve presenting participants with a number of choice sets consisting of two or more alternatives, each described by various levels of a set of attributes and a monetary cost which would finance the changes in the attribute levels described in an alternative. This allows WTP to be calculated using estimated parameters of the choice probability function for the different alternatives. The WTP for a marginal improvement in an attribute can then be calculated as the ratio between the parameter of that attribute and the parameter of the price attribute (See Supplementary Material for analytical details). Choice experiments are commonly used to value changes in ecosystem services and biodiversity (Christie et al. 2006, Jacobsen and Thorsen 2010, Morse-Jones et al. 2012, Dallimer et al. 2014) and offer a wide range of information on trade-offs among the benefits provided by the different alternatives (Adamowicz et al. 1998, Adamowicz et al. 1997).

Survey Design

The focus of the CE was to value changes in ecosystem services across international borders. We used semi-natural grasslands in northern Europe, a study system for which such an analysis is particularly pertinent not least because environmental policy delivered across member states of the European Union has a long-standing international component (e.g., the Birds and Habitats Directives, and the Natura 2000 network of protected areas; (European Commission 1979, 1992, 2000). Semi-natural grasslands have historically been subject to huge losses in extent and quality (Veen et al. 2009), and they are important for cultural and aesthetic reasons (e.g, Sand-Jensen 2007), as well as being a key habitat for biodiversity conservation in Europe. This was acknowledged by Mariann Fischer Boel, the EU Commissioner for Agriculture and Rural Development in 2009, "…grasslands […] represent a key element in Europe's rich diversity of landscapes and the public appreciate the beauty of Europe's meadows" (Veen et al. 2009). Indeed, many grassland systems are included in the continent's register of "High Nature Value Farmland" that recognizes the central place that traditional farming techniques play in maintaining culturally important and biodiverse landscapes (e.g., Knowles 2011). Despite this, and even though they deliver a wide range of ecosystem services (European Commission 2008), grasslands are rarely the subject of non-market valuation exercises.

We selected attributes for the CE based on services which are delivered by semi-natural grasslands, have an international dimension to their management and are likely to span different scales of beneficiaries. Three such services are: the preservation of landscapes that are culturally and aesthetically appreciated, biodiversity conservation and enhanced carbon capture for climate change mitigation.

The EU promotes the preservation of landscapes through the European Landscape Convention (Council of Europe 2000). Regions with a high coverage of semi-natural grasslands often retain features associated with culturally important and aesthetically attractive landscapes, such as traditional buildings, boundaries and field sizes (Sand-Jensen 2007; Veen et al. 2009; Knowles 2011). Traditional landscapes tend to have strong cultural links to the region in which they are found (Jacobsen and Thorsen 2010) and their enjoyment is thus largely a use value. We would expect beneficiaries to be mainly restricted to the country in which a particular region is located.

The conservation of biodiversity and habitats within the EU is governed via instruments such as the Habitats Directive (European Commission 1992) which all member states are expected to implement. Biodiversity is considered central to supporting all ecosystem services (Balvanera et al. 2006). However, there is ongoing debate as to whether biodiversity per se can be considered a service in and of itself (Mace et al. 2012), though the protection of biodiversity clearly has value to people (e.g., Christie et al. 2006, Morse-Jones et al. 2012, Dallimer et al. 2014). For example, the UK National Ecosystem Assessment includes wildlife diversity both as an intermediate service and as a final provisioning and cultural service (UKNEA 2011). We include it as a final service because of its associated use and non-use values for EU citizens (e.g., Bateman et al. 2013, UKNEA 2011). The benefits of the service could therefore be experienced both locally, and potentially globally.

The EU has committed its member states to reducing carbon emissions by 20% below 1990 levels by 2020 (EEA 2010). Enhancing storage and uptake within vegetation and soils is one potential pathway through which part of these targets could be met. Semi-natural grasslands can be managed by manipulating fertilizer application, grazing levels and promoting the presence of certain forbs to increase carbon uptake and storage in some situations (De Deyn et al. 2011). The benefits delivered by this service (in terms of climate amelioration) would be experienced globally.

We elected to use an increase in areas managed for biodiversity as an attribute rather than an increase in species richness or the abundance of key species. This was to ensure our estimates of WTP would not be affected by preferences for certain taxa (e.g., Jacobsen et al. 2008). The landscape preservation attribute was also hectare-based, making it directly comparable to the biodiversity conservation attribute. However, the units for the carbon capture attribute were tC ha^{-1} yr^{-1}. Although these units are perhaps more abstract than a third hectare based attribute, the direct benefit to people from the carbon attribute is the tC captured rather than the number of hectares over which the C is distributed. We therefore use the component that carries the utility directly, even though this may restrict direct comparisons of value between the different services.

Our study system was centered on northern Europe. Within this region we selected regions which were comparable in terms of topography, area, habitat type and the number and extent of designations under the EU Habitats Directive (Supplementary Material Appendix S1). We also wished to cover a range of international cultural differences found in this region and therefore included a western European nation (Denmark), a former communist country (Poland) and a former constituent part of the Soviet Union (Estonia) (Fig. 1). By choosing sites that were similar, we attempted to ensure the CE quantified trans-national effects on the values people ascribe to the sites, rather than, for example, habitat preferences, marginal effects related to how large our example regions were, pre-formed preferences for certain locations or species (Bateman 2009, Jacobsen and Thorsen 2010, Jacobsen et al. 2008).

To estimate measures of economic benefit from changes in the environmental attributes listed above, a cost attribute was included in the design specified as an increase to the householder’s annual taxation bill needed to finance the management measures. Choices would then show how much people are willing to trade-off improvements in an environmental attribute for a decrease in their income. The levels of the cost attribute were determined based on previous studies (Bartczak et al. 2008, Jacobsen and Thorsen 2010), and were adjusted following focus groups and pilot tests. Each nationality was presented with costs in their local currency, with amounts purchasing power parity calibrated to be equivalent.

An optimal design for the CE was generated and we included Bayesian priors from a pilot exercise to improve design efficiency (Ferrini and Scarpa 2007, Scarpa and Rose 2009). This resulted in a CE consisting of 12 choice cards, divided into two blocks. Each respondent therefore faced six choice sets which asked them to choose between four alternatives (for an example see Supplementary Material Appendix S2). These were three “policy-on” options which included different combinations of the attributes (carbon capture, habitat conservation, landscape preservation, region, and the annual tax cost) and a no cost status quo alternative in which no changes would take place across all regions. The “policy-on” options included the baseline of no change and two levels of change in carbon capture, habitat conservation and landscape preservation, and six levels of cost (Table 1).

The questionnaire was initially developed in English and translated by native speakers into the relevant local languages. We used focus groups and a pilot exercise to help finalize the questionnaire in two different ways. Firstly, feedback from participants ensured that translations were understandable to the general population and used appropriate wordings that were relevant to national situations. Final versions of the questionnaire were therefore produced only in Danish, Polish, Estonian and Russian (to account for the Russian speaking population in Estonia) and are available from the authors. Secondly, the focus groups and pilot exercises allowed us to test the structure and meaning of the CE and its associated attitudinal and socio-demographic questions.

Commercial polling companies were used to deliver the survey to an online panel of respondents in winter 2012. Around 3200 individuals were invited to take part in the survey in each country. Data collection was finalized when at least 850 respondents (representative of national population according to age, gender, education, employment) had completed the questionnaire. Initially we were supplied with over 1200 responses from Poland, but we wished to have an equivalent number of respondents in each country, a random sample was selected from these to bring the sample size in line with those in Denmark and Estonia. Of the completed responses, we removed 22 (0.8%) that were completed in less than five minutes (insufficient time to read through the survey) and 25 (1.0%) for which answers to the full set of choice cards were not recorded. The status quo option was chosen for all choice cards by 138 (5.4%) respondents who also gave a motivation for this pattern of answers which was consistent with protesting against the questionnaire itself or the payment vehicle used (Supplementary Material Appendix S3). Although the proportion of protesters was small, standard practice assumes they did not reveal their true preferences and should be excluded from further analyses (Jacobsen and Thorsen 2010, Meyerhoff and Liebe 2008). Remaining data from all countries were merged and analyzed together, resulting in a final sample size of 2367 (approx. 800 respondents/country answering 14202 choice cards). Analyses were conducted in NLOGIT using a mixed logit specification with an error component model (Greene and Hensher 2007, Scarpa Riccardo et al. 2005). Parameter estimates from the simpler conditional logit model were of the same sign and magnitude as the mixed logit, so we report only the results from the more complex model. We included a correction for scale difference (Hensher et al. 1999) between nationalities. Details of the analytical approach and theoretical background are given in Supplementary Material Appendix S1.

Results

Respondents of all nationalities expressed a positive and significant WTP for enhanced ecosystem services (Table 2). Irrespective of where services were to be delivered, people stated a WTP (±S.E.) for habitat conservation of €0.038±0.004 and for landscape preservation of €0.028±0.004 per household per year for the management of one additional ha. WTP for carbon capture was €0.019±0.002 per household per year for an extra tC ha^{-1}.

There were significant preferences for where management actions should take place. Respondents from the complete sample expressed the highest utility for actions in Denmark (as contained within the ASC) of €0.078±0.010. The variable ASC is the 'Alternative Specific Constant', which measures the WTP for taking any form of action. Given that country variables are 0/1 dummies, in order for us to carry out the estimation and not over-specify our models we did not include one country, in this case Denmark. WTP amounts for Denmark are thus confounded with the ASC. WTP measures for Poland and Estonia are relative to the ASC. Thus, across all respondents, the WTP for management actions in Poland was €0.007±0.004 lower than Denmark, and ecosystem services delivered in Estonia were on average significantly less valued across respondents from the three countries, being €0.022±0.006 lower than in Denmark. The overall utility for actions in Estonia was still positive and significantly different from zero. This pattern reflects that all respondents were more likely to choose alternatives based in their own country, and that Polish and Danish respondents chose alternatives in Denmark and Poland respectively more often than they chose provision in Estonia. Similarly, Estonians were largely indifferent in their choices between Denmark and Poland (Table 3).

Although prices were purchasing power parity corrected, we would still expect there to be significant differences between nationalities with respect to marginal utility of income. We accounted for this by including two "nationality x price" interaction variables in the models. As previously noted, because the country variables are 0/1 dummies, we can only include two of them in the model. Thus the parameter estimate for "Price" refers to Danish respondents, and the interaction terms for Poland and Estonia quantify the additional contribution to that price parameter (e.g., for Poland -1.361 – 0.309). The marginal utility of income was therefore significantly higher for Polish and Estonian respondents compared to Danes (Table

4; “Estonian x Price” and “Polish x Price” interactions). As WTP is calculated by dividing the parameter estimate for the environmental attributes by that of price, the precise WTP estimates vary by a fixed ratio between nationalities. For simplicity in the text, we report WTP based on Danish price sensitivity (Table 4; “WTP in € for Danish respondents”).

We wished to separate out the effects of nationality and region to examine the more general issue of how much extra people were willing to pay to have a service delivered in their own country, rather than the exact same service provided elsewhere. We did this by including a variable “own country” (which took the value one when management actions took place in the respondent’s country of residence, and zero otherwise), which was interacted with the environmental attributes. In addition we included interactions between this variable and the region of provision, which were intended to capture latent and unobserved effects of the respondent’s nationality on their preferences. The general pattern remained (Table 4; Fig. 2), with WTP for habitat conservation, landscape preservation and carbon capture €0.034±0.007, €0.018±0.006 and €0.011±0.003 respectively.

The own country region preferences were all significantly different from zero and positive (Table 4), indicating that respondents were willing to pay more for any actions to take place in the country they resided in (Fig. 2). This was especially marked for Estonians, who were willing to pay an additional €0.114±0.015 for actions in Estonia. In contrast, Danes expressed the lowest additional valuation for actions to take place in their own country of €0.033±0.013.

Across all three countries, WTP for habitat conservation and landscape preservation within respondents’ own country more than doubled the WTP estimate for the same actions undertaken elsewhere. For example, the WTP for habitat conservation was €0.034±0.007, while the additional WTP for habitat conservation in a respondents' home country (as captured by the own country x habitat conservation interaction) was €0.047±0.011, giving a total WTP for habitat conservation of €0.081. The own country "patriotic premium" was relatively largest for landscape preservation. The premium for carbon capture delivery in a respondent's own country was smallest, though still of significant size (Table 4; Fig. 2).

Thus far our results support our two main hypotheses, namely (i) there should be a preference for ecosystem services to be delivered locally, as opposed to across international borders; and (ii) this preference should be weaker for more global public goods. However, there are other potential explanations for the patterns so far described. For instance, the preference for services delivered in a respondents' country of residence could be driven by regular outdoor recreationalists being willing to pay higher amounts for locally delivered services for which they gain use value. We accounted for this by including a variable for frequent (more than one visit per month) recreational visitors to the countryside. Finally, although we used purchasing power parity to match tax amounts presented to respondents from different countries, we would expect respondents on relatively high incomes to exhibit a different sensitivity to price compared to those on low incomes. We controlled for this by including an interaction between price and high income respondents (those whose household incomes were in the upper income brackets for their country of residence; Supplementary Material Appendix S4).

Respondents reporting household incomes in the higher brackets for their country and regular recreational users were less sensitive to price (Table 5; "high income x price" interaction €0.171±0.051, and "user x price" interaction €0.181±0.051). Although regular users had a generally higher WTP (the “user x price” term), they were not willing to pay a greater amount for any specific environmental attributes (parameter estimates for “user x habitat conservation”, “user x landscape preservation”, “user x carbon capture” interactions all not significantly different from zero). There was no impact on the magnitude or relative ranking of the preferences for services to be delivered in the respondents own country (Table 5).

Discussion

Across three European countries, we found a significant WTP for enhancements to ecosystem services provided by semi-natural grasslands, regardless of the location of delivery (Table 1). Nevertheless, people were WTP significantly greater amounts for services located in their country of residence (Tables 3, 4). The magnitude of this extra payment was linked to the extent to which the good could be considered local or global. The additional WTP for services with characteristics of a local public good (in our study, habitat conservation and landscape preservation) to be delivered within the respondents’ country of residence was much higher than that for the global public good of carbon capture.

Given that local goods are assumed to have a high use value, perhaps surprisingly we did not find that regular recreational users of the countryside were willing-to-pay more for locally delivered services (although they did have a higher WTP across all services and locations in general). Non-use values can be experienced by people without engaging in specific activities and behaviors. We may, for example, all derive utility from knowing that endangered species are protected even though we may never see them (e.g., Morse-Jones 2012). Such values require no measurable action for us to experience, and are likely to be global in nature, as they are non-rival and no-one can be excluded from receiving benefits. In contrast, use values are accrued through active use, including activities such as wildlife watching and enjoying aesthetically pleasing landscapes. As use values imply a cost for the user, in terms of money, transport and time, people are likely to care about where and how they can be enjoyed. Thus, the values of environmental public goods with large use components are likely to be less global in nature.

By simultaneously considering both respondents from, and ecosystem service delivery within, several countries we demonstrated a strong preference for local delivery, and the value that people can attach to services provided outside their home country. Cultural heritage, shared values and experiences can affect values for public goods (Ready and Navrud 2006; Jacobsen and Thorsen 2010). Here, respondents in Denmark, Poland and Estonia were willing to pay significantly different amounts for management to enhance ecosystem services, suggesting that nationality and international borders were important determinants of value. Nevertheless, political boundaries are not the same as market boundaries when assessing WTP for environmental enhancements (Loomis and White 1996). For example, residents in developed countries are willing to pay for the conservation of species in the developing world (Morse-Jones et al. 2012) and the optimal coverage by rainforest in Costa Rica is markedly higher when global (as well as local) beneficiaries are included in calculations (Bulte et al. 2002). Similarly, nationality is not always a strong determinant of value (Jin et al. 2010).

Since their popularization (MEA 2005), ecosystem services have gained considerable traction amongst researchers and policy makers keen to incorporate values for the natural world in decision making processes (Bateman et al. 2013, UKNEA 2011). Although biodiversity has a role in both underpinning many services (Atkinson et al. 2012, Mace et al. 2012), there is a danger that biodiversity conservation per se will be overlooked in the face of more obviously beneficial and quantifiable services, such as climate mitigation. However, biodiversity plays an important role in delivering cultural services (Mace et al. 2012), and is highly valued by the general public (Christie et al. 2006, Morse-Jones et al. 2012, Dallimer et al. 2014). Across the three countries in our study, when faced with a choice between management for biodiversity conservation and two other services, respondents consistently placed higher values on biodiversity, indicating that it should retain a prominent role in environmental management and policy.

We acknowledge competing explanations for the pattern documented here, not least because many other variables may be entirely confounded with region and nationality and could therefore weaken the patterns we have quantified. For example, it is possible that the size of the chosen regions was an important factor in respondents' WTP for management actions focused on particular locations. We addressed this by ensuring that the study regions were closely matched in terms of their existing areas of semi-natural grassland. However, there remained a substantial difference in the number of species considered to be under threat of extinction between the study sites (47, 54 and 22 for Estonia, Poland and Denmark respectively; Supplementary Material). The fact that Danes expressed the lowest additional WTP for habitat conservation actions to take place in their own country could plausibly be driven by the perception that actions in Denmark would contribute least to biodiversity protection across the three countries. Similarly, although respondents were not presented with the information, the relative rarity of the habitat and landscapes in each country may have played a role. For example, if a habitat is thought to be rare in a certain country, then the marginal benefits of increasing coverage may be greater than in a country where the habitat is perceived to be common. In our study this would translate to respondents demonstrating a preference for investment in habitat conservation in Denmark where semi-natural grasslands are relatively scarce compared to either Poland or Estonia. A further plausible hypothesis might be that people factor into their preference the relative costs across our three study countries. In this case, Denmark, where prices and incomes are highest, would be perceived to be the most costly country in which to undertake management actions, and thus respondents may feel their WTP would need to be greater to deliver the same environmental changes. In both cases within our CE, this would result in higher WTP estimates for actions carried out in Denmark, or via a reduced preference for “own” country among Estonian and Polish respondents. Although we do not see the latter, the WTP estimate for any action to take place in Denmark (as captured by the ASC) was higher than those for Poland or Estonia (Table 1).

Finally, preferences for public goods delivered across international borders may be influenced by the varying levels of trust that exist both within and between people and institutions of different nationalities (e.g., Zak and Knack 2001). For example, Estonians may believe their own country, with its associated laws, compliance and governance structures, is more likely to deliver enhanced ecosystem services than either Denmark or Poland (and vice versa). Alternatively, they may feel more in control of implementation if management is carried out locally (Hanley et al. 2003).

Conclusions

Current prioritization of conservation efforts tends to incorporate biophysical variables together with information regarding the distribution of socio-economic costs of land management (Ando et al. 1998, Bode et al. 2008). Large scale, often supra-national, prioritization may well be the most efficient way to deliver maximum conservation gain (Bladt et al. 2009, Kark et al. 2009). However, this takes no account of how benefits from conservation management that accrue to the human population are distributed.

A supra-national approach to ecosystem management has some support among the general population. However, the values that people express for ecosystem goods and services delivered internationally need to be balanced against the substantially higher WTP for services that are enhanced in their country of residence. Such a finding has important implications for how environmental management and biodiversity conservation are prioritized. The distinct preferences for locally delivered ecosystem services could imply a lower acceptance of international cooperation on environmental issues, coupled with a greater demand for investments in environmental programs in one's own country. In particular, goods with an obvious use value (e.g. biodiversity, aesthetically pleasing landscapes) cannot be considered as truly global public goods. In our study system, as in many others, this raises issues of trust between countries as the potential for free-riding is high. Ecosystem management could proceed in Poland, financed solely by Polish taxes, but people in nearby countries would also benefit. In many other cases, services are shared across international boundaries (e.g., carbon sequestration, catchment level water quality, and migratory species) and cooperative management would be required to maximize their value to residents of all countries.

Acknowledgements

We thank all the participants in our surveys. Additional expert opinion on semi-natural grasslands was provided by Zygmunt Kącki, Aveliina Helm and Hans Henrik Bruun. Anna Filyushkina provided the Russian translation of the questionnaire. M.D. was supported by an EU-FP7 Marie Curie Fellowship (no. 273547). M.D., J.B.J., T.H.L. and B.J.T. thank the Danish National Research Foundation for support through the Center for Macroecology, Evolution and Climate.

Supplementary Material

Extended methodological and site details (Appendix S1), an example choice card in English (Appendix S2), responses that were considered protest votes within the choice experiment (Appendix S3) and income bands allocated to the high income variable for use in the analyses of choice experiment data (Appendix S4) are available online.

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Tables

Table 1. Attributes and levels presented in the choice experiment to determine willingness to pay for ecosystem services delivered across international borders in the European Union.

Attribute	Levels	Status quo
Carbon Capture	2 or 3 tonnes carbon captured per ha per year	1 tonne carbon captured per ha per year
Habitat conservation	An extra 500 or 1000 ha of semi-natural grassland managed for wildlife and habitat conservation	No change
Landscape preservation	An extra 500 or 1000 ha of traditional landscape preserved	No change
Region	Changes only take place in Denmark, Poland or Estonia	No change in any region
Price	0, 100, 200, 400, 800, 1200 Dkr. (Denmark) 0, 25, 50, 100, 225, 350 zł (Poland) 0, 5, 10, 25, 55, 85 € (Estonia)	0 Dkr/zł/€

Table 2. Parameter and willingness to pay estimates for a random parameter error component logit model for the main effects model, based on 14202 observations from 2367 respondents (χ^2 = 9102.99, Pseudo R^2 = 0.231, Log-likelihood = -15137.15). Simulations are based on 1000 Halton draws. The ASC is confounded with the benchmark region of Denmark, and the estimates for Estonia and Poland are additional to it. WTP is reported in € per household per year for management interventions to take place over 1 ha. For carbon WTP is per tC captured on that hectare. ***, ** and * indicate significance at the 0.01, 0.05 and 0.1 levels respectively.

Variable	Parameter (SE)	Standard deviation (SE)	WTP in € (SE)
ASC	0.869 (0.118)***		0.078 (0.011)***
Estonia	-0.246 (0.062)***	1.783 (0.063)***	-0.022 (0.006)***
Poland	-0.082 (0.047)	1.393 (0.054)***	-0.007 (0.004)*
Habitat conservation	0.427 (0.049)***	0.855 (0.073)***	0.038 (0.004)***
Landscape preservation	0.313 (0.045)***	0.497 (0.089)***	0.028 (0.004)***
Carbon capture	0.210 (0.022)***	0.182 (0.053)***	0.019 (0.002)***
Price	-1.507 (0.029)***		

Table 3. Frequency with which alternatives involving the named regions were selected by respondents of each nationality. Status quo indicates that the no change option was selected.

		Region		
Nationality	Status quo	Denmark	Estonia	Poland
All respondents	0.31	0.22	0.19	0.28
Danish	0.36	0.39	0.10	0.15
Estonian	0.41	0.12	0.36	0.11
Polish	0.21	0.18	0.14	0.47

Table 4. Parameter and willingness to pay estimates for a random parameter error component logit model for the own country model, based on 14202 observations from 2367 respondents (χ^2 = 11066.51, Pseudo R^2 = 0.281, Log-likelihood = -14154.9). WTP estimates are presented for each nationality, calculated from the appropriate price parameter. ***, ** and * indicate significance at the 0.01, 0.05 and 0.1 levels respectively. The WTP for each attribute and country is calculated using the preference parameter for the attribute divided by the country's marginal utility of income, e.g. -1.361 for Denmark and -1.670 (-1.361- 0.309) for Poland. WTPs are given in € using the conversion rate of 7.4 Dkr/€.

Variable	Parameter (SE)	Standard deviations (SE)	WTP in € (SE) for Danish respondents	WTP in € (SE) for Estonian respondents	WTP in € (SE) for Polish respondents
ASC[1]	0.588 (0.128)***		0.058 (0.012)***	0.054 (0.011)***	0.048 (0.010)***
Estonia[2]	-0.370 (0.073)***	1.024 (0.061)***	-0.037 (0.007)***	-0.034 (0.007)***	-0.030 (0.006)***
Poland[2]	-0.122 (0.074)	0.842 (0.051)***	-0.012 (0.007)***	-0.011 (0.007)***	-0.010 (0.006)***
Habitat conservation	0.342 (0.071)***	0.734 (0.071)***	0.034 (0.007)***	0.031 (0.007)***	0.028 (0.006)***
Landscape preservation	0.183 (0.065)***	0.590 (0.074)***	0.018 (0.006)***	0.017 (0.006)***	0.015 (0.005)***
Carbon capture	0.111 (0.031)***	0.126 (0.061)*	0.011 (0.003)***	0.010 (0.003)***	0.009 (0.003)***
Own country x Habitat conservation	0.478 (0.111)***		0.047 (0.011)***	0.044 (0.010)***	0.039 (0.009)***
Own country x Landscape preservation	0.412 (0.102)***		0.041 (0.010)***	0.038 (0.009)***	0.033 (0.009)***
Own country x Carbon capture	0.161 (0.047)***		0.016 (0.005)***	0.015 (0.004)***	0.013 (0.004)***
Own country x Denmark	0.332 (0.134)**		0.033 (0.013)**	-	-
Own country x Estonia	1.245 (0.162)***		-	0.114 (0.015)***	-
Own country x Poland	0.970 (0.105)***		-	-	0.079 (0.009)***
Estonian x Price[3]	-0.115 (0.063)		-		
Polish x Price[3]	-0.309(0.077)***		-		
Price	-1.361 (0.047)***		-		

Table 5. Parameter and willingness to pay estimates for a random parameter error component logit model for the frequent user model, based on 12498 observations from 2083 respondents (χ^2 = 9743.25, Pseudo R^2 = 0.281, Log-likelihood = -12454.3). We do not present WTP as price parameters differ significantly across many different sub-groups. ***, ** and * indicate significance at the 0.01, 0.05 and 0.1 levels respectively.

Variable	Parameter (SE)	Standard deviations (SE)
ASC[1]	0.637 (0.135)***	
Estonia[2]	-0.384 (0.078)***	1.028 (0.065)***
Poland[2]	0.126 (0.081)	0.081 (0.054)***
Habitat conservation	0.331 (0.085)***	0.712 (0.077)***
Landscape preservation	0.096 (0.081)	0.575 (0.081)***
Carbon capture	0.146 (0.039)***	0.141 (0.063)*
Own country x Habitat conservation	0.518 (0.120)***	
Own country x Landscape preservation	0.415 (0.111)***	
Own country x Carbon capture	0.145 (0.051)***	
Own country x Denmark	0.376 (0.145)***	
Own country x Estonia	1.293 (0.175)***	
Own country x Poland	0.978 (0.113)***	
Estonian x Price[3]	-0.136 (0.067)*	
Polish x Price[3]	-0.326 (0.083)***	
User x Habitat conservation[3]	0.020 (0.086)	
User x Landscape preservation[3]	0.129 (0.080)	
User x Carbon capture[3]	-0.036 (0.040)	
User x Price[3]	0.181 (0.051)***	
High income x Price[3]	0.171 (0.051)***	
Price	-1.528 (0.065)***	

[1] ASC takes the value 1 for the alternative, and is therefore confounded with the reference level of management action in Denmark.
[2] As compared to management action in Denmark
[3] Additional to "price"

Figure Legends

Figure 1. Northern Europe showing the location of the study regions within Denmark (DK), Poland (PL) and Estonia (EE). For site descriptions as presented to respondents, see Supplementary Material Appendix S1.

Figure 2. Willingness to pay (WTP; € per household per year) for management action over 1000 ha for the own country interactions model (Table 4). The light grey bars indicate the amount participants were willing to pay for actions carried out in their country of residence in addition to the WTP estimate (in dark grey) for actions not taking place in their country of residence. Error bars are standard errors.