

University of Glasgow

Institute of Biodiversity, Animal Health & Comparative Medicine

**Discussion Papers in Environmental and One Health Economics**

2022-02

Where are transboundary pollution reductions most valued? Evidence from a bilateral choice experiment on marine plastic reductions

Keila Meginnis\*, a, c, Tobias Börger b, Nick Hanley c, Robert Johnston d, Tom Ndebeled, Ghamz E Ali Siyal d

a University of Glasgow, Institute of Health and Wellbeing

b Berlin School of Economics and Law

c University of Glasgow, Institute of Biodiversity, Animal Health and Comparative Medicine

d Clark University

\*Author for correspondence

Abstract:

Transboundary pollutions provide a unique challenge for economist and policy makers, due to the fact that domestic abatement expenditures result in benefits for multiple countries. Our ability to predict the likelihood of self-enforcing international environmental agreements emerging depend partly on understanding preferences for pollution abatement benefits which occur both within and outside domestic boundaries. Using the example of marine plastic pollution in the North Atlantic, we explore individual preferences for reductions in pollution damages in multiple spatial settings, specifically those that occur at home, internationally and abroad. We administered a cross-country discrete choice experiment in the UK and US concerning a bilateral transboundary pollution management program between the two countries. We find that individuals value reductions both at home and abroad (with higher benefits for domestic reductions in plastics pollution), with evidence of heterogenous preferences within and between our two country samples. Our findings suggest that self-enforcing international environmental agreements provide welfare benefits beyond just the value of domestic plastic reductions, but these “non-domestic” benefits seem not to exist for all citizens.

1. **Introduction**

The unique challenges associated with transboundary pollutants are increasingly recognized by economists (Livingston, 1989). Although often studied with regard to global climate change and greenhouse gas emissions—perhaps the most archetypal example of a global pollutant—a broad array of transboundary environmental externalities threatens social welfare. The core challenge of management for these pollutants is well known—abatement expenditures by one nation confer benefits on many nations, creating incentives to free ride on the abatement efforts of others (Livingston, 1989). Efficient outcomes hence require international environmental agreements (IEAs) able to accommodate variations in marginal abatement and damage costs across affected nations (Hoel & Schneider, 1997; Maler, 1989). Yet while the theory of transboundary pollutants and IEAs has been explored in some depth (Finus, 2008), there has been less work on the empirical conditions under which effective control is likely to emerge for different types of pollutants—including whether public preferences are consistent with those required for effective IEAs to emerge. This lack of insight is particularly profound for situations beyond the case of climate change.

Barrett (1994) argued that a useful way to think about these transboundary, common property dilemmas is through the lens of self-enforcing IEAs, wherein optimal solutions depend on the extent to which individual countries believe it to be in their self-interest to cooperate over emission reductions. Theoretical contributions show that the emergence and stability of coalitions for IEAs of this type depend on a restrictive set of antecedent conditions (Barrett, 2003). The most obvious of these is that countries must value pollution reductions that occur within their own borders—whether the emissions originated from domestic or foreign sources. However, cooperation is also more likely if the public in each country values pollution reductions *elsewhere* (Kolstad, 2014; van der Pol et al., 2012). Values for non-domestic pollution reductions might arise for various reasons, for example because a person spends time in those countries, holds non-use values for reductions in pollution damages to others (paternalistic altruism), or for other reasons. These issues can be particularly important to IEA emergence and stability when programs can be designed to target (non-uniform) pollution reductions in particular areas.

The economics literature provides extensive evidence on public values (often quantified in terms of willingness to pay (WTP)) for changes in global environmental conditions or iconic resources valued worldwide.[[1]](#footnote-1) However, there is limited evidence on relative values among a domestic public for otherwise identical, quantified improvements in environmental conditions that might occur either domestically or in affected foreign/international locations. To borrow an example from the case study introduced below, an IEA for marine-plastics reductions might lead to different levels of measurable pollution reductions in domestic and foreign coastal waters. To what extent might the public in one country (say, the US) value coastal pollution reductions that occur elsewhere (say, the UK), compared to otherwise identical reductions at home? To what extent do residents of both countries express positive values for changes that occur internationally, beyond the territorial waters of each nation? And what do these relative values imply about the likelihood that different types of IEAs could gain simultaneous approval by the public in both countries? The standard “textbook” assumption—that only domestic pollution reductions are valued in each country—is likely incorrect and leads to potentially misguided perspectives on the potential for stable international IEAs to emerge.

When considering preferences of this type, it is also important to recognize that preferences for reductions in transboundary pollutants may be heterogenous (Carlsson et al., 2012), and perhaps strongly so. For example, there may be broad classes (or types) of individuals in each country with distinct and categorizable preferences for non-domestic pollution reductions. The type of heterogeneity encountered within the context of transboundary pollutants may differ from that commonly encountered when seeking to estimate WTP for domestic environmental improvements or public goods. In the latter case, one might assume that WTP is predominantly positive—with heterogeneity in the magnitude of (mostly positive) values. However, in the case of transboundary pollutants, one can envision cases in which certain members of the public might systematically disfavor agreements that produce greater improvements in other countries—leading to potentially *negative* values for certain types foreign or international improvements. Preference heterogeneity of this type is not merely an academic or scholarly curiosity. Polarized preferences can have profound implications for the feasibility of cooperative policy solutions for global environmental problems (Farrell 2016), including the extent to which stable agreements are likely to emerge.

Although issues such as these have been considered in theoretical and conceptual terms, the economics literature provides limited evidence on (1) how residents in multiple countries are affected by different types of non-uniformly mixed transboundary pollutants, (2) how these residents value pollution reductions domestically versus internationally, (3) how residents’ preferences are affected by the specific location of non-domestic pollution reductions, and (4) how these preferences vary over populations in different countries. Information of this type is unavailable for multiple transboundary pollutants recognized as urgent global policy concerns, including marine plastics (Vince and Hardesty 2018; Abate et al. 2020). We are aware of no published research that systematically evaluates public preferences and WTP for marine plastics reductions from the vantage point of transboundary improvements across different domestic and non-domestic locations.

To better understand the types of preferences that can emerge for transboundary pollutants of this type—and the potential implications for management solutions—this article presents the results of a multi-country discrete choice experiment (DCE) designed to evaluate preferences for joint actions to control and remediate marine plastics pollution in the North Atlantic Ocean. We focus on preferences for potential bilateral US and UK control programs that would reduce pollution damages across multiple possible locations, including: (a) domestically, (b) international waters, and (c) foreign waters. Unlike past studies of public preferences for marine plastics reductions, valuation scenarios directly quantified measurable (absolute and relative) changes in plastic that would be realized across these different areas. To eliminate possible confounding of preferences (or scenario adjustment, Cameron et al. 2011) due to assumed burden-sharing within agreements of this type, we control for the degree to which costs would be shared across collaborating partners. Parallel DCEs are implemented on samples in the United States (US) and United Kingdom (UK), enabling parallel preferences (and preference heterogeneity) for foreign versus domestic improvements to be compared across the two countries.

Results show evidence of (strongly) divergent preferences for marine plastics reductions both within and between the US and UK samples—including divergence whether and how improvements in foreign and international waters are valued. These findings highlight the potential challenges in developing stable coalitions for attenuating the marine plastics problem. Results also suggest the limitations of studies seeking to elicit values for marine plastics policy solutions in primarily domestic terms. For example, foreign and international impacts are not always preference-neutral for domestic populations—but can be associated with either positive or negative welfare impacts depending on the group considered. Taken together, results suggest the presence of complex and heterogeneous preferences for domestic and foreign marine plastics reductions that defy “one size fits all” policy solutions.

The remainder of the paper is organized as follows. Section 2 describes the case study of marine plastic as a transboundary pollution example in more detail. Section 3 and 4 describe DCE study design and methodology. Section 5 outlines study results including the latent class models and the associated willingness to pay (WTP) estimates. Section 6 discusses the differences across the two data sets and concludes the paper.

1. **Understanding public preferences for transboundary pollution control.**

Widely studied examples of transboundary pollution include acid rain and global warming. In the latter case, greenhouse gases (GHGs) released into the atmosphere from any one country cause environmental damages worldwide (see Du et al., (2020)). Acid deposition from fossil fuel emissions is an example where impacts tend to be regional rather than global (Bateman et al., 2005). In these and other cases, damages imposed by a given quantity of pollution emissions from a particular location can be experienced unequally across nations, due to heterogeneous biophysical, economic and behavioral factors. Furthermore, a country’s decisions over whether and how to respond to such environmental problems depends on the degree to which it values reductions in both domestic and foreign territories (Anthoff & Tol, 2010). A common assumption is that residents of each country fail to consider pollution damages realized in other countries when determining pollution controls (or give insufficient consideration to these damages), because they do not realize those damages directly. These conditions result in market failure for managing transboundary pollution: incentives exist for countries to free ride on the abatement actions of other countries, making cooperation and the stability of IEAs difficult to uphold (Maler, 1989).

The theoretical and game-theoretic properties of IEAs for transboundary pollution management have been studied extensively (Barrett, 1994; de Frutos & Martín-Herrán, 2019; Finus, 2008; Maler, 1989; Missfeldt, 1999; Siqueira, 2003). The solutions that emerge from these theoretical exercises depend on assumptions regarding the preferences (or values) held in each country for pollution reductions that occur in that country and elsewhere. That is, one cannot predict what type of transboundary management will emerge (or will be viewed as optimal) without information on how individuals in different countries value pollution reductions that occur domestically and internationally. Although some empirical work in the economics literature has addressed related topics (see below), the issue has remained unexplored for many, if not most, important transboundary pollutants. Responding to this general lack of attention in the literature, this paper aims to understand individuals’ preferences for reductions of a transboundary pollutant which occur across multiple national and international locations.

The idea that households might value environmental improvements beyond the borders of their home country is not new. Multiple studies in the economics literature seek to estimate WTP for environmental improvements that occur at least in part beyond respondents’ home countries. The literature addressing WTP for climate-change policy frequently addresses impacts (either implicitly or explicitly) that occur outside the home country (e.g., Carlsson et al., 2011; Lee & Cameron, 2008; Ščasný et al., 2017; Svenningsen & Thorsen, 2020). Examples beyond climate change include WTP estimation for improvements to iconic ecosystems such as the Amazon rainforest (Siikamaki et al., 2019; Strand et al., 2017), migratory or multi-country species conservation (e.g., Diffendorfer et al., 2014; Haefele et al., 2019; Jin et al., 2010; Ressurreição et al., 2012; Vogdrup-Schmidt et al., 2019), or improvements to transboundary parks or conservation areas (e.g., Bakhtiari et al., 2018; Valasiuk et al., 2017). Other studies estimate WTP for potential natural hazard reductions that occur in foreign waters (Loureiro & Loomis, 2013), for improvements to water quality in international water bodies (e.g., Ahtiainen et al., 2014) for or for environmental or ecosystem service improvements that span different countries (e.g., Bateman et al., 2005; Dallimer et al., 2015; Obeng & Aguilar, 2021). Some of these studies are motivated—as this one is—by the potential relevance of the studied preferences for international cooperation (e.g., Bakhtiari et al., 2018; Dallimer et al., 2015; Jin et al., 2010).

Yet despite this heterogeneous body of literature, there are few preference elicitations framed directly in terms of a formal bilateral or multilateral IEA for transboundary pollution reduction (beyond the case of climate change), where different types of measurable domestic impacts, foreign impacts and cost-sharing vary systematically across partners for different types of agreements. For example, to what extent do residents of each country value pollution reductions in their own country (say, plastic reductions in domestic waters) relative to otherwise identical reductions elsewhere, as part of a bilateral IEA? Among the key dimensions of these values are the extent to which parallel preferences for foreign outcomes emerge from each side of the agreement (e.g., WTP of Country A’s residents to reduce impacts in Country B versus the parallel WTP of Country B’s residents to reduce impacts in Country A). There are also few studies that explore differences between preferences related to improvements to international waters versus those that occur within the territorial waters of individual nations.

* 1. **Marine plastics pollution as a transboundary control problem.**

We implement the analysis for the case study of marine plastic pollution, a transboundary pollutant that is increasingly acknowledged as an urgent global policy concern (Lau et al., 2020; Vince & Hardesty, 2018). Although the issue has been given little attention by economists, the IUCN describes it as “the most serious problem affecting the marine environment” (Thevenon & Carroll, 2015, pg. 9). Every nation emits plastic pollution into the ocean, (via river mouths, mismanaged landfills, littering, or seaborne sources). This plastic accumulates in different marine areas. Due to ocean currents, tides and winds, each country’s plastic waste can remain in domestic waters or be transported internationally. As such, the damages that any one country suffers from marine plastics pollution levels depend not only on its own emission and abatement measures, but on those of other countries. Damages suffered by a given country depend on the amount of plastic circulating in the ocean, movement of this plastic over time and space, the exposure of this country (e.g., length of coastline), the distribution of differing marine and coastal habitats, and the preferences of local people.

Globally, around 20% of marine plastic originates from ocean-based sources (e.g. the fishing and shipping industries, offshore oil and gas extraction, and recreational boaters) whilst 80% originates from land-based sources (Mouat et al., 2010). Land-based waste ends up as marine litter through many channels: public litter which enters waterways, mismanaged municipal waste, sewerage-relate debris, and storm water overflow discharges (Kirk, 2018; Mouat et al., 2010). It has been estimated that, of this land-based plastic, about 80% is transmitted to the sea through 1,000 rivers globally (Meijer et al., 2021) making rivers the most important vector of this pollutant. Due to factors such as ocean currents and weather patterns, as well as country-specific human factors such as coastal population density, waste management policies and existing environmental protection, plastic inputs to a shared ocean not only differ across countries, but accumulate at different rates in different areas of the ocean (Barnes et al., 2009).

Controlling plastic pollution at sea requires local, national, and international control efforts since actions taken by any one country to reduce plastic inputs to their own territorial waters can contribute to pollution reductions in international waters and to beaches and coastal waters of other countries, due to ocean currents and wind moving plastics long distances across space (Cózar et al., 2014; Law et al., 2010). Relevant actions include prevention through improved waste management and wider use of plastic substitutes, to removal efforts like beach clean ups. The Ocean Conservancy (2019) report recorded worldwide beach clean-up efforts which involved over 1 million individuals collecting GBP 23.3 million worth of marine pollution. Mouat et al. (2010) quantify the economic impact marine litter has on communities in the UK, estimating that the UK spends around EUR 18 million/year on beach clean ups, EUR 2.4 million/year to clean harbors, whilst costs to the fishing industry (e.g. from entangled propellers and blocked intake pipes) amount to between EUR 11.7-13 million/year in damage. Further large scale initiatives such as the US National Oceanic and Atmospheric Administration’s (NOAA) mission aims to remove plastic pollution from large, remote, marine protected environments (Dameron et al., 2007). NOAA’s ten-year report of their Marine Debris Program estimates over 18,800 metric tons of waste was removed, spending USD 11.8 million on prevention, removal and research projects (NOAA, 2020). The United Nations Environment Program (2014) report estimates the overall economic impact of plastic pollution to marine ecosystems to be at least USD 13 billion per year (UNEP, 2014).

Despite the widely recognized challenges associated with marine plastics, few studies have considered preferences for marine-plastics reductions in general (Abate et al., 2020; Beaumont et al., 2019; Leggett et al., 2018; McIlgorm et al., 2011; Meginnis et al., 2022; NOAA, 2019; Tyllianakis & Ferrini, 2021), and none of these have addressed the transboundary nature of the problem explicitly. As a result, the literature provides no insight into whether and how public values are consistent with the emergence of stable agreements for marine-plastic reductions, or more broadly how values for marine plastics reductions depend on where those reductions are realized. This article seeks to provide insight into this issue. Specifically, we seek to understand how citizens of two countries sharing the same ocean resource value possible future reductions in marine plastics pollution which might come about as the result of an IEA between these two nations, including how preferences vary for reductions in plastics in domestic, international and foreign-country waters. In doing so, we also present an example of how WTP elicitation can be framed more broadly around a bilateral agreement for transboundary pollution reductions.

1. **The Discrete Choice Experiment**

The DCE was designed to assess preferences and welfare changes from prospective pollution reductions in US, UK and international waters from the perspectives of US and UK citizens. Preference elicitation was framed within the context of a formal bilateral IEA for marine plastic reductions in the North Atlantic. The DCE was focused on a bilateral agreement between two countries to reduce survey complexity and thereby lessen cognitive burden on respondents.[[2]](#footnote-2) At the same time, the DCE design maintained the underlying characterization of a bilateral agreement for transboundary pollutant reductions in that plastic waste emitted/reduced from the US can directly impact the UK, and vice versa. We focus on the UK and the US as two of the most important players in any future agreement to reduce marine plastic pollution in the North Atlantic.[[3]](#footnote-3)

Independent, parallel DCE surveys were administered in each country (the UK and US). The two DCE versions are mirrors of each other with respect to the transboundary nature of the problem, the only difference being that the choice context within each is oriented to the country where it was administered. That is, the UK version considers the UK as the home country and the US as the foreign country while the mirror US version considers the US as the home country and the UK as the foreign country. Each version classified existing plastic accumulation and future plastic waste reduction into different marine locations across home, foreign and international waters. This stratification was central to communicating the transboundary nature of marine plastics policy.

For the remainder of this paper, we use *home country* to refer to the country where the survey was administered (i.e., the UK for the UK version) and *foreign country* as the other participating country which would benefit from emission reduction in the home country (i.e., the US for the UK version). The interconnectedness of oceans implies that any one country can reduce their own plastic pollution through abatement measures taken at home, but these will also impact, and be impacted by, efforts taken abroad. Our primary hypotheses concern how WTP varies according to whether otherwise identical pollution reductions occur in: (1) beaches or coastal waters in the home country, (2) beaches and coastal waters of the foreign country, or (3) international waters. We also consider preferences around the allocation of the cost burden that could be reached as part of bilateral IEA. A visual representation of how the North Atlantic was segregated in the study is shown in Figure 1.

**3.1. Defining the spatial extent of (reductions in) marine plastic pollution**

Grounded in the structure outlined above, each DCE question included three choice alternatives (or options), each characterized by six attributes. Alternatives 1 and 2 presented possible options for a bilateral marine plastics pollution reduction program involving the US and UK. Option 3 was a business-as-usual status quo with no change in marine plastic and zero household cost. In addition to attributes corresponding to the quantity and location of plastic reductions detailed below, each alternative included an attribute communicating how program costs are split between the two participating countries, as well as a hypothetically binding cost to each household in the home country, in increased taxes and fees, required to implement the program.[[4]](#footnote-4)

The first four choice attributes quantified baselines and potential reductions in plastic abundance or density in different areas of the marine environment. These reductions are stated both in terms of the absolute number of pieces of waste plastic and as percentage reductions relative to status quo levels (Table 1). Baselines and possible changes for these attributes were informed by data on current plastic levels in various marine and coastal environments and modeling of potential changes in those levels that could occur due to changes in plastic disposal and clean-up in various locations.[[5]](#footnote-5)

The first attribute reflected average plastic pollution density on home country beaches (pieces/100m). For the UK version, the outcome was reductions in plastic waste on beaches in England, North Ireland, Scotland, and Wales. For the US version, the outcome was reductions in plastic waste on US East Coast beaches (excluding the Western coast of Florida along the Gulf of Mexico). Possible reductions relative to the status quo ranged from 0% to 90%. To aid in comprehension of this attribute, respondents were shown graphical illustrations of different quantities of plastic as they might appear on a representative beach.

The second attribute quantified plastic density in the home country’s coastal waters (pieces/km2). Coastal waters were defined as waters one kilometer out from shore, which mainly accumulate waste from the home country’s mismanaged land waste (Lebreton, Egger, & Slat, 2019). Possible reductions in this plastic density relative to the status quo ranged from 0% to 20%.[[6]](#footnote-6)

The third attribute quantified average plastic density in international waters (items/km2). This was described the same way in both survey versions as the area belonging to no single country, but where plastic waste accumulates from all countries, far out in the open North Atlantic Ocean. The general location and relative plastic density in these international accumulation areas was shown in maps displayed in the online survey. Possible reductions in this plastic density relative to the status quo ranged from 0% to 15%. These possible changes were lower than those in coastal waters, due to the increased difficulty of cleaning up pollution in deep ocean waters.

The fourth and final biophysical attribute quantified average plastic density over the foreign country’s beaches and coastal waters (items/km2). This attribute combined effects on foreign beaches and coastal waters into one attribute, based on input from focus groups that this distinction was not salient to respondents when considering foreign country reductions. Possible reductions in this plastic density relative to the status quo ranged from 0% to 20%.

Figure 1 – Visual breakdown of North Atlantic segments

Shape

Description automatically generated

Table 1 summarizes the attributes and levels included in the DCE. For the plastic baselines and reductions, respondents were presented with both percentage decrease figures and the corresponding absolute value changes, following guidance in Johnston et al. (2012). Both survey versions considered the same percentage decreases; however, given the differing status quo (baseline) values in the UK and the US, the absolute reduction levels differed across the two versions.

**3.2. Questionnaire design and administration**

Survey materials were developed over a roughly two-year period from February 2019 to November 2020. Both the survey questionnaire and the included DCE were tested and iteratively revised with input from 8 focus groups, with an equal number conducted in the US and UK. A subsequent pilot survey was carried out with 400 participants in the UK and the US.

The survey instrument was divided into several parts. First, we began by presenting respondents with a description of what is meant by ‘plastic items’, which were described as ranging from small items, such as a cigarette butts, to plastic water bottles and shopping bags, to large accumulations of derelict fishing gear.[[7]](#footnote-7) We also asked questions about respondent’s experience with plastic wastes at home and abroad. Second, we described the program attributes to respondents and presented them with example choice sets. Third, respondents were asked to answer five DCE questions following the structure outlined above. Finally, the survey included a set of follow-up questions relating to environmental attitudes, consequentiality statements and household characteristics. A copy of both survey versions are available on request.

The survey incorporated multiple elements to support valid preference elicitation, following recommended practices (Johnston et al. 2017). These included prompts and acknowledgement questions to emphasize payment and policy consequentiality, reminders of the household’s budget constraint, an overview of valid reasons why a household might vote for either Option 1, 2 or 3, and a reminder that “Whatever your reasons, choosing either Option 1, 2 or 3 is legitimate.” Respondents were asked to acknowledge that they were free to select whichever alternative they considered to be best for their household (i.e., a positive cost program or the zero-cost status quo), acknowledge they would consider the cost seriously, and only select options for which they were willing to pay the stated amount. Respondents were instructed to consider each choice question as an independent and hypothetically binding vote. To emphasize consequentiality, respondents were informed that the survey “has been developed in partnership with government officials who will consider survey results in making their decisions.”[[8]](#footnote-8) The choice was framed as a vote between the three competing policy options in each question.

The experimental design for the DCE was developed using a Bayesian Db-efficiency criterion for a choice model covariance matrix (Scarpa & Rose, 2008). Although optimized for Db-efficiency, S-efficiency was also used to evaluate sample sizes required to estimate preference parameters for each assumed utility specification (Rose et al., 2008; Rose & Bliemer, 2009; Scarpa & Rose, 2008). Diffuse priors were applied (Ferrini & Scarpa, 2007), with signs based on information from focus groups, expert opinion, theory and findings from the literature. For each survey, the resulting design included 75 profiles blocked into 15 survey versions, each with 5 choice tasks consisting of two program alternatives and a status quo option. The choice sets were designed using Ngene (ChoiceMetrics, 2018) whilst the survey was created and administered using Lighthouse Studio (Sawtooth, 2018).

The final survey was administered in the UK and US in March 2021. A total of 4,690 responses were collected with 2,028 respondents in the UK and 2,662 in the US. Each respondent answered five choice questions, resulting in 10,140 and 13,310 choice observations for the UK and US survey versions, respectively. Respondents were randomly sampled using an online panel service that aimed to obtain a nationally representative sample according to gender, geographical location and age. In the UK, respondents from the entire country were sampled, whereas only participants from US states on the eastern seaboard were recruited.[[9]](#footnote-9)

Table 1- Attribute descriptions and levels, UK version (US version)\*

|  |  |  |
| --- | --- | --- |
| Variable Name*ŧ* | Description | Levels |
| Beach\_Q | Reduction in the amount of plastic on UK beaches (items per 100 meters)  *Reduction in the amount* *of plastic on US East Coast beaches (items per 100 meters)* | Percentage decrease: 0ʈ,25,50,75,90  Absolute values:  85, 64, 42, 21, 8  *Absolute values:*  *217, 163, 108, 54, 22* |
| Coastal\_Q | Reduction in the amount of plastic in UK coastal waters (items per sq km)  *Reduction in the amount* *of plastic in US coastal waters (items per sq km)* | Percentage decrease:  0 ʈ, 5, 10, 15, 20  Absolute values:  170, 161, 153, 144, 136  *Absolute values:*  *435, 413, 391, 370, 348* |
| Intl\_Q | Reduction in the amount of plastic in international waters (items per sq km)  *Reduction in the amount* *of plastic in international waters (items per sq km)* | Percentage decrease:  0 ʈ, 3, 5, 10, 15  Absolute values:  1807, 1753, 1717, 1626, 1536  *Absolute values:*  *1807, 1753, 1717, 1626, 1536* |
| Foreign\_Q | Reduction in the amount of plastic in US East Coast beaches and coastal waters (items per sq km)  *Reduce the amount of plastic in UK beaches and coastal waters (items per sq km)* | Percentage decrease:  0 ʈ, 5, 10, 15, 20  Absolute values:  2605, 2475, 2344, 2214, 2084  *Absolute values:*  *1020, 969, 918, 867, 816* |
| Csplit\_Qhome | Percentage of program cost paid by home country and foreign country | Percentage split:  25% home, 75% foreign ʈ  50% home, 50% foreign  75% home, 25% foreign |
| Cost | Cost to your household per year paid through an increase in annual taxes and fees | £0 ʈ, £35, £60, £75, £150, £230  *$0* ʈ*, $50, $80, $100, $200, $300* |
| None | Alternative specific constant for the status quo alternative | 1ʈ, 0 |
| \* The italicized description and levels explain the US version attribute description and associated levels. Both versions had the same percentage levels, but due to the different status quo starting values, the absolute values differ across the two versions.  *ŧ*The Q in each attribute variable name corresponds to the percentage decrease. For example, beach\_25 represents a decrease in beach plastic by 25%. For the Csplit\_home attribute, the Q signifies the home country share, i.e., in the UK survey version, Csplit\_25home means 25% paid by the UK and 75% paid by the US.  ʈ Represents the status quo base level which is omitted for analysis. | | |

Figure 2 presents an example choice card from the UK (panel A) and US (panel B) versions.

Figure 2- Example DCE choice card

Graphical user interface, application, table

Description automatically generated

Note: Figure 2.A is an example choice card from the UK version and Figure 2.B is from the US version.

* 1. **Summary Statistics**

Sample summary statistics are presented in Table 2.

Table 2- Sample statistics for UK and US versions

|  |  |  |
| --- | --- | --- |
| Socio-Economic Variable\* | UK Version (N=2,028) | US Version  (N=2,662) |
| Age | 47 | 59 |
| Female | 0.49 | 0.40 |
| Number of adults ≥18 in household | 2.10 | 1.98 |
| Children < 18 | 0.49 | 0.35 |
| Education level:  No formal qualification  GCSEs/O-levels (Less than HS)  A-levels (HS or equivalent)  Diploma/technical qualification (Associate’s degree)  Bachelor’s degree  Master’s degree  PhD or higher | 0.39  0.19  0.16  0.17  0.30  0.11  0.03 | 0.0008  0.0083  0.21  0.14  0.35  0.22  0.06 |
| Combined annual household income:  ≤£12,500 (≤$15,000)  £12,501-£20,000 ($15,001-$24,999)  £20,001-£30,000 ($25,000- $34,999)  £30,001-£50,000 ($35,000- $49,999)  £50,001-£70,000 ($50,000-74,999)  £70,001-£100,000 ($75,000-$99,999)  £100,001-£150,000 ($100,000-$149,999)  ≥£150,001 ($150,000-$199,999)  (≥$200,000) | 0.13  0.14  0.22  0.27  0.12  0.07  0.02  0.01 | 0.38  0.57  0.57  0.11  0.18  0.17  0.22  0.09  0.06 |
| Employment:  Employed full-time  Employed part-time  Self-employed  Temporarily unemployed  Unable to work due to sickness/disability  Looking after a home full-time  Student  Retired | 0.40  0.12  0.07  0.05  0.03  0.05  0.05  0.20 | 0.35  0.06  0.04  0.04  0.03  0.03  0.01  0.44 |
| \*Income levels in brackets () represent equivalent, or near equivalent, levels presented in the US version. Numbers may not add to 1 due to rounding and due to the US version not requiring all follow-up questions to be mandatory (due to ethics restrictions). | | |

1. **Empirical Model**

The presented model results are based on a latent class model (LCM) specification, designed to study the heterogeneity in preferences across different groups of households in each country. Models of this type allow parameters to differ systematically across individuals in a finite set of discrete classes (Train, 2009). The LCM allows us to explore preference heterogeneity and use individual characteristics to explore sources that explain this heterogeneity (Boxall & Adamowicz, 2002). This specification of the model corresponds to (a) the goal of the study to evaluate preferences as they vary across different population groups, and (b) the corresponding hypothesis that these different groups may have strongly divergent preferences for marine plastics reductions that might not be represented adequately by the types of continuous parameter distributions (e.g., normal, lognormal) typically assumed within mixed logit models (Train, 2009). This choice was supported by empirical comparison of different preliminary model specifications including random parameters mixed logit. We estimate independent latent class models for each of the two country datasets separately, using the same individual covariates in each dataset as predictors of class membership probability.

* 1. **Model Specification**

The LCM is grounded in a standard random utility framework. We assume that individual *n* is presented with a choice task consisting of three alternatives (*J = Option 1, Option 2, Option 3*): Option 1 and Option 2 represent competing proposed marine plastic pollution reduction programs and Option 3 represents the status quo (i.e., no program). Options 1 and 2 deliver pollution reductions at different spatial levels at specified costs to the individual but Option 3 offers no changes at zero cost. The utility function for individual *n* for choice alternative *i* in choice situation *t* can be expressed as:

Eq.1

where is a vector containing the attribute levels of alternative *i* in choice situation *t* as well as an indicator if it is a change option or the status quo. The utility function in Equation 1 includes the attributes described in Table 1: (i) four location-of-pollution-reduction attributes (each entering the model with four dummies, with the base level being 0% reduction); (ii) two cost split attribute dummies (with the omitted baseline level being the home country paying 25% and the foreign country paying 75%); (iii) cost attribute, entering as a continuous variable; and (iv) an alternative specific constant (ASC) indicating the status quo alternative.[[10]](#footnote-10) The parameters in β in Equation 1 are the attributes’ preference weights to be estimated, and εnit is an i.i.d. error term following an extreme value Type I distribution.

The LCM explores distinct groups of respondent preferences by assuming respondents fall into *S* number of classes, which are unobservable by the researcher but probabilistically identifiable based on stated choices. Preferences within classes are assumed to be homogeneous but differ across classes, i.e. for . Following standard practice, the probability that individual *n* selects their utility-maximizing alternative *i* in situation *t* conditional on their membership in class *s* is expressed as:

Eq. 2

The probability that individual *n* belongs to class *s* is equally modelled using the multinomial logit model,

where Eq. 3

where Zn is a vector of respondent-specific covariates which help explain class membership and γs is a conforming parameter vector to be estimated.[[11]](#footnote-11) To reduce the chances of the model converging at a local optimum, we used 500 different sets of starting values and selected the model with the best fit to the data.

Variables specified in the class membership model include sociodemographic characteristics (e.g., gender), environmental attitude measures (e.g., concern towards marine plastic pollution), and responses to three statements about consequentiality.[[12]](#footnote-12) The Likert-scale covariates are shown in Table 3, along with the distribution of responses and mean scores. [[13]](#footnote-13) Each was included as a continuous variable in the class membership model. Socio-economic variables (i.e., gender, income and age, shown in Table 2) were also included as continuous variables aside from gender, which was specified as a dummy variable (equal to 1 if female).

Table 3- Distribution of responses to Likert-Scale questions by survey version

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| Statement | Variable Name | Strongly Disagree | Disagree | Neither Agree nor Disagree | Agree | Strongly Agree | Mean Scoret |
| ***UK Version*** | | | | | | | |
| I already pay enough in taxes. | *pay\_enough* | 1.58% | 5.77% | 34.66% | 32.94% | 25.05% | 3.74 |
| I would like to reduce marine plastic, but it is not my responsibility to pay for it. | *not\_my\_respon* | 4.49% | 21.50% | 32.25% | 25.59% | 16.17% | 3.27 |
| I think it is really important to reduce the amount of plastic in marine environments. | *important\_reduce* | 0.49% | 0.94% | 7.54% | 31.26% | 59.76% | 4.48 |
| My responses to this survey will have an influence on whether these measures will be implemented or not. | *policy\_con* | 3.25% | 7.79% | 35.95% | 36.05% | 16.96% | 3.55 |
| If the government carries out the measures described above, I believe I will be charged a tax to support them. | *payment\_con* | 0.59% | 2.66% | 19.72% | 49.56% | 27.47% | 4.00 |
| If the program is implemented, I think there will be a change in the amount of plastic in the North Atlantic. | *impact\_con* | 1.58% | 5.42% | 25.94% | 47.49% | 19.58% | 3.78 |
| ***US Version*** | | | | | | | |
| I already pay enough in taxes. | *pay\_enough* | 1.70% | 5.55% | 30.75% | 30.98% | 31.02% | 3.84 |
| I would like to reduce marine plastic, but it is not my responsibility to pay for it. | *not\_my\_respon* | 7.72% | 24.75% | 31.53% | 18.83% | 17.18% | 3.12 |
| I think it is really important to reduce the amount of plastic in marine environments. | *important\_reduce* | 0.49% | 0.64% | 6.92% | 35.90% | 56.05% | 4.46 |
| My responses to this survey will have an influence on whether these measures will be implemented or not. | *policy\_con* | 3.95% | 9.77% | 41.54% | 30.19% | 14.55% | 3.41 |
| If the government carries out the measures described above, I believe I will be charged a tax to support them. | *payment\_con* | 0.79% | 1.99% | 20.08% | 49.08% | 28.06% | 4.01 |
| If the program is implemented, I think there will be a change in the amount of plastic in the North Atlantic. | *impact\_con* | 2.59% | 5.49% | 22.89% | 48.65% | 20.38% | 3.78 |
| \* US Version may not add up to 100 due to US participants able to skip questions, as was necessary for IRB approval.  tCoded as 1- strongly disagree to 5- strongly agree. | | | | | | | |

* 1. **Choice Model Results**

To explore preferences for pollution reduction in (1) spatially distinct areas and (2) preference heterogeneity, independent but parallel LCM specifications are presented for the UK and US choice data sets. [[14]](#footnote-14) To determine the most appropriate number of classes for these LCMs,[[15]](#footnote-15) we compare model fit criteria such as log-likelihood, Bayesian information criterion (BIC), and the pattern (signs and significance) of parameter estimates for both LCMs estimated with sequentially increasing numbers of latent classes (Danley et al., 2021). Although BIC indicates improving model fit for up to five classes, models with more than 3 classes have positive cost parameter estimates and/or multiple classes with insignificant parameters indicating possible issues of overfitting, not uncommon in latent class modelling (see Danley et al., 2021; Scarpa & Thiene, 2005). We therefore present results the 3-class choice model for both the UK and US datasets.

We begin with an overview of model results and the latent classes that emerge. This is followed by a more detailed discussion around preferences for plastic reductions in different locations, among the different classes in each country. First, we consider LCM results for the UK version presented in Table 4. The model identifies 3 preference classes which can be described in terms of the choice model and composition of classes based on the included covariates. The majority of respondents probabilistically belong in Class 1 (58%), with the remaining belonging to Class 2 (24%) and Class 3 (18%).

Table 4- LCM regression results for UK Version

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  | Class 1 | | Class 2 | | Class 3 | |
| Attributes | Coef. | s.e. | Coef. | s.e. | Coef. | s.e. |
| beach\_25 | 0.293\*\*\* | (0.071) | -0.451 | (0.762) | 0.952\*\*\* | (0.251) |
| beach\_50 | 0.476\*\*\* | (0.071) | 0.698 | (0.461) | 0.849\*\*\* | (0.260) |
| beach\_75 | 0.647\*\*\* | (0.068) | 0.678 | (0.449) | 1.199\*\*\* | (0.263) |
| beach\_90 | 0.688\*\*\* | (0.078) | 0.609 | (0.480) | 1.241\*\*\* | (0.276) |
| coastal\_5 | 0.152\*\* | (0.074) | -0.632 | (0.416) | 0.293 | (0.262) |
| coastal\_10 | 0.208\*\*\* | (0.073) | -0.987 | (0.552) | 0.563\*\* | (0.265) |
| coastal\_15 | 0.419\*\*\* | (0.071) | -0.407 | (0.405) | 0.162 | (0.248) |
| coastal\_20 | 0.345\*\*\* | (0.076) | -0.719 | (0.464) | 0.805\*\*\* | (0.262) |
| intl\_3 | 0.003 | (0.068) | 0.599 | (0.520) | 0.046 | (0.223) |
| intl\_5 | 0.164\*\* | (0.073) | 0.482 | (0.544) | -0.508\* | (0.262) |
| intl\_10 | 0.064 | (0.074) | 0.627 | (0.527) | 0.125 | (0.228) |
| intl\_15 | 0.157\*\* | (0.064) | 0.863\* | (0.482) | -0.483\* | (0.242) |
| foreign\_5 | -0.078 | (0.077) | -0.079 | (0.414) | 0.241 | (0.221) |
| foreign\_10 | 0.090 | (0.067) | 0.155 | (0.416) | -0.162 | (0.210) |
| foreign\_15 | 0.032 | (0.071) | -0.343 | (0.514) | 0.508\*\* | (0.248) |
| foreign\_20 | -0.109 | (0.071) | -0.416 | (0.469) | -0.014 | (0.252) |
| csplit\_50home | 0.175\*\*\* | (0.053) | -0.360 | (0.402) | -0.138 | (0.217) |
| csplit\_75home | -0.166\*\*\* | (0.060) | -0.175 | (0.363) | -0.834\*\*\* | (0.190) |
| ASC | -2.041\*\*\* | (0.108) | 4.558\*\*\* | (0.699) | -3.651\*\*\* | (0.438) |
| Cost ($/year/HH) | -0.007\*\*\* | (0.000) | 0.002 | (0.002) | -0.060\*\*\* | (0.005) |
| Class Membership Model | | | | | | |
| Class Probability | 0.58 | | 0.24 |  | 0.18 |  |
| Intercept | . | | 0.702 | (0.576) | -0.957 | (0.677) |
| pay\_enough | . | | 0.913\*\*\* | (0.097) | 0.575\*\*\* | (0.098) |
| not\_my\_respon | . | | 0.712\*\*\* | (0.085) | 0.158\*\* | (0.080) |
| important\_reduce | . | | -0.820\*\*\* | (0.104) | -0.343\*\*\* | (0.117) |
| policy\_con | . | | -0.225\*\*\* | (0.083) | -0.182\*\* | (0.086) |
| payment\_con | . | | -0.584\*\*\* | (0.097) | -0.082 | (0.108) |
| impact\_con | . | | -0.530\*\*\* | (0.091) | -0.326\*\*\* | (0.097) |
| age | . | | 0.025\*\*\* | (0.004) | 0.019\*\*\* | (0.005) |
| income | . | | -0.640\*\*\* | (0.193) | -0.651\*\*\* | (0.215) |
| female | . | | 0.358\*\*\* | (0.139) | 0.722\*\*\* | (0.150) |
| Log Likelihood | -7482.285 | |  |  |  |  |
| BIC | 15573.716 | |  |  |  |  |
| Adj. R2 | 0.489 | |  |  |  |  |
| # of individuals | 2027 | |  |  |  |  |
| Note: \* p<0.10, \*\* p<0.05, \*\*\* p<0.01 | | | | | | |

Results show clear evidence of preferences that differ strongly across classes. Although we focus the discussion around preferences for plastic reductions in different domestic and foreign locations, these differences extend across all variables. The first class (Class 1) in the UK model can be characterized loosely as *“home and international reducers”.*These respondents have significant and positive marginal utility for all plastic reduction levels at home beaches, home coastal waters, and international waters (at the 5% and 15% reduction level). However, they have insignificant preferences for reductions in foreign waters.[[16]](#footnote-16) The second UK class (Class 2) can be characterized as *“program rejecters.”* These respondents are indifferent to all program attributes (including virtually all reductions regardless of location) and prefer the status quo.[[17]](#footnote-17) The third and final class (Class 3) might be described as *“home reducers and international non-reducers.”* These respondents have significant and positive marginal utility for all home beach plastic reduction levels and some (10% and 20%) domestic coastal reductions, but significantly dislike (or are indifferent to) reductions in international waters, depending on the reduction (at the 10% significance level).[[18]](#footnote-18)

Parallel LCM results for the US version presented in Table 5. About half of respondents are estimated to be in Class 1 (50%), with a further 26% in Class 2 and 24% belonging to Class 3.

Table 5- US Version Covariate LCM Output

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  | Class 1 | | Class 2 | | Class 3 | |
| Attributes | Coef. | s.e. | Coef. | s.e. | Coef. | s.e. |
| beach\_25 | 0.376\*\*\* | 0.066 | 0.636 | 1.649 | 0.290\*\* | 0.130 |
| beach\_50 | 0.624\*\*\* | 0.067 | -0.158 | 1.496 | 0.350\*\*\* | 0.136 |
| beach\_75 | 0.872\*\*\* | 0.064 | -1.857 | 2.105 | 0.472\*\*\* | 0.133 |
| beach\_90 | 0.794\*\*\* | 0.072 | -2.249 | 1.478 | 0.690\*\*\* | 0.135 |
| coastal\_5 | -0.034 | 0.069 | -0.022 | 1.227 | 0.646\*\*\* | 0.134 |
| coastal\_10 | 0.237\*\*\* | 0.069 | 2.236 | 1.480 | 0.149 | 0.135 |
| coastal\_15 | 0.350\*\*\* | 0.066 | -4.330 | 2.296 | 0.387\*\*\* | 0.129 |
| coastal\_20 | 0.352\*\*\* | 0.072 | 0.377\* | 1.712 | 0.194 | 0.130 |
| intl\_3 | 0.085 | 0.064 | -2.617 | 1.534 | 0.270\*\* | 0.128 |
| intl\_5 | 0.112\* | 0.067 | -3.102 | 2.023 | -0.175 | 0.129 |
| intl\_10 | 0.216\*\*\* | 0.072 | -0.130 | 1.595 | -0.160 | 0.127 |
| intl\_15 | 0.212\*\*\* | 0.060 | -2.810 | 2.390 | 0.048 | 0.127 |
| foreign\_5 | -0.027 | 0.074 | -1.014 | 1.327 | -0.019 | 0.121 |
| foreign\_10 | 0.141\*\* | 0.063 | -1.602 | 1.398 | -0.357\*\*\* | 0.115 |
| foreign\_15 | 0.085 | 0.066 | 1.699 | 1.725 | -0.099 | 0.123 |
| foreign\_20 | 0.022 | 0.065 | -1.891 | 2.167 | -0.151 | 0.130 |
| csplit\_50home | 0.256\*\*\* | 0.050 | -0.275 | 1.032 | 0.299\*\*\* | 0.108 |
| csplit\_75home | 0.114\*\* | 0.056 | -1.601 | 1.181 | -0.169\* | 0.094 |
| Cost ($/yeah/HH) | -0.004\*\*\* | 0.000 | -0.160\*\*\* | 0.050 | -0.020\*\*\* | 0.001 |
| ASC | -2.267\*\*\* | 0.123 | -7.341\*\*\* | 3.295 | -1.183\*\*\* | 0.186 |
| Class Membership Model | | | | | | |
| Class Probability | 0.50 | | 0.26 |  | 0.24 |  |
| Intercept | . | | 2.070\*\*\* | 0.654 | 1.593\*\* | 0.648 |
| pay\_enough | . | | 1.015\*\*\* | 0.092 | 0.424\*\*\* | 0.081 |
| not\_my\_respon | . | | 0.659\*\*\* | 0.074 | 0.075 | 0.066 |
| important\_reduce | . | | -0.915\*\*\* | 0.104 | -0.647\*\*\* | 0.104 |
| policy\_con | . | | -0.467\*\*\* | 0.076 | -0.145\* | 0.075 |
| payment\_con | . | | -0.571\*\*\* | 0.087 | -0.288\*\*\* | 0.085 |
| impact\_con | . | | -0.638\*\*\* | 0.085 | -0.158\* | 0.088 |
| age | . | | 0.029\*\*\* | 0.005 | 0.025\*\*\* | 0.005 |
| income | . | | -0.747\*\*\* | 0.129 | -0.463\*\*\* | 0.119 |
| female | . | | 0.411\*\*\* | 0.130 | 0.202 | 0.125 |
| Log Likelihood | -9485.143 | |  |  |  |  |
| BIC | 19601.234 | |  |  |  |  |
| Adj. R2 | 0.479 | |  |  |  |  |
| # of individuals | 2662 | |  |  |  |  |
| Note \* p<0.10, \*\* p<0.05, \*\*\* p<0.01 | | | | | | |

As in the UK, results for the US model show strong evidence of preference divergence across classes. However, the type of classes and preferences that emerge differ from those found in the UK. Again, we focus primarily on preferences for plastic reductions in different locations. The first US class (Class 1) can be characterized as *“everywhere reducers”*. These respondents have significant and positive marginal utility for plastic reduction in all beach levels, most coastal and international levels (except the lowest levels for each), and some foreign reductions.[[19]](#footnote-19) Class 2 respondents, in contrast, might be described as *“any program supporters”* respondents have insignificant marginal utility for all program attributes (including plastic reductions in nearly all locations), with the exception of negative and significant marginal utility for the cost attribute and the status quo.[[20]](#footnote-20)

Finally, Class 3 respondents in the US can be characterized as *“home reducers and foreign non-reducers”*. These respondents have positive and significant marginal utility for all beach plastic reductions, and some coastal reductions. They exhibit positive and significant marginal utility for the smallest plastic reduction amount in international waters (3%) but are indifferent over higher (>3%) international reduction amounts. This class exhibit negative and significant marginal utility for 10% foreign waters reductions, suggesting a preference for home-based policies.[[21]](#footnote-21)

* 1. **Willingness to Pay and the Location of Marine Plastic Reductions**

Although interpretation of raw marginal utilities (i.e., preference-space parameter estimates) provides insight into whether and how program attributes influenced responses across the two samples considered independently, these estimates cannot be directly compared across samples due to confounding with the logit scale parameter. Hence, to allow meaningful comparison of results across the UK and US versions, e present parallel marginal WTP estimates, which are not confounded by scale. These marginal WTP estimates are represented graphically in Figure 3, and the underlying numerical estimates are presented in Appendix C. All non-cost attribute levels that were insignificant in Tables 4 and 5 are plotted as $0 marginal WTP. Results for Class 2 are omitted, as parameter estimates for this class produce no significant welfare estimates in either model (and may reflect protest responses as discussed below). We therefore limit our discussion to only classes 1 and 3.[[22]](#footnote-22)

When interpreting these welfare estimates, we take a “weight-of-evidence” approach. As noted above, we allow WTP to vary freely over different discrete levels for each attribute (e.g., 25%, 50%, 75% and 90% for beach plastic). We do not apply functional forms to the preference function that impose monotonic preferences (e.g., assuming preferences are linear in beach plastic improvements). Hence, for some attributes, we find non-monotonic preference patterns or significant WTP for some attribute levels and not others. Considering this, we distinguish attributes for which there is clear and robust evidence of positive WTP from those for which evidence of positive welfare is mixed or sporadic.

Figure 3- Marginal mean WTP for UK and US Versions ($/year/household) Black whiskers indicate 95% confidence intervals as simulated using the procedure suggested by Krinsky & Robb (1986).

Chart, box and whisker chart

Description automatically generated

As shown in Figure 3, there are differences across the two countries in terms of marginal WTP for marine plastic pollution reduction in different locations. Nonetheless, robust and consistent welfare patterns emerge across US and UK respondents. Across both samples, Class 1 (the dominant class) exhibits higher marginal WTP while Class 3 exhibits lower marginal WTP. This is not surprising given that in both sample versions, respondents in Class 3 were likely to have lower income levels, believe they already pay enough in taxes, and have lower levels of beliefs regarding the importance of reducing marine plastic.

The dominant finding is that domestic improvements are highly valued in both the US and UK, with the most immediate and visible type of reductions (to beach plastic) associated with the highest WTP. The largest marginal WTP across both classes and versions is for *home beach* plastic reduction, followed by *home coastal* plastic reduction. For the US, Class 1 exhibits a nonlinear trend for home beach plastic reduction, where marginal WTP is increasing then begins to decline slightly for plastic reduction levels beyond 75%. For Class 3 in both versions, results suggest monotonically increasing marginal WTP for beach plastic reduction, but much smaller in magnitude than parallel estimates for Class 1. Both cases provide evidence for positive marginal WTP for domestic beach plastic reductions.

Similar nonlinear trends are found for coastal plastic reductions, among Class 1 respondents in both samples. Marginal WTP is increasing until 15% reduction, at which point marginal WTP stabilizes (for the US) or begins to decrease (for the UK). However, these patterns are less intuitive for Class 3 respondents over different reduction levels, particularly in the US. Class 3 US respondents exhibit the highest marginal WTP (~$33) for the lowest level of coastal plastic reduction, falling to $20 for a 15% reduction. For UK Class 3 respondents, the marginal WTP for coastal plastic reduction is increasing between significant levels, ranging between $5 to $14.

Despite these variations, results provide robust evidence that domestic plastic reductions are valued by most survey respondents (Class 1 and 3), across both samples. Beach plastic reductions—those most likely to be experienced directly by respondents and potentially affecting use values—are associated with larger (and better behaved) welfare estimates than parallel reductions in coastal waters. This suggests the greater salience of beach plastic reductions to most respondents.

Inconsistent results are found for foreign (coastal and beach plastic) reductions. In the UK sample, only Class 3 exhibited positive WTP for any level of improvement in foreign waters, ($8 marginal WTP for 15% reduction only). In the US, we find a similar results: WTP is significant for only a single reduction level for Class 1 ($32 for a 10% reduction). In contrast, there is evidence of *negative* marginal WTP for US Class 3 respondents—a subsidy of $18 would be required for them to accept a 10% reduction in foreign plastic pollution as part of a bilateral IEA.

Taken together, these results provide *no convincing evidence of domestic support or welfare gains resulting from foreign reductions in coastal plastic*. These findings tend to support the standard textbook assumption that improvements in foreign waters have no value to the domestic public. Indeed, for the 16 marginal welfare estimates that are evaluated for foreign improvements (US and UK combined), one would expect that between 1-2 of these estimates might appear to be significant merely due to Type I error at a significance level of p=0.10 (false positives), even if there is no true effect (e.g., a true non-zero marginal WTP for foreign improvements). Viewed from this perspective, our results provide minimal evidence of any true positive WTP for marine plastic reductions in foreign territorial waters.

The assumption of zero value for non-domestic improvements, however, may not hold for changes to *international* waters—those outside the territorial waters of any nation. Both the UK and US Class 1 have significant and positive marginal WTP for international plastic reduction. Marginal WTP ranges from $22-24 for UK respondents and $48-49 for US individuals. Results are mixed for Class 3. In the US, these respondents have a slightly positive marginal WTP for the lowest level of plastic reduction ($13), while Class 3 in the UK exhibit a very small and negative marginal WTP for 15% international reduction (-$8).[[23]](#footnote-23) Taken together, these results suggest the importance of distinguishing foreign from international improvements when considering welfare gains from marine plastic reductions—with only the latter showing reliable weight-of-evidence for positive WTP.

* 1. **Voting Simulation Analysis**

Further insight into the salience of these results may be obtained by considering implications for aggregate voting support across the two samples. Information of this type (i.e., on predicted public support for different types of programs) may be more directly relevant than parallel information on welfare estimates from the perspective of IEA ratification. We therefore simulate public support for a bilateral agreement between the UK and US based on different scenarios of international and foreign pollution reductions.

To implement the simulation, we predict the choice probabilities of a set of specific marine plastic reduction programs at different individual cost levels against a ‘do-nothing’ alternative. This illustrates the predicted outcome of a binary referendum on such a policy proposal. For both the UK and US voting simulations, we consider a proposal that has 25% domestic beach plastic reduction, 10% domestic coastal plastic reduction and a proposed 50-50 cost share between the two countries. We then vary the level of foreign and international pollution reduction amounts and predict choice probabilities for each of these specifications over a ‘do-nothing’ alternative. We use the parameters from the LCM, and weight the probability of each class voting yes to a program by its respective class size. We consider only international and foreign reduction levels that have at least one class with significant marginal utility for that reduction level. This results in 6 possible programs in the UK and 8 possible programs in the US. In Figure 4, the solid green lines represent programs with 0% foreign reductions and increasing international reductions (light green to dark green). Similarly, the dashed purple lines represent the programs with high foreign reductions and increasing international reductions (from light purple to dark purple).

Figure 4- Voting SimulationsChart

Description automatically generatedChart, histogram

Description automatically generated

With no international or foreign reductions in prospect, the tax price that would result in a 50% probability that the referendum passes is $135 for the UK and $160 for the US. For the UK, results suggest that foreign plastic reductions have little impact on this probability (or the probability of support at any other cost)—higher levels of foreign plastic pollution reduction only result in higher probability of passing if they are coupled with positive levels of international reductions. One can view the same invariance of voting to foreign plastic reductions from other perspectives. As an example, as program costs exceed $150, a program with high international reductions has almost the same probability of passing regardless of whether any foreign reduction is offered.

In the US, most programs which provide foreign reductions have a lower probability of passing if offered at a cost below $200. That is, the provision of foreign reductions *reduces* the likelihood of passage at low costs. At the 50% probability mark, a program which provides neither international nor foreign reductions, is equally as likely to pass as a program that reduces international plastic as much as possible. Below $200, the least likely program to pass offers only foreign reductions; however, as a program begins to increase its international reductions, the probability of passing increases towards the 50% line. At higher costs per household (> $200), programs with the largest likelihood of passing (albeit below the 50% line) provide both international and foreign reductions.

Although the illustrated effects on predicted voting support are not large (typically from 1% - 2%), close to the social indifference point they could make a difference between whether an IEA obtains majority support. International plastic reductions tend to increase voting support, whereas foreign reductions have minimal or sometimes negative impacts. Moreover, and perhaps most interesting, the most favored policy option surrounding the indifference point (50% vote) differs between the UK and US samples. In the UK, the most highly favored program around the indifference point offers reductions in both international and foreign plastics. In the US, in contrast, the most highly favored program around the indifference point includes positive international reductions and no foreign reductions. Results of this type highlight the troubling possibility that the inclusion of greater foreign pollution reductions in an IEA, ceteris paribus, could actually reduce the probability of home-country approval on the margin.

1. **Discussion**

Voter preferences for pollution reductions in different geographic locations are likely to be important in determining political support for IEAs on marine plastics pollution. Results of the DCE suggest that on average, individuals in our two case study countries care about reducing marine plastic, but their degree of support depends on where improvements occur. Domestic reductions in pollution are most highly valued. We also find strong evidence that “one size does not fit all” when considering values for marine plastics reduction. For example, some individuals are willing to pay for pollution control which does not provide any obvious increases in use-value (reductions in pollution in international waters which are distant from the home coastline), while others have zero identifiable value for the same reductions.

Our work aligns with one of the key theoretical findings from Kolstad (2014) relative to coalition formation for IEAs. He finds that when countries have “other-regarding” preferences, agents are more likely to cooperate. We find that many individuals in our sample have other-regarding preferences, but primarily for pollution in *international waters* rather than pollution in the domestic waters of other countries. Hence, at least for some types of respondents in each country, improvements to pollution in international waters might provide welfare effects that encourage the emergence of self-enforcing IEAs, beyond motivations provided by the value of domestic plastic reductions.[[24]](#footnote-24) A corollary conclusion is that self-enforcing IAEs are more likely to emerge (and be stable) when they explicitly target improvements to international waters, beyond improvements to the territorial waters of each signatory nation.

One explanation for our choice modeling results is that they reflect a tendency among the public to distinguish between improvements to international and foreign waters. One might speculate, for example, that the domestic public might feel greater responsibility for international waters (and hence be willing to pay more for improvements there), because those waters are not viewed as being under the protection or authority of any nation. In contrast, the public might view foreign water improvements as being the responsibility of other nations, and not wish to support those improvements via bilateral or multilateral IEAs.

However, another possibility—and not one explored directly here—is that these preferences could reflect a type of macro-level distance decay (cf. Hanley et al. 2003; Bateman et al. 2006). Under this possibility, respondents are more willing to pay for improvements that are categorically closer in distance, with beaches, domestic coastal waters, international waters and foreign waters being successively more distant from residents of both the US and UK, on average. Although the present study was not designed to disentangle the possible reasons why international plastic reductions might be more highly valued than foreign pollution reductions, this is an intriguing area for future work.

Regardless of the underlying preference motivations, results of the present study provide evidence that public support and WTP for bilateral IEAs (here for marine plastics reductions) cannot be accurately predicted solely in terms of “domestic versus foreign” effects. Results show different WTP for different types of domestic impacts (e.g., beach versus coastal), along with different WTP for international versus foreign effects. These preference patterns also vary across countries (here the US versus UK). Findings of this type, if found to hold more broadly, suggest that extensions to the theory of IEA coalitions may be warranted—and specifically those that accommodate the type of preference heterogeneity identified here for marine plastics.

As a final point, we emphasize that a nontrivial group of individuals exist in each sample made choices that did not appear to be consistent with fully compensatory neoclassical welfare optimization over the presented program attributes. In the UK sample, these respondents (Class 2) prefer the status quo over any proposed reduction intervention and are insensitive to variations in program costs. Given the fact that these respondents also score comparably high on protest attitudes and low on consequentiality perceptions, it seems likely that these may be protest responses (Meyerhoff & Liebe, 2008). In the US sample, a group of respondents support all types of pollution reduction programs but are completely indifferent to where the reductions occur and by how much (classic yea-saying). An advantage of LCM is that it can enable responses of this type to be identified and isolated, thereby distinguishing apparent protest responses from responses that are at least seemingly more consistent with neoclassical welfare estimation. These models also allow the estimation of preferences and WTP for the other classes (here Class 1 and 3), unaffected by the apparent protest behavior of the problematic class (here, Class 2).

1. **Conclusion**

Transboundary pollution control requires abatement efforts from multiple countries. However, this also gives rise to free rider concerns that occur when considering the public good nature of international pollution control. Understanding individuals’ willingness to support different cooperative programs is key in designing effective pollution control when countries need to cooperate. Our paper uses a stated preference approach to investigate what types of program domestic voters would support, and therefore what type of IEA they would back their political representatives to sign. This is the first study to explore WTP for transboundary pollution control which explicitly controls for where and by how much pollution damages are reduced between domestic, foreign and international territories.

Results for our case study of marine plastic show that respondents in both the US and UK were willing to pay the most to reduce plastic at home, on domestic beaches (particularly) and to a lesser extent in domestic coastal waters. This is true for respondents with both high and low absolute WTP values. However, as programs begin to introduce reductions in pollution farther afield, preferences are less consistent. There is evidence that international pollution reductions are valued by the majority of UK and US respondents. However, results provide little unequivocal evidence of value for reductions of pollution in foreign territories.

Multiple caveats should be considered when interpreting our empirical results. For example, our analysis does not consider the overall emissions that each country contributes, rather it considers current (estimated) total accumulation within a country’s boundaries. However, through our focus group testing, we found that the proportion of total pollution that can be attributed to one’s home country seemed like a negligible consideration of individuals; what mattered was how much plastic was removed, not the geographic origin of this plastic waste. People seemed well-informed about the extent to which plastic wastes mix and travel across the ocean.

As noted above, the study also limited the context of our choices to a potential IEA between two signatory countries. Due to the transboundary nature of the pollutants in question, other countries will receive positive externalities from any coalition and introduced reduction measures. Future research would be necessary to explore related issues such as (1) preferences for members of the coalition towards reductions in non-signatory nations and (2) whether increasing the number of signatories impacts an individual’s preferences for altruistic behavior. van der Pol, Weikard, & van Ierland (2012) find that the stability of a coalition increases with impartial altruism (i.e. it becomes more favorable to be in the coalition than out) and community altruism (i.e. when members of the coalition feel stronger levels of altruism towards other signatories than towards non-signatories).

Finally, the results presented here should be interpreted within the context of our present case study and sample. Our results reflect the realized sample of survey responses from a set of opt-in internet panels. Although sample demographics appear to be reasonably representative of the sampled areas (Table 1), stated-preference surveys rarely produce samples that are perfectly representative of the target population over both observable and unobservable dimensions (Johnston & Abdulrahman, 2017). In addition, although multiple steps were taken to produce a choice experiment with high consequentiality and content validity, three-alternative choice experiments (the most common structure in environmental economics applications) cannot be considered strictly incentive compatible unless (among other requirements) respondents have uniform priors concerning the preferences of other decision makers (Collins & Vossler, 2009). The presented results should be interpreted accordingly.

Acknowledgements:

This work was conducted in the project “The Economics of Marine Plastic Pollution – What are the Benefits of International Cooperation” funded by the UK’s Economic and Social Research Council (ES/S002448/1).

**References:**

Abate, T. G., Börger, T., Aanesen, M., Falk-Andersson, J., Wyles, K. J., & Beaumont, N. (2020). Valuation of marine plastic pollution in the European Arctic: Applying an integrated choice and latent variable model to contingent valuation. *Ecological Economics*, *169*, 106521. https://doi.org/10.1016/j.ecolecon.2019.106521

Ahtiainen, H., Artell, J., Czajkowski, M., Hasler, B., Hasselström, L., Huhtala, A., Meyerhoff, J., Smart, J. C. R., Söderqvist, T., Alemu, M. H., Angeli, D., Dahlbo, K., Fleming-Lehtinen, V., Hyytiäinen, K., Karlõševa, A., Khaleeva, Y., Maar, M., Martinsen, L., Nõmmann, T., … Semeniene, D. (2014). Benefits of meeting nutrient reduction targets for the Baltic Sea–a contingent valuation study in the nine coastal states. *Journal of Environmental Economics and Policy*, *3*(3), 278–305. https://doi.org/10.1080/21606544.2014.901923

Almroth, B. C., & Eggert, H. (2019). Marine Plastic Pollution: Sources, Impacts, and Policy Issues. *Review of Environmental Economics and Policy*, *2016*, 1–11. https://doi.org/10.1093/reep/rez012

Anthoff, D., & Tol, R. S. J. (2010). On international equity weights and national decision making on climate change. *Journal of Environmental Economics and Management*, *60*(1), 14–20. https://doi.org/10.1016/j.jeem.2010.04.002

Bakhtiari, F., Jacobsen, J. B., Thorsen, B. J., Lundhede, T. H., Strange, N., & Boman, M. (2018). Disentangling Distance and Country Effects on the Value of Conservation across National Borders. *Ecological Economics*, *147*(January), 11–20. https://doi.org/10.1016/j.ecolecon.2017.12.019

Barnes, D. K. A., Galgani, F., Thompson, R. C., & Barlaz, M. (2009). Accumulation and fragmentation of plastic debris in global environments. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *364*(1526), 1985–1998. https://doi.org/10.1098/rstb.2008.0205

Barrett, S. (1994). Self-Enforing International Environmental Agreements. *Oxford Economic Papers*, *46*, 878–894.

Barrett, S. (2003). *Environment and Statecraft: The Strategy of Environmental Treaty-Making*. Oxford University Press. https://doi.org/10.1093/0199286094.003.0001

Bateman, I. J., Cooper, P., Georgiou, S., Navrud, S., Poe, G. L., Ready, R. C., Riera, P., Ryan, M., & Vossler, C. A. (2005). Economic valuation of policies for managing acidity in remote mountain lakes: Examining validity through scope sensitivity testing. *Aquatic Sciences*, *67*(3), 274–291. https://doi.org/10.1007/s00027-004-0744-3

Beaumont, N. J., Aanesen, M., Austen, M. C., Börger, T., Clark, J. R., Cole, M., Hooper, T., Lindeque, P. K., Pascoe, C., & Wyles, K. J. (2019). Global ecological, social and economic impacts of marine plastic. *Marine Pollution Bulletin*, *142*, 189–195. https://doi.org/10.1016/j.marpolbul.2019.03.022

Boxall, P. C., & Adamowicz, W. L. (2002). Understanding Heterogeneous Preferences in Random Utility Models: A Latent Class Approach. *Environmental and Resource Economics*, *23*, 421–446. https://link.springer.com/content/pdf/10.1023%2FA%3A1021351721619.pdf

Carlsson, F., Kataria, M., Krupnick, A., Lampi, E., Löfgren, Å., Qin, P., Chung, S., & Sterner, T. (2012). Paying for Mitigation: A multiple country study. *Land Economics*, *88*(2), 326–340. https://doi.org/10.3368/le.88.2.326

Carlsson, F., Kataria, M., Lampi, E., Löfgren, Å., & Sterner, T. (2011). Is fairness blind?-The effect of framing on preferences for effort-sharing rules. *Ecological Economics*, *70*(8), 1529–1535. https://doi.org/10.1016/j.ecolecon.2011.03.015

ChoiceMetrics. (2018). *Ngene 1.2 User Manual and Reference Guide*. *Australia*. www.choice-metrics.com

Collins, J. P., & Vossler, C. A. (2009). *Incentive compatibility tests of choice experiment value elicitation questions*. https://doi.org/10.1016/j.jeem.2009.04.004

Cózar, A., Echevarría, F., Ignacio González-Gordillo, J., Irigoien, X., Úbeda, B., Hernández-León, S., Palma, Á. T., Navarro, S., García-De-Lomas, J., Ruiz, A., Fernández-De-Puelles, M. L., & Duarte, C. M. (2014). Plastic debris in the open ocean. *PNAS*. https://doi.org/10.1073/pnas.1314705111

Czajkowski, M., Ahtiainen, H., Artell, J., Budziński, W., Hasler, B., Hasselström, L., Meyerhoff, J., Nõmmann, T., Semeniene, D., Söderqvist, T., Tuhkanen, H., Lankia, T., Vanags, A., Zandersen, M., Żylicz, T., & Hanley, N. (2015). Valuing the commons: An international study on the recreational benefits of the Baltic Sea. *Journal of Environmental Management*, *156*, 209–217. https://doi.org/10.1016/j.jenvman.2015.03.038

Dallimer, M., Bredahl, J., Lundhede, T. H., Takkis, K., Giergiczny, M., & Thorsen, B. J. (2015). Patriotic values for public goods: Transnational trade-offs for biodiversity and ecosystem services? *BioScience*, *65*(1), 33–42. https://doi.org/10.1093/biosci/biu187

Dameron, O. J., Parke, M., Albins, M. A., & Brainard, R. (2007). Marine debris accumulation in the Northwestern Hawaiian Islands: An examination of rates and processes. *Marine Pollution Bulletin*, *54*(4), 423–433. https://doi.org/10.1016/j.marpolbul.2006.11.019

Danley, B., Sandorf, E. D., & Campbell, D. (2021). Putting your best fish forward: Investigating distance decay and relative preferences for fish conservation. *Journal of Environmental Economics and Management*, *108*(May 2019), 102475. https://doi.org/10.1016/j.jeem.2021.102475

de Frutos, J., & Martín-Herrán, G. (2019). Spatial effects and strategic behavior in a multiregional transboundary pollution dynamic game. *Journal of Environmental Economics and Management*, *97*, 182–207. https://doi.org/10.1016/j.jeem.2017.08.001

Diffendorfer, J. E., Loomis, J. B., Ries, L., Oberhauser, K., Lopez-Hoffman, L., Semmens, D., Semmens, B., Butterfield, B., Bagstad, K., Goldstein, J., Wiederholt, R., Mattsson, B., & Thogmartin, W. E. (2014). National valuation of monarch butterflies indicates an untapped potential for incentive-based conservation. *Conservation Letters*, *7*(3), 253–262. https://doi.org/10.1111/conl.12065

Du, X., Jin, X., Zucker, N., Kennedy, R., & Urpelainen, J. (2020). Transboundary air pollution from coal-fired power generation. *Journal of Environmental Management*, *270*(June). https://doi.org/10.1016/j.jenvman.2020.110862

Eriksen, M., Lebreton, L. C. M., Carson, H. S., Thiel, M., Moore, C. J., Borerro, J. C., Galgani, F., Ryan, P. G., & Reisser, J. (2014). Plastic Pollution in the World’s Oceans: More than 5 Trillion Plastic Pieces Weighing over 250,000 Tons Afloat at Sea. *PLoS ONE*, *9*(12), 1–15. https://doi.org/10.1371/journal.pone.0111913

Ferrini, S., & Scarpa, R. (2007). Designs with a priori information for nonmarket valuation with choice experiments: A Monte Carlo study. *Journal of Environmental Economics and Management*, *53*(3), 342–363. https://doi.org/10.1016/j.jeem.2006.10.007

Finus, M. (2008). Game Theoretic Research on the Design of International Environmental Agreements: Insights, Critical Remarks, and Future Challenges \*. *International Review of Environmental and Resource Economics*, *2*, 29–67. https://doi.org/10.1561/101.00000011

Greene, W. H., & Hensher, D. A. (2003). A latent class model for discrete choice analysis: Contrasts with mixed logit. *Transportation Research Part B: Methodological*, *37*(8), 681–698. https://doi.org/10.1016/S0191-2615(02)00046-2

Haefele, M. A., Loomis, J. B., Lien, A. M., Dubovsky, J. A., Merideth, R. W., Bagstad, K. J., Huang, T. K., Mattsson, B. J., Semmens, D. J., Thogmartin, W. E., Wiederholt, R., Diffendorfer, J. E., & López-Hoffman, L. (2019). Multi-country Willingness to Pay for Transborder Migratory Species Conservation: A Case Study of Northern Pintails. *Ecological Economics*, *157*(April 2018), 321–331. https://doi.org/10.1016/j.ecolecon.2018.11.024

Herriges, J., Kling, C., Liu, C.-C., & Tobias, J. (2010). What are the consequences of consequentiality? *Journal of Environmental Economics & Management*, *59*, 67–81. https://doi.org/10.1016/j.jeem.2009.03.004

Hoel, M., & Schneider, K. (1997). Incentives to Participate in an International Environmental Agreement. *Environmental and Resource Economics*, *9*, 153–170.

Jambeck, J., Geyer, R., Wilcox, C., Siegler, T., Perryman, M., Andrady, A., Narayan, R., & Law, K. L. (2015). Plastic waste inputs from land into the ocean. *Marine Pollution*, *347*. https://doi.org/10.1126/science.1260352

Jin, J., Indab, A., Nabangchang, O., Thuy, T. D., Harder, D., & Subade, R. F. (2010). Valuing marine turtle conservation: A cross-country study in Asian cities. *Ecological Economics*, *69*(10), 2020–2026. https://doi.org/10.1016/j.ecolecon.2010.05.018

Johnston, R. J., & Abdulrahman, A. S. (2017). Systematic non-response in discrete choice experiments: implications for the valuation of climate risk reductions. *Journal of Environmental Economics and Policy*, *6*(3), 246–267. https://doi.org/10.1080/21606544.2017.1284695

Johnston, R. J., Schultz, E. T., Segerson, K., Besedin, E. Y., & Ramachandran, M. (2012). Enhancing the content validity of stated preference valuation: The structure and function of ecological indicators. *Land Economics*, *88*(1), 102–120. https://doi.org/10.3368/le.88.1.102

Kirk, E. A. (2018). *Marine plastics: Fragmentation, effectiveness and legitimacy in international lawmaking*. https://doi.org/10.1111/reel.12261

Kolstad, C. D. (2014). International Environmental Agreements among Heterogeneous Countries with Social Preferences. In *NBER Working Paper Series* (Issue Working Paper 20204). http://search.proquest.com/docview/1687917293?accountid=27468%0Ahttp://sfx.nelliportaali.fi/nelli32b?url\_ver=Z39.88-2004&rft\_val\_fmt=info:ofi/fmt:kev:mtx:journal&genre=preprint&sid=ProQ:ProQ%3Aabiglobal&atitle=International+Environmental+Agreements+among+

Krinsky, I., & Robb, a L. (1986). On Approximating the Statistical Properties of Elasticities. *The Review of Economics and Statistics*, *68*(4), 715–719.

Lau, W. W. Y., Shiran, Y., Bailey, R. M., Cook, E., Stuchtey, M. R., Koskella, J., Velis, C. A., Godfrey, L., Boucher, J., Murphy, M. B., Thompson, R. C., Jankowska, E., Castillo, A. C., Pilditch, T. D., Dixon, B., Koerselman, L., Kosior, E., Favoino, E., Gutberlet, J., … Palardy, J. E. (2020). Evaluating scenarios toward zero plastic pollution. *Science*, *369*(6509), 1455–1461. https://doi.org/10.1126/SCIENCE.ABA9475

Law, K. L., Morét-Ferguson, S., Maximenko, N. A., Proskurowski, G., Peacock, E. E., Hafner, J., & Reddy, C. M. (2010). Plastic accumulation in the North Atlantic subtropical gyre. *Science*, *329*(5996), 1185–1188. https://doi.org/10.1126/science.1192321

Lebreton, L. C. M., Greer, S. D., & Borrero, J. C. (2012). Numerical modelling of floating debris in the world’s oceans. *Marine Pollution Bulletin*, *64*(3), 653–661. https://doi.org/10.1016/j.marpolbul.2011.10.027

Lebreton, L., Egger, M., & Slat, B. (2019). A global mass budget for positively buoyant macroplastic debris in the ocean. *Scientific Reports*, *9*.

Lee, J. J., & Cameron, T. A. (2008). Popular Support for Climate Change Mitigation: Evidence from a General Population Mail Survey. *Environ Resource Econ*, *41*, 223–248. https://doi.org/10.1007/s10640-007-9189-1

Leggett, C. G., Scherer, N., Haab, T. C., Bailey, R., Landrum, J. P., & Domanski, A. (2018). Assessing the Economic Benefits of Reductions in Marine Debris at Southern California Beaches : A Random Utility Travel Cost Model. *Marine Resource Economics*, *33*(2).

Livingston, M. L. (1989). Transboundary Environmental Degradation: Market Failure, Power, and Instrumental Justice. *Journal of Economic Issues*, *23*(1), 79–91. https://doi.org/10.1080/00213624.1989.11504868

Loureiro, M. L., & Loomis, J. B. (2013). International Public Preferences and Provision of Public Goods: Assessment of Passive Use Values in Large Oil Spills. *Environmental and Resource Economics*, *56*(4), 521–534. https://doi.org/10.1007/s10640-012-9556-4

Maler, K.-G. (1989). The Acid Rain Game. In H. Folmer & E. van Ierland (Eds.), *Valuation Methods and Policy Making in Environmental Economics*. Elsevier.

McIlgorm, A., Campbell, H. F., & Rule, M. J. (2011). The economic cost and control of marine debris damage in the Asia-Pacific region. *Ocean and Coastal Management*, *54*(9), 643–651. https://doi.org/10.1016/j.ocecoaman.2011.05.007

Meginnis, K., Domanski, A., & Toledo-Gallegos, V. M. (2022). Is it up to business, governments, or individuals to tackle the marine plastic problem? A hybrid mixed logit approach. *Marine Pollution Bulletin*, *174*. https://doi.org/10.1016/J.MARPOLBUL.2021.113169

Meijer, L. J. J., Van Emmerik, T., Van Der Ent, R., Schmidt, C., & Lebreton, L. (2021). More than 1000 rivers account for 80% of global riverine plastic emissions into the ocean. *Sci. Adv*, *7*. http://advances.sciencemag.org/

Meyerhoff, J., & Liebe, U. (2008). Do protest responses to a contingent valuation question and a choice experiment differ? *Environmental and Resource Economics*, *39*(4), 433–446. https://doi.org/10.1007/s10640-007-9134-3

Missfeldt, F. (1999). GAME-THEORETIC MODELLING OF TRANSBOUNDARY POLLUTION. *Journal of Economic Surveys1*, *13*(3).

Mouat, J., Lozano, R. L., & Bateson, H. (2010). *Economic Impacts of Marine Litter*. www.iStockphoto.com/matsou,

NOAA. (2019). *The Effects of Marine Debris on Beach Recreation and Regional Economies in Four Coastal Communities: A Regional Pilot Study Final Report*.

NOAA. (2020). *NOAA Marine Debris Program Accomplishments Report*. https://marinedebris.noaa.gov/report/accomplishments-report

Obeng, E. A., & Aguilar, F. X. (2021). Willingness-to-pay for restoration of water quality services across geo-political boundaries. *Current Research in Environmental Sustainability*, *3*, 100037. https://doi.org/10.1016/j.crsust.2021.100037

Ocean Conservancy. (2019). The Beach And Beyond. *The Beach and beyond - 2019 Report*, 1–30. https://oceanconservancy.org/wp-content/uploads/2019/09/Final-2019-ICC-Report.pdf

*PPP and exchange rates*. (2020). OECD.Stat. https://stats.oecd.org/index.aspx?DataSetCode=SNA\_Table4

Ressurreição, A., Gibbons, J., Kaiser, M., Dentinho, T. P., Zarzycki, T., Bentley, C., Austen, M., Burdon, D., Atkins, J., Santos, R. S., & Edwards-Jones, G. (2012). Different cultures, different values: The role of cultural variation in public’s WTP for marine species conservation. *Biological Conservation*, *145*(1), 148–159. https://doi.org/10.1016/j.biocon.2011.10.026

Rolfe, J., & Windle, J. (2012). Distance Decay Functions for Iconic Assets: Assessing National Values to Protect the Health of the Great Barrier Reef in Australia. *Environ Resource Econ*, *53*, 347–365. https://doi.org/10.1007/s10640-012-9565-3

Rose, J. M., & Bliemer, M. C. J. (2009). *Transport Reviews Constructing Efficient Stated Choice Experimental Designs*. https://doi.org/10.1080/01441640902827623

Rose, J. M., Bliemer, M. C. J., Hensher, D. A., & Collins, A. T. (2008). Designing efficient stated choice experiments in the presence of reference alternatives. *Transportation Research Part B: Methodological*, *42*, 395–406. https://doi.org/10.1016/j.trb.2007.09.002

Sawtooth. (2018). *Sawtooth Software SSI Web*. http://www.sawtoothsoftware.com/

Scarpa, R., & Rose, J. M. (2008). Design efficiency for non-market valuation with choice modelling: how to measure it, what to report and why\*. *Australian Journal of Agricultural and Resource Economics*, *52*(3), 253–282.

Scarpa, R., & Thiene, M. (2005). Destination choice models for rock climbing in the Northeastern Alps: A latent-class approach based on intensity of preferences. *Land Economics*, *81*(3), 426–444. https://doi.org/10.3368/le.81.3.426

Ščasný, M., Zvěřinová, I., Czajkowski, M., Kyselá, E., & Zagórska, K. (2017). Public acceptability of climate change mitigation policies: a discrete choice experiment. *Climate Policy*, *17*, S111–S130. https://doi.org/10.1080/14693062.2016.1248888

Siikamaki, J. V., Krupnick, A., Strand, J., & Vincent, J. R. (2019). International Willingness to Pay for the Protection of the Amazon Rainforest. *International Willingness to Pay for the Protection of the Amazon Rainforest*, *March*. https://doi.org/10.1596/1813-9450-8775

Siqueira, K. (2003). International externalities, strategic interaction, and domestic politics. *Journal of Environmental Economics and Management*, *45*(3), 674–691. https://doi.org/10.1016/S0095-0696(02)00023-2

Strand, J., Carson, R. T., Navrud, S., Ortiz-Bobea, A., & Vincent, J. R. (2017). Using the Delphi method to value protection of the Amazon rainforest. *Ecological Economics*, *131*, 475–484. https://doi.org/10.1016/j.ecolecon.2016.09.028

Svenningsen, L. S., & Thorsen, B. J. (2020). Preferences for Distributional Impacts of Climate Policy. *Environmental and Resource Economics*, *75*(1), 1–24. https://doi.org/10.1007/s10640-019-00386-z

Thevenon, F., & Carroll, C. (2015). Plastic debris in the ocean: the characterization of marine plastics and their environmental impacts, situation analysis report. In *Plastic debris in the ocean: the characterization of marine plastics and their environmental impacts, situation analysis report*. https://doi.org/10.2305/iucn.ch.2014.03.en

Train, K. (2009). Discrete Choice Methods with Simulation. In *Cambridge University Press*. Cambridge university press.

Tyllianakis, E., & Ferrini, S. (2021). *Personal attitudes and beliefs and willingness to pay to reduce marine plastic pollution in Indonesia*. https://doi.org/10.1016/j.marpolbul.2021.113120

UNEP. (2014). *Valuing Plastics: The Business Case for Measuring, Managing and Disclosing Plastic Use in the Consumer Goods Industry*. www.gpa.unep.org

Valasiuk, S., Czajkowski, M., Giergiczny, M., Żylicz, T., Veisten, K., Elbakidze, M., & Angelstam, P. (2017). Are bilateral conservation policies for the Białowieża forest unattainable? Analysis of stated preferences of Polish and Belarusian public. *Journal of Forest Economics*, *27*, 70–79. https://doi.org/10.1016/j.jfe.2017.03.001

van der Pol, T., Weikard, H. P., & van Ierland, E. (2012). Can altruism stabilise international climate agreements? *Ecological Economics*, *81*, 112–120. https://doi.org/10.1016/j.ecolecon.2012.06.011

Vince, J., & Hardesty, B. D. (2018). Governance solutions to the tragedy of the commons that marine plastics have become. *Frontiers in Marine Science*, *5*(JUN). https://doi.org/10.3389/fmars.2018.00214

Vogdrup-Schmidt, M., Abatayo, A. Lou, Shogren, J. F., Strange, N., & Thorsen, B. J. (2019). Factors Affecting Support for Transnational Conservation Targeting Migratory Species. *Ecological Economics*, *157*(October 2018), 156–164. https://doi.org/10.1016/j.ecolecon.2018.11.011

Vossler, C. A., & Watson, S. B. (2013). Understanding the consequences of consequentiality: Testing the validity of stated preferences in the field. *Journal of Economic Behavior and Organization*, *86*, 137–147. https://doi.org/10.1016/j.jebo.2012.12.007

**Appendix A – Status Quo Calculations**

Table A.1- Attribute Status Quo Levels

|  |  |  |
| --- | --- | --- |
| Attribute | UK Version | US Version |
| 1. Home Beach (items per 100m) | 85 items | 217 items |
| 1. Home Coastal (items per sq km) | 170 items | 435 items |
| 1. International (items per sq km) | 1807 items | 1807 items |
| 1. Foreign beach and coastal (items per sq km) | 2605 items | 1020 items |

The only attribute that has the same status quo level for both survey versions is: international water plastic. This value was taken from Eriksen et al. (2014) Table 1 which models total particles by count and weight for the World’s oceans. As we were only interested in macro and meso plastic (i.e. > 5mm) we consider there to be 7.3 x 1010 + 0.2 x 1010 pieces of macro and meso plastic, respectively. Given the North Atlantic is approximately 41,490,000 square kilometers, this equates to approximately 1,807 pieces of plastic per square kilometer.

For attributes 1, 2, and 4 we used information from Jambeck et al. (2015) supplementary materials (available at <https://jambeck.engr.uga.edu/landplasticinput>) that lists estimates for 192 countries. This was calculated for every country based on the population living within 50 kilometers of the coast. We converted that information into grams per kilometer of coastline and then used the midpoint estimate of 3.15% to calculate the amount of mismanaged waste that ends up in the ocean. Finally, to ease respondents’ understanding, we converted this into ‘items of plastic’ by assuming the average weight of a piece of plastic is 10grams (roughly the weight of an empty plastic bottle). This process is outlined in Table A.2.

Table A.2 Plastic Mismanaged Waste Calculations

|  |  |  |
| --- | --- | --- |
|  | UK Version | US version |
| Tones of mismanaged waste (W) | 67,549 | 275,424 |
| Kilometer of coastline\* (C) | 12429 | 19924 |
| Grams of mismanaged waste per km of coastline  = (W/C)\*1000000 | 5434792.552 | 13823755.11 |
| Percentage of mismanaged waste that goes into ocean (midpoint estimate) | 3.15 | 3.15 |
| Grams of mismanaged waste in ocean | 171195.96 | 435448.28 |
| Mismanaged items in ocean per sq km | 17119.596 | 43544.828 |
| \* Taken from https://www.citypopulation.de/en/world/bymap/Coastlines.html | | |

In order to calculate the different values in different bodies of water, we referred to Almroth and Eggert (2019), who estimated that from marine plastic pollution, 95% ends up on the sea floor, 5% ends up on beaches and 1% remains on ocean surfaces. These calculations are shown in Table A.3.

Table A.3 Items of Plastic Estimates for discrete marine environments

|  |  |  |
| --- | --- | --- |
|  | UK Version | US Version |
| Mismanaged items in ocean per sq km (Table A.2) | 17119.59 | 43544.82 |
| Beach estimate (items per km) | 855.95 | 2177.2 |
| Beach estimate (items per 100m) | 85.6 | 217.7 |
| Surface estimate (items per km) | 171.2 | 435.45 |
| Beach and surface estimate (items per km) | 1027.15 | 2612.65 |

**Appendix B- MNL results**

Table B.1 MNL Model Output

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | UK Version | | US Version | |
| Attributes | Coefficient | St. Err. | Coefficient | St. Err. |
| beach\_25 | 0.257\*\*\* | (0.055) | 0.297\*\*\* | (0.048) |
| beach\_50 | 0.407\*\*\* | (0.055) | 0.464\*\*\* | (0.048) |
| beach\_75 | 0.533\*\*\* | (0.054) | 0.602\*\*\* | (0.046) |
| beach\_90 | 0.595\*\*\* | (0.057) | 0.643\*\*\* | (0.049) |
| coastal\_5 | 0.150\*\*\* | (0.055) | 0.135\*\*\* | (0.047) |
| coastal\_10 | 0.132\*\* | (0.055) | 0.183\*\*\* | (0.047) |
| coastal\_15 | 0.320\*\*\* | (0.054) | 0.265\*\*\* | (0.046) |
| coastal\_20 | 0.257\*\*\* | (0.055) | 0.256\*\*\* | (0.047) |
| intl\_3 | 0.050 | (0.052) | 0.086 | (0.045) |
| intl\_5 | -0.048 | (0.054) | -0.048 | (0.046) |
| intl\_10 | -0.060 | (0.054) | -0.007 | (0.046) |
| intl\_15 | 0.065 | (0.051) | 0.111\*\* | (0.044) |
| foreign\_5 | 0.019 | (0.054) | -0.018 | (0.047) |
| foreign\_10 | -0.033 | (0.051) | -0.050 | (0.043) |
| foreign\_15 | 0.025 | (0.052) | 0.012 | (0.045) |
| foreign\_20 | -0.140\*\*\* | (0.053) | -0.070 | (0.045) |
| csplit\_50home | 0.108\*\*\* | (0.042) | 0.189\*\*\* | (0.036) |
| csplit\_75home | -0.215\*\*\* | (0.042) | 0.026 | (0.036) |
| ASC | -0.237\*\*\* | (0.073) | 0.158\*\* | (0.063) |
| Cost ($/HH/year) | -0.006\*\*\* | (0.000) | -0.0051\*\*\* | (0.000) |
| LL |  | -10252.360 |  | -13842.650 |
| BIC |  | 20711.180 |  | 27897.190 |
| N of indiv. |  | 2028 |  | 2662 |
| Note: \* <0.10, \*\*<0.05, \*\*\*<005 | | |  |  |

Table B.2 Marginal WTP MNL Model

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  | UK Version | | | US Version | | |
|  | Mean | LB | UB | Mean | LB | UB |
| beach\_25 | 40.33 | 22.73 | 57.93 | 58.27 | 39.12 | 77.42 |
| beach\_50 | 64.10 | 46.25 | 81.94 | 91.07 | 71.10 | 111.04 |
| beach\_75 | 84.00 | 66.68 | 101.33 | 118.14 | 98.95 | 137.33 |
| beach\_90 | 93.92 | 74.71 | 113.13 | 126.18 | 104.56 | 147.80 |
| coastal\_5 | 23.81 | 6.76 | 40.86 | 26.51 | 8.42 | 44.61 |
| coastal\_10 | 21.12 | 4.03 | 38.21 | 35.95 | 17.58 | 54.32 |
| coastal\_15 | 50.45 | 33.62 | 67.28 | 52.06 | 34.07 | 70.05 |
| coastal\_20 | 40.47 | 23.24 | 57.70 | 50.25 | 31.78 | 68.72 |
| intl\_3 |  |  |  |  |  |  |
| intl\_5 |  |  |  |  |  |  |
| intl\_10 |  |  |  |  |  |  |
| intl\_15 |  |  |  | 21.76 | 4.96 | 38.57 |
| foreign\_5 |  |  |  |  |  |  |
| foreign\_10 |  |  |  |  |  |  |
| foreign\_15 |  |  |  |  |  |  |
| foreign\_20 | -22.27 | -38.87 | -5.68 |  |  |  |
| csplit\_50home | 17.41 | 4.65 | 30.18 | 37.07 | 23.50 | 50.64 |
| csplit\_75home | -33.78 | -46.64 | -20.93 |  |  |  |
| none | -37.54 | -59.32 | -15.75 | 31.07 | 5.86 | 56.27 |
| Unit: USD | | | | | | |

**Appendix C**

Table C.1 Marginal WTP for LCM (depicted visually in Figure 3)

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  | UK Version | | | | | |
|  | Class 1 | | | Class 3 | | |
|  | Mean | LB | UB | Mean | LB | UB |
| beach\_25 | 41.72 | 21.56 | 62.60 | 15.91 | 8.04 | 24.03 |
| beach\_50 | 67.65 | 47.41 | 88.45 | 14.19 | 5.81 | 23.28 |
| beach\_75 | 91.96 | 72.81 | 111.36 | 20.04 | 11.59 | 29.09 |
| beach\_90 | 97.84 | 75.46 | 121.46 | 20.74 | 12.14 | 29.67 |
| coastal\_5 | 21.58 | 0.77 | 41.91 |  |  |  |
| coastal\_10 | 29.55 | 9.52 | 50.48 | 9.40 | 0.95 | 17.84 |
| coastal\_15 | 59.59 | 39.50 | 79.95 | 0.00 |  |  |
| coastal\_20 | 49.04 | 27.52 | 70.18 | 13.45 | 5.23 | 21.42 |
| intl\_3 |  |  |  |  |  |  |
| intl\_5 | 23.31 | 2.63 | 43.29 | -8.48 | -16.65 | 0.09 |
| intl\_10 |  |  |  |  |  |  |
| intl\_15 | 22.34 | 4.61 | 40.42 | -8.07 | -16.13 | -0.08 |
| foreign\_5 |  |  |  |  |  |  |
| foreign\_10 | |  |  |  |  |  |
| foreign\_15 | |  |  | 8.48 | 0.40 | 16.44 |
| foreign\_20 | |  |  |  |  |  |
| csplit\_50home | 24.90 | 10.37 | 39.66 |  |  |  |
| csplit\_75home | -23.54 | -40.44 | -6.45 | -13.94 | -20.52 | -7.88 |
|  | US Version | | | | | |
|  | Class 1 | | | Class 3 | | |
|  | Mean | LB | UB | Mean | LB | UB |
| beach\_25 | 86.00 | 54.58 | 120.43 | 14.90 | 2.03 | 28.79 |
| beach\_50 | 142.84 | 110.29 | 179.05 | 18.00 | 4.43 | 32.42 |
| beach\_75 | 199.54 | 165.00 | 238.72 | 24.28 | 10.58 | 38.26 |
| beach\_90 | 181.61 | 144.11 | 225.49 | 35.48 | 21.69 | 50.34 |
| coastal\_5 |  |  |  | 33.22 | 19.61 | 47.21 |
| coastal\_10 | 54.17 | 22.51 | 86.83 |  |  |  |
| coastal\_15 | 79.96 | 48.80 | 113.22 | 19.89 | 7.01 | 33.04 |
| coastal\_20 | 80.43 | 47.60 | 115.86 |  |  |  |
| intl\_3 |  |  |  | 13.90 | 1.06 | 27.88 |
| intl\_5 |  |  |  |  |  |  |
| intl\_10 | 49.47 | 17.74 | 82.26 |  |  |  |
| intl\_15 | 48.40 | 21.96 | 76.01 |  |  |  |
| foreign\_5 |  |  |  |  |  |  |
| foreign\_10 | 32.16 | 3.75 | 61.17 | -18.35 | -30.36 | -6.70 |
| foreign\_15 | |  |  |  |  |  |
| foreign\_20 | |  |  |  |  |  |
| csplit\_50home | 58.52 | 36.85 | 81.20 | 26.06 | 0.88 | 51.82 |
| csplit\_75home | 26.06 | 0.88 | 51.82 |  |  |  |
| Note: LB = lower bound; UB = upper bound; Unit: USD | | | | | | |

1. An example of the former includes environmental quality improvements to large bodies of water that border multiple countries, such as the Baltic Sea (Ahtiainen et al., 2014; Czajkowski et al., 2015). Examples of the latter include iconic ecosystems such as the Amazon rainforest, great barrier reef, or iconic species such as marine mammals (Ressurreição et al., 2012; Rolfe & Windle, 2012; Strand et al., 2017). [↑](#footnote-ref-1)
2. An alternative would have been to frame the DCE in terms of a multilateral agreement with different effects on plastic across three or more nations. Such a framing would have required greater complexity and a larger number of attributes in the DCE. [↑](#footnote-ref-2)
3. The UK and US are the largest individual contributors to plastic accumulation in the North Atlantic gyre; with Europe contributing between 16-24% of total mass and Central and North America contributing 64-65% of total accumulation (Lebreton, Greer, & Borrero, 2012). [↑](#footnote-ref-3)
4. In a comparable study for greenhouse gas reductions, Lee & Cameron (2008) found such a cost share to be an important aspect of US citizens’ willingness to support a climate change reduction program. [↑](#footnote-ref-4)
5. An outline of how the status quo values were established for each attribute is described in Appendix A. [↑](#footnote-ref-5)
6. The difference between possible reductions on beaches versus coastal waters reflects the fact that removal of plastic pollution already in the natural environment is most effective on land, whereas necessary technology to clean up water-borne plastic waste is still under development. [↑](#footnote-ref-6)
7. This study explicitly focus on macro-debris (>20mm in diameter) and meso-debris (5-20mm in diameter) (Barnes et al., 2009), and did not include discussion on micro plastics. [↑](#footnote-ref-7)
8. The project’s stakeholder advisory board (which includes direct representation from both UK and US government agencies and policy communities) provides a mechanism to make good on this statement. [↑](#footnote-ref-8)
9. Connecticut, Delaware, Florida, Georgia, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, North Carolina, Pennsylvania, Rhode Island, South Carolina, Virginia, Washington D.C. [↑](#footnote-ref-9)
10. For analysis, we have converted the UK version cost levels to the USD equivalent using the OECD purchasing power parity exchange rate at time of writing. This was 1 USD = 0.6995 GBP (*PPP and Exchange Rates*, 2020). [↑](#footnote-ref-10)
11. See Greene & Hensher (2003) for more information on latent class models. [↑](#footnote-ref-11)
12. Policy and payment consequentiality are prerequisites for incentive compatibility (e.g. Herriges, Kling, Liu, & Tobias, 2010; Vossler & Watson, 2013). Other covariates were considered (e.g., whether an individual is retired, education level) but were removed due to insignificance. [↑](#footnote-ref-12)
13. A set of tests rejects the null hypothesis (p<0.05) that the distribution of responses is the same for the UK and US versions, for all Likert variables except payment consequentiality (*payment\_con*). [↑](#footnote-ref-13)
14. Other model specifications were estimated. Output from the standard multinomial logit model is presented in Appendix B. [↑](#footnote-ref-14)
15. Within any LCM, one must determine the number of classes and the variables (if any) that will determine class membership. The number of latent classes cannot be determined a priori and there is no theory to guide the initial number of classes. Previous studies have relied mainly on information criteria such as Akaike information criteria (AIC) and Bayesian information criteria (BIC) to make this choice. [↑](#footnote-ref-15)
16. Class 1 also prefers cost to be split 50:50 between the UK and US and dislikes any program that has the UK paying a larger share than the US. The significant and negative parameter on the ASC indicates significant preferences for any program over the status quo alternative, independent of changes in choice attributes. [↑](#footnote-ref-16)
17. Relative to Class 1, these respondents are more likely to believe they already pay enough in taxes and that it is not their responsibility to pay for plastic reduction. They are less likely to agree with consequentiality statements. Class 2 are more likely to be female, older, and lower income than Class 1. Around 21.5% of these respondents always selected the status quo. [↑](#footnote-ref-17)
18. These respondents also dislike a cost split where the UK pays more than the US, and are less likely to choose the status quo. They are significantly more likely to believe they pay enough in taxes and that this problem is not their responsibility, and less likely to believe that it is important to reduce marine plastic. These respondents are more likely to be older, women, and lower income. Additionally, they are less likely to believe in the policy and impact consequentiality statements, compared to Class 1. [↑](#footnote-ref-18)
19. This class has a strong preference for 50:50 or 75:25 cost split between the US and the UK, compared to the baseline level of 25:75. The negative marginal utility for the ASC suggests preferences for a program that reduces marine plastic pollution over the current status quo. [↑](#footnote-ref-19)
20. Relative to Class 1, these individuals are more likely to believe they pay enough in taxes and that it is not their responsibility to pay for plastic reductions. They are also less likely to believe that it is important to reduce marine plastic and significantly less likely to agree with all consequentiality statements. They are more likely to be female, older and lower income. [↑](#footnote-ref-20)
21. These individuals prefer program cost to be split evenly and dislike programs where the US pays more. Compared to Class 1, Class 3 respondents are more likely to believe that they pay enough in taxes and less likely to believe that plastic pollution is important. They are also less likely to believe in the payment consequentiality statement and more likely to be older and have lower income. [↑](#footnote-ref-21)
22. We do not include marginal WTP for the cost-split attribute, as it is unclear the interpretation of WTP for X% of the cost share. Additionally, exploration of this cost share attribute is explored in other related papers. [↑](#footnote-ref-22)
23. UK respondents exhibited significant marginal utility for 5% international reduction, however the 95% confidence interval spans across zero, therefore it is plotted along the zero line in Figure 3. [↑](#footnote-ref-23)
24. Kolstad (2014) also finds that when agents are heterogenous based on country size, and therefore heterogenous in their polluting potential, this does not affect the size of coalitions forming, but does influence the overall abatement level with coalitions of large countries abating more. Although not a primary focus of the present study, we also found that individuals of the larger country (i.e., the US) exhibit higher WTP in general, for most types of plastic reductions. Therefore, the marginal per capita return, i.e., the ratio between marginal benefit and marginal cost, will be higher in the US than the UK, if we assume equal levels of marginal cost. [↑](#footnote-ref-24)