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Article

Which policy instrument do citizens and civil servants prefer? A choice experiment on Swedish marine policy

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Abstract

In the choice between alternative environmental policy instruments, economists tend to favor policies capable of attaining cost-efficiency, but other considerations may be more important to stakeholders. We perform a choice experiment modeled on Swedish water and marine policy to estimate preferences for different policy instruments among citizens and municipal civil servants. Both the modal citizen and the modal civil servant prefer direct regulation and subsidies to nutrient trading. Moreover, nutrient trading is unlikely to deliver sufficiently large cost savings for civil servants to prefer it to other instruments. These results are consistent with the apparent reluctance to adopt water quality trading in Europe.

Keywords: Choice experiments, Instrument choice, Nutrient trading, Policy acceptance, Marine policy

JEL codes: H23, Q53, Q58

1 Introduction

In the choice between alternative environmental policy instruments, economists tend to favor incentive-based policies capable of attaining a given environmental quality in a cost-efficient way (Goulder and Parry 2008); yet if support among citizens or decision-makers is lacking, this is likely to strongly impact the likelihood that such policies will be implemented (Schneider and Volkert 1999; Oates and Portney 2003). Within the domain of water quality, and specifically policies to mitigate eutrophication problems, nutrient credit schemes are frequently suggested in the economic literature (Horan and Shortle 2005). Fisher-Vanden and Olmstead (2013) list a number of such trading schemes that are now in operation in the USA, Canada, and Australia, and there is also a trading system in New Zealand; see Duhon et al. (2015). However, despite water quality problems being widespread, takeup outside of these countries has not occurred, and most trading systems are local in scope. What is the cause of the apparent reluctance to adopt nutrient trading?

One likely reason, of course, is that incentive-based policies tend to be less straightforward for water than for air, where trading has been relatively more successful. This is typically due to both physical and institutional constraints, including spatial variation in environmental damages (Montgomery 1972; Krupnick et al. 1983; Hung and Shaw 2005); the inherent difficulty of monitoring non-point emissions (e.g. from agriculture) compared to emissions from point sources (Malik et al. 1993; Horan 2001; Horan and Shortle 2005); and interactions between nutrient trading and existing policies for the relevant sectors.

Although such design difficulties may have limited the spread of nutrient trading schemes in general, their use in some regions but not in others suggests that other factors may explain the decision to implement trading in a given country. One such factor is preferences of regulators and the general public for different types of policy instruments. Thus, the purpose of this paper is to investigate whether regulators and/or the general public hold preferences over nutrient trading and other types of policy instruments, *regardless of their environmental or economic impact*.

We do so by estimating the willingness to pay (WTP) of citizens and civil servants in (local) government for meeting water quality targets specifically through the use of nutrient trading as opposed to direct regulation ('command-and-control') or subsidies. The setting for our study is Sweden, which is one out of nine countries surrounding the severely eutrophicated Baltic Sea (HELCOM 2014). Out of the nine countries, Sweden is the only one where nutrient trading has been seriously considered by the legislature, based on a series of public inquiries made by the Swedish Environmental Protection Agency.

We are aware of no other paper studying preferences for policy instruments targeting water quality; this being said, the general idea that some policy instruments may, all else being equal, be preferred to others is not new. There is, for example, a well-documented tendency for a large fraction of lab subjects to vote against proposed 'taxes' that impose marginal costs on certain decisions but are expected to increase individual payoffs as well as efficiency (e.g. Kallbekken et al. 2011; Cherry et al. 2012, 2014, 2017; Heres et al. 2017). These choices do not appear to be caused by confusion: Kallbekken et al. (2011) found that explaining the properties of the tax made experimental subjects more likely to correctly predict its impact, but did not make them significantly more likely to vote in its favor. Moreover, Cherry et al. (2017) found clear evidence that policy aversion is mediated by subjects' cultural worldview, for example, whether public intervention against private interests is considered legitimate.

While such laboratory studies grant researchers a desirable degree of control over choice environments, this may come at the expense of realism. We therefore take the complementary approach of estimating instrument-specific preferences within a stated-preference discrete choice experiment explicitly modeled on actual Swedish marine policy. The respondent pool consists of two groups: (i) Swedish citizens and (ii) municipal civil servants specializing mainly in issues related to the environment or water quality and sewerage. The inclusion of municipal civil servants is motivated by existing marine and water policies typically being applied at local or regional levels; thus, municipal employees have important roles to play in the introduction of any nutrient-trading scheme. Each choice set in the experiment asks respondents to choose between alternative policies for meeting Swedish obligations under the 2007 Baltic Sea Action Plan. Each option includes an attribute describing whether the policy instrument used is direct regulation, agri-environmental subsidies, or nutrient trading.

Preferences for the instrument-type attribute need to be estimated with care. While it seems likely that preferences for any policy instrument depend on its perceived attributes, certain characteristics are arguably included in the very definition of the policy instrument. For instance, it is difficult to see how a preference for emissions trading *per se* could be separated from a preference for a market-based quantity instrument. Other attributes are not fully correlated with the type of policy instrument employed. For example, even though theory predicts that an emission-trading system reaches a given target at minimum cost, the actual cost advantage compared to command-and-control will depend on cost heterogeneity among firms, the willingness of firms to actually trade their permits and relative administrative costs. Moreover, the ability of a water quality trading system to meet water quality targets with high accuracy is dependent on whether trading ratios between different types of sources are correctly determined (Horan and Shortle 2005).

Our design is based on the assumption that respondents are capable of viewing partially correlated attributes as separate from the policy instrument itself. Thus, we aim to capture preferences for policy instruments *per se*, i.e. only for those characteristics that are strictly inherent to each policy instrument. To this end, each choice situation in our experiment explicitly includes and accounts for correlated attributes such as cost or 'delivery uncertainty' (Glenk and Colombo 2011); estimates of policy-instrument preferences may otherwise mostly reflect those perceived correlations.

In our view, any remaining preferences for policy instruments *per se* may well be ethical in nature; for instance, see Kverndokk (2013) and Braaten et al. (2015) for explorations of the ethics of emission trading. In particular, our background information describes a trading scheme similar to actual Swedish policy proposals, where regulated-point sources are allowed to buy offsetting abatement measures elsewhere. Since this departs from the 'polluter pays principle', it may activate fairness preferences if regulated sources are thought to evade their just responsibilities. In our approach, such fairness implications would be inherent to the proposed instrument, and hence not controlled for by our design.

Only a few choice experiments have explicitly considered preferences across different policy instruments. Among these studies, the one with a design most similar to ours is the choice experiment by Johnston and Duke (2007). They estimate preferences for alternative land conservation measures (zoning, outright purchase, etc.), and include additional attributes with the explicit aim of controlling for outcomes that respondents may consider correlated with instrument type. Other studies compare policy-instrument preferences within the climate domain. For example, Alberini et al. (2018) similarly include policy instrument directly as an attribute in their choice sets, finding that subsidies are preferred to information-based approaches as well as to taxes. Bristow et al. (2010) and Brännlund and Persson (2012) use the alternative approach of labeling entire choice-set alternatives by instrument. Brännlund and Persson (2012) find that the generic label 'Other' is preferred to 'Tax'; in Bristow et al. (2010), instrument labels are confounded with attribute levels, making clean comparisons across instruments difficult.

The set of stated-preference studies that specifically compare emission trading with other instruments is smaller still. The one most relevant to our paper is the contingent-valuation study of Kotchen et al. (2013). They elicit WTP among US citizens for carbon taxes, emission trading, and direct regulation of greenhouse gases, finding few significant differences. In their study, however, each respondent is confronted only with a single policy instrument and thus does not choose between, for example, taxes and cap-and-trade. As a result, respondents cannot be instructed to hold all aspects of policy except instrument type equal, so the resulting estimates may reflect preferences for correlated policy attributes as well as for policy instruments as such, which is where our focus lies.

As a final note, our article also builds on a small set of choice experiments that examine differences between members of the general public and experts or civil servants (Carlsson et al. 2011; Rogers 2013; Nordén et al. 2017; Eggert et al. 2018). All of these studies differ from ours in that they examine preferences or WTP for some environmental target or good, irrespective of the policy instrument used. We do, however, follow most of them in one important respect: framing our questionnaire slightly differently across the two samples. Because we are interested in policy support, we ask citizens to consider impacts on their own finances, while civil servants are invited to make a more detached professional judgment. As a result, trade-offs with respect to policy costs have different interpretations across samples

(Rogers 2013). However, varying the framing in this way allows us to represent with greater realism the contribution of each sampled group to the policy process.

Thus, this paper makes a two-fold contribution to the literature. It is the first study of preferences for policy instruments for water quality. It also contributes to the general literature on environmental valuation by assessing whether civil servants as well as citizens have systematic preferences for policy *instruments*, rather than their preferred level of an environmental good.

The remainder of this article is structured as follows. The following section provides brief background on the issues underlying the choice experiment, i.e. Swedish marine and water policy. We then describe the implementation and design of our survey in more detail and outline our econometric strategy. Next, we summarize our results. Finally, we provide some concluding remarks.

2 Swedish water and marine policy

The 2007 Baltic Sea Action Plan set country and basin-specific 'Maximum Allowable Inputs' (MAI) of nitrogen and phosphorus, to be reached by 2021. In the analysis of the Swedish Agency for Marine and Water Management (2016), Sweden was found in compliance with all but two of these basin- and nutrient-specific targets. First, the MAI for nitrogen discharges to the Bothnian Bay is set at 17,924 tons/year, but actual discharges were calculated at 19,500 tons/year. Second, the MAI for phosphorus discharges to the Baltic Proper is at 308 tons/year, but current discharges were estimated at 780 tons/year.

Of these targets, the second is likely the more problematic one, for at least three reasons. First, eutrophication is more severe in the Baltic Proper than in the Bothnian Bay (HELCOM 2014). Second, the limiting nutrient in the Bothnian Bay is not nitrogen, but phosphorus (Swedish Environmental Protection Agency 2014). Third, the calculations for the sixth Pollution Load Compilation separated total loads into anthropogenic and background loads, and found that the background load alone for phosphorus in the Baltic Proper (370 tons/year) exceeds the Swedish MAI to the same basin. Thus, meeting this target is likely to be very challenging.

Forty per cent of the Swedish anthropogenic net phosphorus load to the Baltic Proper arises within the agricultural sector, with wastewater treatment plants being the second largest source (22 per cent). However, discharges from large wastewater treatment plants have been substantially reduced since the year 2000, mainly because of bans on using phosphates in detergents (South Baltic Water Authority 2014a).

Beyond the BSAP targets, Sweden is also obligated by the EU Urban Wastewater Treatment Directive (91/271/EEG) and the EU Water Framework Directive (2000/60/EC) to reduce emissions of nutrients to inland lakes, rivers, and coastal waters. The Water Directive requires all such water bodies to achieve 'good' or 'high' ecological status, including with respect to eutrophication, by 2021 or 2027. These targets are currently relatively far from being met. Within the Baltic Proper catchment area, 28 per cent of all water bodies have yet to be classified, but of those remaining, only 48 per cent currently have good or high ecological status with respect to nutrients.

As for Swedish marine and water policy, it relies heavily on environmental subsidies for agriculture, and legal requirements and permitting for point sources (wastewater treatment plants and industries). Point-source regulation involves permit requirements set based on environmental quality standards ('EQ standards') mainly corresponding to good or high ecological status. Most of these are local in scope and concern lakes, rivers, and Swedish coastal waters. In agriculture, the Swedish Rural development program provides funding (supplemented with information campaigns) for farmers willing to take abatement measures.

While it is clear that Swedish policies have been effective in reducing nutrient loads, economists have found that outcomes have not been cost-efficient (e.g. Gren et al. 1997; Elofsson 2010, 2012). The past few decades have seen increased interest in economic instruments capable of bringing down abatement costs, especially nutrient discharge trading systems, although proposals by the Swedish EPA to introduce such systems have ultimately been unsuccessful. An initial proposal (Swedish Environmental Protection Agency 2008) outlined a trading scheme where a regulated sector (e.g. municipal wastewater treatment plants) is able to meet binding emissions standards by financing compensatory measures within a non-regulated sector (e.g. agriculture), with these transactions handled by the regulating authority, which acts as a clearinghouse.

The proposal was, however, widely criticized by stakeholders for conflicting with existing regulations. First, agri-environmental subsidies may undermine farmers' incentives to supply compensatory measures. Second, credit payments for measures that are already subsidized likely conflict with additionality requirements within the Rural development program. Third, the point of trading systems is to carry out load reductions (with respect to the Baltic) where they are least expensive. A potential side effect is that measures may be diverted from inland waters subject to EQ standards. While it may be possible to add special provisions to avoid such regulatory conflicts, these auxiliary rules will undermine the cost-efficiency of the trading system. The situation is especially problematic if, as is the case, EQ standards are both abundant and stringent.

A second proposal for nutrient trading was presented in Swedish Environmental Protection Agency (2012). The updated trading system was less ambitious, covering only municipal wastewater treatment plants and not allowing offsets from, for example, agriculture. This resolved conflicts with existing policies to some degree. Also, trading covered only nitrogen emissions. EQ standards for water quality largely concern phosphorus, and include nitrogen obligations only for Swedish coastal waters, where significant synergies with BSAP targets are likely. Despite this, the government decided against adoption, stating that the current approach of regulating nitrogen and phosphorus by permitting is 'difficult to reconcile with a charge system for these emissions'. Subsequently, Swedish policy initiatives have again focused mainly on direct regulatory approaches and agri-environmental subsidies (e.g. North Baltic Water Authority 2016).

3 Materials and methods

3.1 Implementation

Our study separately estimates the preferences of the general public and municipal civil servants with respect to the policy-instrument type attribute. Previous research (e.g. Colombo et al. 2009; Carlsson et al. 2011; Rogers 2013; Nordén et al. 2017) has demonstrated that preferences for environmental policy can differ substantially between citizens and experts, bureaucrats or various stakeholder groups, with the experts and civil servants typically expressing a higher valuation of attributes related to environmental quality (for an exception, see Nordén et al. 2017). However, they focus on the stringency of environmental policy itself rather than the choice of instrument in attaining a given target. Our experiment is designed to permit estimation of preferences for instrument type among citizens and civil servants.

Our sample consists of two groups of respondents. First, in April 2017, the choice experiment was sent out to a panel of Swedish citizens aged 18–75.[1] Data collection concluded by May 2017 and yielded 2001 complete responses from this group. Second, also in May 2017, we sent an email to the registry office of each municipality in Sweden, requesting that an online link to the survey be forwarded to as many municipal employees as possible working within the water quality domain. We specifically asked for civil servants of any rank working with (i) environmental issues, particularly water quality, (ii) water and sewerage, and/or (iii) local business. Note that the open-ended nature of our invitation makes response rates difficult to evaluate. By August 2017 (after some reminders), 146 civil servants had completed at least part of the survey, and 115 respondents had completed the entirety of the choice experiment (though not necessarily the full post-experimental questionnaire).[2]

Prior to the main data collection phase just described, the survey was pretested in two ways. First, we carried out qualitative pre-testing through interviews with approximately ten members of the general public (all non-researchers with a university degree) as well as representatives of the Swedish Association of Local Authorities and Regions. This pre-testing aimed to ensure that the language and structure of the survey were well understood and credible to both municipal employees and members of the general public, and several modifications were made as a consequence of the discussions. Second, we recruited a sample of sixty-six citizen respondents to conduct a pilot study of the full design, including some preliminary statistical tests. The data and results of this pilot indicated that the survey was generally well designed and understood, and only minor changes were made.

Online Appendix Table A.1 presents summary statistics for both samples as well as for the population of Sweden, with statistical tests performed wherever possible.[3] In terms of the geographical distribution across counties, there are no obviously major differences between our samples and the general population, with the possible exception that civil servants from Stockholm County are relatively rare. Citizen respondents do have somewhat more education, live in smaller households, and have slightly higher incomes than the population at large.

3.2 Survey design

The survey was an online questionnaire consisting of several parts. Respondents first read a general description of the study design, including a cheap-talk script similar to that developed by Carlsson et al. (2005). The script emphasizes respondents' tendency to state a high WTP without thinking carefully about impacts on their household budget (a translated version of the script is given in Online Appendix B.1), and including this message has been shown to yield significantly lower and arguably more reliable WTP estimates. This section of the survey also contained a link to optional background information on Swedish marine and water policy, which described the Swedish BSAP target for phosphorus discharges to the Baltic Proper, as well as EU targets for good ecological status in lakes, streams, and coastal waters within the Baltic Proper catchment area. A translation of this information is given in Online Appendix B.2.

Following the existing research on the benefits from initiating preference formation (Ben-Akiva and Gershenfeld 1998; Cohen and Liechty 2007), the next part of the survey familiarized respondents with each attribute included in the choice experiment. We included this section to make sure that each participant understood the meaning of all attributes and as an *ex-ante* measure against attribute non-attendance. First, participants read a brief description of an attribute, and were asked to select their preferred level for that attribute. After this process had been repeated for each attribute, respondents then progressed to the choice experiment itself.[4]

Table 1 lists all attributes and their associated levels; translations of the attribute and level descriptions are given in Online Appendix B.3. Our main attribute of interest is 'Type of policy instrument'. Since it is central to our research question, below we reproduce the (translated) description of nutrient trading given to respondents. The description is designed to match existing water-quality-trading schemes (Shortle 2013). Compared to the Swedish trading proposal discussed in Section 2, the main difference is that we do not mention the regulator as a potential clearinghouse for trades.

Table 1. Attributes and levels in the choice experiment.

Attribute	Opt-out	Alternative
Target compliance: Baltic Proper (per cent)	12	40; 70; and 100
Target compliance: lakes, streams, and coastal waters (per cent)	50	65; 80; and 100
Type of policy instrument	N/A	Legislation and permitting; emissions trading; and environmental subsidies
Likelihood that policy is effective	N/A	Very certain; rather certain; rather uncertain; and very uncertain
Cost to farmers (SEK per year per farmer)	0	+10,000; +20,000; +30,000
Cost to taxpayers (SEK per year per taxpayer)	0	+100; +150; +200; +250; and +300

Emission trading: This is a policy instrument that aims to create a market for pollution abatement measures. For example, stricter obligations to reduce emissions from municipal wastewater treatment plants may be supplemented with the possibility for the wastewater treatment plants to 'buy' pollution-abating measures elsewhere. The treatment plants could, for example, compensate farmers that construct wetlands or riparian strips on their property. These measures can then replace measures in the treatment plants, provided the effect on the environment is equally large. Such trading possibilities are not present [under legislation and permitting].

As Johnston and Duke (2007) argue, it is likely that preferences for different types of policy instruments are partly driven by, and thus confounded with, preferences for outcomes believed to be correlated with the use of those instruments. Within the context of marine policy, for example, a preference for environmental subsidies could be driven by a desire to safeguard farmers' competitiveness. Explicitly adding farmer profits as an attribute within the choice sets may alleviate this problem, leading to better estimates of preferences for each instrument *per se*, which was the main objective of the survey. We therefore included the cost to farmers as a separate attribute, along with two others likely to be seen as correlated with instrument type, namely (i) compliance with targets for good/high ecological status in domestic inland waters and (ii) perceived 'delivery uncertainty' as to whether the policy will have the intended effect. Delivery uncertainty can be included in choice sets as a quantitative (Glenk and Colombo 2011) or qualitative (Lundhede et al. 2015) attribute. Because of the inherent difficulty in quantifying *ex ante* the likelihood that a policy will be effective, using exact probabilities might make our scenario less credible to respondents, and we therefore used qualitative levels.

While we cannot rule out the existence of additional omitted attributes, the discussion in Section 2 suggests that the Swedish debate on instrument choice in marine and water policy is largely framed around these three additional attributes. Another important caveat (Lundhede et al. 2015) is that respondents may not simply accept stated attribute levels, but may engage in 'scenario adjustment' (Cameron et al. 2011), i.e. basing their choice partly on their priors regarding, for example, the perceived effect of policy. This suggests that while our approach of including 'control' attributes is probably useful, it may not fully solve the omitted-variables problem even with respect to those attributes. In Online Appendix C, we therefore look for indirect evidence of scenario adjustment by studying interactions between instrument type and other included attributes. We conclude that there is little evidence of such interactions, supporting our approach.[5]

An example choice set is presented in Online Appendix B.4 (along with screenshots in Online Appendix B.5). At any point during the choice experiment, participants could review the background information, and could also recall attribute descriptions by hovering the mouse cursor over an attribute. Each choice set in the experiment included an invariant opt-out option reflecting the likely outcome in the absence of changes to current Swedish marine and water policy, as well as two alternative, more ambitious policy packages. Since our survey consistently framed choices as relating only to *additional* policies or measures to reduce emissions beyond the status quo, the opt-out level for the uncertainty attribute was given as a blank space rather than, for example, 'Very certain'. For the same reason, the opt-out level of the 'Type of policy instrument' attribute, likewise framed in terms of additional future policy, was also given as a blank.

We used a D-efficient design generated by the market research firm GfK Norm under the assumption of a fully dummy-coded preference-space model with zero priors. Attribute levels were calibrated to real-world figures,[6] and their combinations were chosen to maximize estimation efficiency given two restrictions: neither 'Emissions trading' nor 'Environmental subsidies' was ever combined with the highest level of 'Cost to farmers' (+30,000). We included these restrictions to make choice situations realistic and policy-relevant.

The survey faced by each respondent consisted of twenty choice sets. In Section 4.3, we check whether this relatively large number of choice sets elicited inconsistent preferences, such as a fatigue effect leading to less considered choices in later tasks (Swait and Adamowicz 2001). At this point, we simply note that increased survey length need not be detrimental to choice reliability and precision. Hess et al. (2012) examine fatigue effects across five data sets, concluding that at least up until around fifteen to twenty choice sets, greater length might even improve reliability through a learning effect. Johnson and Orme (1996) and Carlsson et al. (2012) report similar findings. Furthermore, we used a heterogeneous design where each subject faced one out of fifty distinct surveys, each including a set of twenty tasks. Such a heterogeneous design approach increases the variation in attribute levels across the entire sample of respondents and has been shown to reduce scale bias and provide substantial efficiency improvements, especially when the number of attributes is relatively large, as in our study (Sándor and Wedel 2005).[7]

The final stage of the survey was a questionnaire on mainly demographic and socioeconomic characteristics. The endline survey given to citizens was significantly longer than for civil servants and included, for example, items on respondents' prioritization of economic growth versus environmental protection, as well as their trust in private corporations and public institutions. Both survey versions included a two-part item on attribute non-attendance, asking respondents whether they had 'ever disregarded the level of some attribute while choosing among alternatives', and if so, which attribute(s) they had disregarded. We elicited stated non-attendance in this way—i.e. after the entire choice experiment was complete rather than after each choice task—to avoid priming effects, as suggested, for example, by Scarpa et al. (2010). Finally, we also asked all respondents whether they had read the background and attribute information, and if so, how difficult to understand they found those descriptions.[8]

Since we wanted civil servants to respond in their capacity as municipal employees rather than as private citizens and/or taxpayers, certain aspects of the pre-experimental information differed between the samples. In particular, while citizens were asked to select the alternative that they would prefer for society to adopt, the civil servants were prompted to select the alternative that 'is the best, given the conditions that apply within your municipality'. For civil servants, the tax-cost attribute was also framed in less personal terms, for example, by replacing the label 'Cost to you as a taxpayer' faced by citizens with 'Cost to taxpayers' (see Online Appendix B.3). Thereby, their responses provide information on the trade-off that municipal civil servants make between taxpayer costs and other attributes, and this may not reflect their own private WTP. There were no other substantial differences across the two surveys.

3.3 Econometric specification

We base our empirical analysis of both citizens' and civil servants' responses on a standard random-utility framework. For citizens, we may think of choice as reflecting maximization of private utility; for civil servants, following Carlsson et al. (2011), we assume that choice reflects maximization of the utility of a representative citizen within the relevant municipality. As in that paper, we cannot and do not rule out that motivation could be more complex. For instance, choices by civil servants could be driven by paternalism or misperceptions of what the representative citizen prefers, and citizen choice could reflect altruism or some other social preferences. Such various motivations are not inconsistent with utility maximization (e.g. Konow 2000; Andreoni and Miller 2002; Fisman et al. 2007) and thus do not affect the basic applicability of the random-utility model.

In either sample, each respondent i's utility of selecting an alternative j in choice set t is assumed to be given by

$$U_{ijt} = v(\mathbf{X}_{ijt}) + \epsilon_{ijt}, \tag{1}$$

where v is a systematic component and ϵ is a random error term, here assumed to be distributed type one extreme value. The function v is assumed to take as argument an observable vector $\mathbf{X}_{ijt}$, which could include attribute levels as well as personal characteristics. We will focus on the case where $\mathbf{X}_{ijt}$ contains only attribute levels and an alternative-specific constant (ASC) associated with choosing a non-opt out policy package. Individual i is taken to choose an alternative $j = A$ over $j = B$ if $U_{iAt} > U_{iBt}$.

We estimate a random parameter (or mixed) logit model in WTP space (Train and Weeks 2005), with 'Cost to taxpayers' as the base attribute. This model specifies the systematic component of equation (1) as $v = \theta_i(C_{ijt} + \boldsymbol{\lambda}_i'\mathbf{Z}_{ijt})$. In this expression, θ_i is respondent i's (scaled) marginal utility of the tax-cost attribute C_{ijt}, while $\boldsymbol{\lambda}_i$ is the vector of WTP parameters for the other attributes, $\mathbf{Z}_{ijt}$. This involves making two assumptions. First, systematic utility is assumed linear in each attribute. Second, there is potential heterogeneity in tastes: both α_i and each WTP ratio $\lambda_{ik} = \beta_{ik}/\theta_i$ (for non-tax attribute k and its marginal utility β_{ik}) are allowed to vary across respondents according to some prespecified probability distribution. Random parameter logit additionally exploits the panel structure of our choice data in that parameters are assumed to be constant within each individual, i.e. across all choice sets faced by a given respondent.

For estimation we use the Matlab package used in, e.g. Czajkowski et al. (2016).[9] We assume that both θ_i and each λ_{ik} parameter (including that for the ASC) are normally distributed across the population. The likelihood function associated with random parameter logit cannot be evaluated directly (Revelt and Train 1998), so we estimate the parameters using maximum simulated likelihood with 20,000 scrambled Sobol draws for our main analysis, and 2,000 draws in all other cases.

4 Results

The structure of this section is as follows: First, we report the results from our random parameter logit regression in WTP space, including various extensions. Second, we perform a robustness test with respect to attribute non-attendance. Finally, we explore whether choices differed across choice tasks in the experiment, for instance, because of fatigue or learning effects. We begin with the WTP estimates.

4.1 Estimates of marginal rates of substitution

Table 2 presents estimates of respondents' marginal rate of substitution between tax cost and each other attribute. For expositional purposes, we refer to these figures as WTP for both citizens and civil servants. However, as noted in Section 3.2, civil servants were

Table 2. WTP-space random parameter logit estimates.

	Citizens		Civil servants		Cross-sample comparison
	Mean WTP	Standard deviation	Mean WTP	Standard deviation	*P*
ASC, non-opt out	1.225***	1.867***	5.342**	3.004*	0.124
	(0.070)	(0.119)	(2.674)	(1.661)	
Baltic Proper	0.296***	0.432***	1.175*	1.619*	0.175
	(0.022)	(0.027)	(0.648)	(0.866)	
Lakes, streams, and coastal waters	0.215***	0.295***	0.649	1.472	0.314
	(0.023)	(0.033)	(0.430)	(0.915)	
Emissions trading	−0.092***	0.225***	−0.829*	0.896*	0.095*
	(0.010)	(0.014)	(0.441)	(0.475)	
Legislation and permitting	0.013	0.222***	−0.205	1.141*	0.258
	(0.011)	(0.016)	(0.192)	(0.620)	
Rather certain	−0.104***	0.065***	−0.372*	0.174	0.211
	(0.009)	(0.014)	(0.214)	(0.289)	
Rather uncertain	−0.461***	0.244***	−1.550*	0.765*	0.181
	(0.025)	(0.016)	(0.814)	(0.422)	
Very uncertain	−0.642***	0.408***	−2.271*	1.133*	0.170
	(0.034)	(0.024)	(1.188)	(0.608)	
Cost to farmers	−0.013***	0.022***	−0.059*	0.093*	0.149
	(0.001)	(0.001)	(0.032)	(0.049)	
Cost to taxpayers (marginal utility)	−4.434***	3.041***	−1.527*	0.584*	
	(0.245)	(0.187)	(0.807)	(0.326)	
Observations	40,020		2,449		
Respondents	2,001		146		
R-squared (constants only)	0.429		0.398		

Note: Standard errors are given within parentheses. Asterisk (*), double asterisk (**), and triple asterisk (***) denote coefficients significant at 10, 5, and 1 per cent, respectively. Target compliance variables (Baltic Proper; lakes, streams, and coastal waters) are coded as proportions, i.e. take values between 0 and 1; these coefficients thus reflect a 100 per cent difference. Cost to farmers and cost to taxpayers expressed in thousands of SEK. Estimation is in WTP space. The marginal utility of taxpayer cost and all WTP ratios are assumed normally distributed; 20,000 scrambled Sobol draws were used. Column 'Cross-sample comparison' presents *P* values from two-sample *z* tests of the null hypothesis that mean WTP does not differ between citizens and civil servants.

discouraged from framing taxes in terms of their personal budget, and only the results for citizens should be interpreted strictly as individuals' private WTP. WTP for civil servants can instead be interpreted as the trade-off made between a given attribute and the cost incurred by a representative citizen.

Mean coefficients of continuous attributes (Baltic Proper; Lakes, streams, and coastal waters; and Cost to farmers) should be interpreted as the average WTP for a positive or negative one-unit change in each variable, expressed in thousands of SEK.[10] In the first column, for example, respondents are willing to pay an estimated additional 13 SEK to reduce farmer costs by 1000 SEK,[11] and 0.88 * 296 = 260 SEK to increase Swedish BSAP compliance from the status-quo level of 12 per cent to full compliance.[12] For discrete variables, mean estimates describe the average WTP to move from the reference attribute level to the level of the variable.[13]

For citizens (columns 1 and 2), mean WTP is generally highly significant. As expected, it is positive for both environmental attributes. Furthermore, WTP to avoid uncertain policy outcomes is monotonically increasing in the degree of uncertainty, and there is a positive WTP to avoid increased farmer cost. The coefficient for costs to taxpayers is also negative. Finally, on average, citizens and civil servants prefer both environmental subsidies and legislation and permitting to nutrient trading. The point estimates on emissions trading suggest

Table 3. Distribution of estimated individual preferences for instrument type.

Preference orderings	Respondent share, citizens (per cent)	Respondent share, civil servants (per cent)
Subsidies ≻ Legislation ≻ Trading	17.6	26.1
Subsidies ≻ Trading ≻ Legislation	12.8	20.6
Trading ≻ Subsidies ≻ Legislation	17.5	10.2
Trading ≻ Legislation ≻ Subsidies	8.5	2.6
Legislation ≻ Subsidies ≻ Trading	33.1	35.2
Legislation ≻ Trading ≻ Subsidies	10.4	5.2
Sum	100	100
Top choice: subsidy	30.5	46.7
Top choice: trading	26.0	12.8
Top choice: legislation	43.5	40.4
Sum	100	100

Note: Table based on 100,000 draws from normal distributions with means and standard deviations as given in Table 2. In each simulation draw, WTP values relating to instrument type are multiplied by the taxpayer-cost coefficient to produce utility differentials compared to environmental subsidies. Population shares reflect how the signs of those marginal utilities are distributed; for instance, the row labeled 'Trading ≻ Subsidies ≻ Legislation' counts the proportion of draws where the utility differential of 'Emissions trading' is found positive, while that of 'Legislation and permitting' is negative.

that Swedish citizens have a mean WTP of approximately 90 SEK/year for basing national marine and water policy on environmental subsidies rather than emissions trading.

Estimated standard deviations are highly significant among citizens, indicating there was substantial preference heterogeneity. This holds particularly true for the instrument-type attribute, where estimated means are much smaller than corresponding standard deviations. It also applies to taxpayer cost, implying that roughly 7 per cent of the normally distributed marginal-cost coefficients are positive rather than negative; among civil servants, this is true for fewer than half a percent.

Marginal rates of substitution for civil servants (columns 3 and 4) are less precisely estimated than those of citizens, largely yielding results significant only at the 10 per cent level. Point estimates are qualitatively similar to those of citizens, though larger in magnitude, possibly reflecting a smaller disutility of tax costs. In particular, mean WTP for emissions trading compared to subsidies is about an order of magnitude larger for civil servants than for citizens. When we run two-sample z tests of the null hypothesis that WTP means are equal across samples (thus comparing across the private/public choice contexts of citizens and civil servants), only the difference in emissions-trading WTP is marginally significant, at $P = 0.095$.

To further explore the distribution of preferences for instrument type, we derived population shares by making 100,000 draws from the normal distributions estimated in Table 2. Specifically, we first make separate draws from the WTP distribution of 'Emissions trading' and 'Legislation and permitting', as well as from that of 'Cost to taxpayers'. In each case, we assume that means and standard deviations are as in Table 2; for instance, WTP for 'Emissions trading' is assumed to follow $\mathcal{N}(-0.092, 0.225)$. Note that making separate draws imposes independence between the parameters. Then, within each draw, we multiply the tax-cost parameter by each of the two mean-WTP parameters. This produces the utility differential of 'Emissions trading' and 'Legislation and permitting' compared to 'Environmental subsidies', respectively, allowing us to infer how different preference orderings are distributed within the sample(s).

The result of this exercise is given in Table 3. The table confirms that there is substantial heterogeneity within the population, with the modal preference ordering (Legislation

Table 4. Compensation required for nutrient trading to be preferred to other instruments, by attribute.

	General public		Civil servants	
Attribute	Subsidy	Legislation	Subsidy	Legislation
Baltic Proper (per cent)	31.0	35.4	70.6	53.1
Lakes, streams, and coastal waters (per cent)	42.6	48.6	127.7	96.1
Cost to farmers (SEK/year)	−7,206	−8,216	−13,957	−10,503
Cost to taxpayers (SEK/year)	−92	−105	−829	−624

Note: Table presents how much more favorable each attribute would need to be in order to make trading preferred to other instruments, all else equal. The delivery-uncertainty attribute is not included because it is coded as discrete variables, making comparisons impractical.

≻ Subsidies ≻ Trading) applying to fewer than half of respondents in either sample. As shown in the lower half of the table, about 44 per cent of citizens prefer legislation and permitting to all other instrument types, while 47 per cent of civil servants hold environmental subsidies as their most preferred instrument type. While these shares do not quite reach the majority threshold, it is simple to show that in both samples the modally most preferred instrument would be chosen under any pairwise-majority rule. Finally, the table confirms that support for emissions trading is low in both samples, with no more than about 25 per cent of respondents preferring it to the other instrument types.

Given that respondents in either sample clearly do not prefer nutrient trading, a natural question is: How much more favorable would a policy involving trading need to be along other dimensions in order to be preferred to another policy based on subsidies or direct regulation? While the number of possible combinations makes it infeasible to provide a full answer to this question, some observations may be made. In Table 4, we use our mean WTP estimates (Table 2) to make the comparison separately for each attribute. For example, a representative citizen choosing between one policy package involving nutrient trading and another involving agri-environmental subsidies would prefer the former, if it improves target compliance with respect to the Baltic Proper by at least $-(-0.092/0.296) = 31.1$ percentage points, all else being equal. Note that such trade-offs are not necessarily feasible in practice, and the comparisons are made for illustration purposes.

Taking point estimates at face value, the relative strength of the aversion to trading seems substantially greater among civil servants than the general public. Civil servants often need at least twice as much compensation to be willing to switch instrument type. In fact, the improvement required to compensate a policy switch from subsidies to trading is unattainable for the ‘Lakes, streams, and coastal waters’ attribute, exceeding 100 per cent; thus, a necessary condition for acceptance would be that at least one other attribute improves as well. Arguably, the same applies to taxpayer costs: the required compensation is on the order of 600–800 SEK/year, which is much larger than the upper bound of 300 SEK/year used in the experiment.

In Online Appendix D, we also perform some exploratory analyses of specific dimensions of sample heterogeneity. First, in Online Appendix Table D.1, we estimate the moderating effect on all WTP means of a dummy for whether a respondent’s municipality lies within the Baltic Proper catchment area. Neither citizens’ nor civil servants’ WTP seem to be affected by this variable. Second, columns 1–2 of Online Appendix Table D.2 show that, while citizens who prioritize economic growth over environmental protection do not exhibit differential policy-instrument WTP, they are less willing to pay for the Baltic Sea environmental attribute. They are also less aversive to uncertainty and less willing to bear cost increases to farmers. Nearly the opposite preference pattern emerges for respondents with high institutional trust (columns 3–4 of Online Appendix Table D.2): these citizens have higher WTP for environmental attributes, are more aversive to uncertainty, and more willing to accept tax costs. However, there is again no difference with respect to preferences for instrument type.[14]

4.2 Attribute non-attendance

As noted in Section 3.2, it is possible that certain respondents ignored some attributes in the choice sets, and that such non-attendance may affect the results in Table 2. In our end-of-session questionnaire, 32.2 per cent of citizen respondents stated that they disregarded at least one attribute at some point during the choice experiment, with instrument type the most commonly disregarded attribute. This is comparable to self-stated non-attendance rates reported in other studies (e.g. Hensher et al. 2005; Nguyen et al. 2015). Stated attribute non-attendance was considerably higher among civil servants, where a majority disregarded one or more attributes; and close to 30 per cent disregarded the instrument-type attribute.

Given that instrument type is our main variable of interest, it seems useful to see whether such attribute non-attendance mediates our results. We use a non-attendance validation model (Hess and Hensher 2010), estimating two sets of (potentially non-zero) mean coefficients for each attribute in the design: one for subjects stating that they did not ignore the attribute in question and one for those stating that they did. This is implemented through interactions with a dummy for non-attendance, and thus the second group of mean parameters capture the differential between the two types of respondents. Note that respondents may have ignored the tax-cost attribute, in which case their marginal disutility from this attribute is not estimated in a WTP-space model. To avoid having to impose model restrictions to handle such situations, we estimate the coefficients in preference space.

Online Appendix Tables D.3 and D.4 present the results for citizens and civil servants, respectively. For comparison purposes, columns 1 and 2 of each table report a benchmark preference-space model that does not account for attribute non-attendance; the non-attendance models are given in columns 3 and 4. We find that a number of interaction means are significant in the citizen sample, suggesting that attribute non-attendance does make a difference for preferences. On the other hand, both the mean and the standard deviation parameters of those that did not ignore a given attribute are all nearly identical to the benchmark model. Thus, when basing WTP for each attribute solely on respondents that did not ignore it or the tax-cost attribute, as in, for example, Van Loo et al. (2018), very little information would likely be lost by disregarding attribute non-attendance. For civil servants, where estimates are generally less precise, the discrepancies are slightly larger; but only a single interaction parameter is significant. Overall, we conclude that attribute non-attendance does not appear to be a major threat to our conclusions.

4.3 Learning and fatigue effects

As discussed in Section 3.2, the questionnaire contained a large number of discrete choice tasks, so dynamic effects reflecting, for example, learning or fatigue are potentially a concern. Online Appendix Table D.5 reports output from WTP-space models run only on choice sets 1–10 (columns 1 and 2) or choice sets 11–20 (columns 3 and 4). Online Appendix Table D.6 presents the same analyses for the municipal civil servants. To check for changes in preferences over time, the last column performs pairwise z tests of the null hypothesis that mean WTP for each variable is constant across both halves of the choice experiment.

For civil servants, the test is clearly underpowered (note that standard errors are generally larger in the subsample regressions), leading to non-significant results. By contrast, citizens exhibit several significant differences between the first and second half. In particular, respondents valued environmental improvements more highly in the second half, and were more aversive to uncertain policy outcomes. However, our main focus is on the policy-instrument attribute, and these point estimates are remarkably stable over time, suggesting that mean WTP for different policy instruments is little affected by learning or fatigue.

5 Concluding remarks

This article has presented results from a choice experiment designed to estimate the preferences of citizens and municipal civil servants for three different types of policy instruments applicable to marine and water policy: (i) agri-environmental public subsidies, (ii) legislation and permitting, and (iii) nutrient trading. Respondents were presented with a hypothetical scenario based on actual Swedish conditions, and were asked to choose between alternative policies for the Baltic Proper catchment area.

Results of models estimated in WTP space indicate that both groups clearly and significantly prefer subsidies as well as legislation and permitting to nutrient trading. This applies even when explicitly holding other attributes fixed, including impacts on the Baltic Sea and inland waters, the likelihood that each policy alternative will be effective, and costs to farmers and/or taxpayers. Citizens, but not civil servants, also have a less pronounced preference for legislation and permitting over subsidies.

The choice experiment was framed in slightly different ways for citizens and civil servants, emphasizing their respective roles in the policy process. As a result, estimates of marginal rates of substitution with respect to the tax-cost attribute among civil servants are not directly comparable to WTP estimates among citizens. That being said, our estimates do suggest that civil servants require much more compensation in terms of improvements to any other attribute in order to accept a shift from another policy regime to trading. For example, taxpayer costs need to be about 600–800 (90–100) SEK/year lower under trading for it to be preferred by civil servants (citizens). There are only a few empirical estimates of the cost savings from nutrient trading in Sweden, or other regions or countries surrounding the Baltic Sea (e.g. Elofsson 2010; Swedish Environmental Protection Agency 2010, p. 86). The single most relevant study may be Elofsson (2012), who compared actual Swedish nitrogen and phosphorus reductions over 1995–2005 with a cost-efficient solution achieving the same reductions to each Baltic Sea basin. The cost differential was estimated at only around 5 MEUR per year, equivalent to approximately 7 SEK/year and Swedish taxpayer. While attaining BSAP targets (to a greater degree) would entail higher costs and thus is likely to enhance the cost savings from using economic instruments, it seems implausible that nutrient trading could deliver the compensating cost savings required.

In our view, these findings offer a partial explanation of the reluctance of Swedish policymakers to adopt nutrient trading. It seems likely that the aversion to trading partly reflects ethical concerns with such policies *per se*. We also cannot fully rule out the interpretation that respondents dislike trading simply because it has not yet been applied in actual Swedish policy. Although our survey did not discuss the current Swedish marine policy mix, we noted in Section 2 that it combines direct regulation with agri-environmental subsidies. Thus, it is possible that well-informed respondents may have taken that baseline into account, for example, by failing to ignore the sunk-fixed costs of setting up the current policy regime. This problem, of course, is not unique to our study, given that a baseline policy mix exists in any setting. Indeed, a full examination of how instrument-type preferences depend on baseline policy would require credible variation in what the baseline is, and such a broader (e.g. multi-country) design would go significantly beyond the scope of our paper.

By contrast, our data and design do plausibly rule out a number of other potential motivations. In general, we might expect Swedish preferences against water quality trading to be driven by perceptions that, for example, (i) trading systems are too complex, and information too limited or asymmetric, for regulators or trading agents to handle; (ii) the impact of emissions on the Baltic Sea and on inland waters is only partially correlated, creating hotspot issues and conflicts with existing regulation; and (iii) nutrient trading involves large administrative costs. Our design arguably accounts for all of these issues, related as they are to uncertainty about target achievement, impacts on inland waters, and costs.

Assuming that our remaining preference estimates are driven by considered views rather than, for example, confusion (cf. Cherry et al. 2014), they represent an explicit and quantitative measure of policy acceptance. More broadly, results like ours (and those of the wider policy-acceptance literature) lay bare what seems to us a pressing issue: How should economists handle citizens' and civil servants' apparent willingness to trade-off larger tax payments for a change in policy regime? In particular, consider any stated-preference study where the valuation scenarios are framed in terms of specific policy instruments. It then seems doubtful that WTP estimates can be interpreted as a measure of preferences *only* for policy outcomes, rather than as the sum of outcomes and the policy instrument.[15] A review may be valauble to determine how common such interpretations are in the literature, and to determine to what extent policy-instrument type can be accounted for without undue increases in survey complexity.

Supplementary material

Supplementary data are available at *QOPEN* online.

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Conflict of Interest Statement

The authors have no conflict of interest to declare.

End Notes

1 All data collection was carried out in collaboration with market research firm GfK Norm, which maintains this panel.

2 The subset of 110 civil servants who also responded to survey items regarding their work location and occupational field represent eighty-two unique municipalities, with ninety-seven respondents stating environmental or water/sewerage as their field.

3 All hypothesis tests in this article are two-tailed. The tests discussed here are of statistical representativity with respect to the Swedish population. Generally speaking, we reject most tests for the larger citizen sample, where statistical power is greater. Still, point estimates do not always differ very substantially between sample and population, and indeed are often larger in magnitude for the smaller sample of civil servants, where we tend not to reject the null. Thus, our exposition here focuses mainly on the point estimates.

4 While it is possible that this module might have introduced a priming effect, we believe any such effect will be minor, since the exercise covered all attributes equally and in the same order as in subsequent choice sets.

5 An anonymous reviewer commented that an alternative approach might have been to estimate a hybrid choice model (Ben-Akiva et al. 1999). If respondents fail to hold, for example, delivery uncertainty constant, then it is conceivable that some (latent) variable reflecting attitudes or preferences toward that attribute is still influencing the instrument-type coefficient, in which case a multi-equation model might be useful. Unfortunately, our survey did not elicit any indicators that might be used to identify such latent variables.

6 For instance, South Baltic Water Authority (2014b) calculated that a package of measures aimed at attaining compliance with Swedish water quality regulations would cost a total of SEK 4.5 billion, with SEK 2 billion borne by farmers (pp. 168–9). This was mostly assumed to involve direct regulatory policies, so we treat these estimates as an upper bound. Dividing the total costs by the number of taxpayers and farmers in the Baltic Proper catchment area produces costs of about SEK 28,000 per farmer and SEK 350 per taxpayer (the details of these calculations are available upon request).

7 We rotated the fifty versions such that the first participant to enter the survey website saw design 1, the second saw design 2, and the fifty-first again saw design 1. However, since some respondents accessed but did not complete the survey, the data for both citizens and civil servants include gaps in this looping sequence. Nevertheless, among citizens, each of the fifty designs was completed by at least thirty-five respondents. For civil servants, forty designs were fully completed twice or more, and forty-nine out of fifty designs were fully completed at least once. For the one remaining design (out of fifty), our data set includes a single completed choice task (out of twenty).

8 In unreported regressions, we estimate the moderating effect of these final survey items. While there are some differences from not having read the information or having found it difficult, we crucially find no significant moderation effects on the policy-instrument attribute coefficients. About 90 per cent of citizens and nearly all civil servants claim to have read the descriptions.

9 The package is available at http://www.github.com/czaj/DCE under a Creative Commons Attribution 4.0 license.

10 According to averaged exchange rates provided by the Swedish Riksbank, 1 euro equaled 9.631 SEK across 2017.

11 As of 2016, there were 7,135,173 people aged 18–75 in Sweden, and 171,400 farmers working full or part time (Statistics Sweden 2018, p. 104). Excluding farmers from the total, this implies 40.6 non-farmers per farmer, so total WTP to reduce farmers' costs by 1000 SEK is $13 \cdot 40.6 = 527.8$ SEK, or 52.8 per cent of total farmer costs. This may be compared to the 2016 PSE (producer support estimate) for the EU, which is 21 per cent and represents total support in relation to total production value (OECD 2017). Although our estimate is relatively large, it seems plausible that the WTP is higher for environmental measures specifically than for financial support generally.

12 Ahtiainen et al. (2014) use contingent-valuation methods to estimate mean WTP for full BSAP attainment compared to baseline. Their policy scenario is more far-reaching than ours, involving compliance by all Baltic Sea countries rather than Sweden alone. In line with this fact, their WTP estimate is also higher: the authors estimate purchasing power parity adjusted WTP for Swedish citizens at 75.7 euros, translating to about 923 SEK when adjusting back for differences in purchasing power. For comparison, the Swedish share of the Baltic Proper phosphorus target is about 5 per cent.

13 Since the levels of the discretized attributes (e.g. policy-instrument types) never appear as part of the opt-out alternative, the ASC will be confounded with these attributes in any regression. The constant thus captures the combined effect of (i) rejecting the opt-out alternative in favor of some other options, as well as (ii) moving from the blank attribute values associated with the opt-out alternative to a set of non-opt out reference levels. As a result, for instance, the coefficient on 'Very uncertain' should be interpreted as the difference between that level and the omitted non-opt out level for that attribute, which is 'Very certain'. Our findings are robust to using other reference levels.

14 We have also explored the occupational background and current field of work of the civil servants (see Online Appendix Table A) as moderators of WTP. Different background and current-field categories can and do overlap in our data, so we strive to maximize statistical power by focusing on modal category combinations. This reveals no evidence of heterogeneity: WTP is not significantly different among civil servants that are trained scientists but lack expertise in other fields (the single most common background), nor among those working with environmental *and* water issues (the modal field-of-work combination).

15 This mirrors points raised in the literature on the role of payment vehicles in contingent valuation. It is well recognized that the payment vechicle used (e.g. voluntary donations, price changes, mandatory taxes, or tax reallocations that leave income and prices unaffected) matters for WTP (Bergstrom et al. 2004; Ivehammar 2009; Nunes and Travisi 2009). Note that, as in our choice experiment, payment vehicle and policy instrument are not the same thing. For instance, an income tax (payment vehicle) may be used to raise funds for an abatement subsidy (policy instrument).

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