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Using Ecosystem Service Values to Evaluate Tradeoffs in Coastal Hazard Adaptation

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ABSTRACT
The benefits of coastal adaptation depend on both the conservation of coastal assets and effects on other ecosystem services. Evaluating these benefits requires approaches that can disentangle values related to the assets that are conserved and the methods through which conservation is achieved. This article illustrates paired theoretical and empirical models designed to quantify values related to the methods and outcomes of coastal adaptation. Particular attention is given to valuation challenges associated with dual outcomes that influence human welfare both directly and indirectly. An illustrative empirical application is drawn from a stated preference, discrete choice experiment implemented in the coastal communities of Waterford and Old Saybrook, Connecticut, United States, grounded in storm and flooding scenarios developed for the Coastal Resilience decision-support platform. Results enable estimation of households' willingness to pay for outcomes such as the reduction of flood risk for coastal homes and the protection of services from coastal marshes and beaches. These estimates enable the evaluation of tradeoffs in social value related to the use of alternative adaptation strategies. Comparison across communities illustrates how differences in context can lead to variations in values and tradeoffs.

KEYWORDS
adaptation; choice experiment; economics; ecosystem service; flood; stated preference; valuation; willingness to pay

Introduction

Climate-driven sea-level rise – combined with changing coastal morphology and projected increases in the frequency and intensity of storm-related floods – pose increasing threats to built and natural coastal assets (Ashton, Donnelly, and Evans 2008; Jevrejeva et al. 2008; Kirshen, Knee, and Ruth 2008; Luisetti et al. 2011; Moser, Williams, and Boesch 2012). Amidst these changes, natural and nature-based features (NNBF) such as tidal marshes are increasingly promoted as a component of adaptation that promotes resilience and social benefits (Barbier et al. 2011, 2013; Temmerman et al. 2013; Bridges et al. 2015). This is due in part to the capacity of NNBF to provide ecosystem services beyond those linked to coastal protection alone (Barbier et al. 2011).

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While progress has been made in understanding the ecosystem services provided by NNBF, gaps still remain that limit analyses of when, where, and how specific adaptation strategies are beneficial. All adaptation requires tradeoffs. The capacity of NNBF to reduce the risks to built assets depends on multiple factors, and reliable protection of some assets cannot be achieved through NNBF alone (Kirshen, Knee, and Ruth 2008; Temmerman et al. 2013; Bridges et al. 2015; Whelchel 2016). Hence, the ecosystem service benefits of NNBF may be at least partially offset by a reduced or uncertain capacity to ameliorate flood risks in some areas.[1] At the same time, the use of hardened structures for coastal defense can contribute to the loss of natural assets and the ecosystem services they provide. For example, marshes and beaches may be progressively lost as they are "squeezed" between hardened shorelines and rising sea levels (Pethick 1993; Klein, Nicholls, and Mimura 1999; Luisetti et al. 2011; Temmerman et al. 2013). Seawalls or other hard defenses may deflect wave energy, causing the erosion of unprotected shorelines nearby (Pilkey and Wright 1988; Hall and Pilkey 1991; Cheong et al. 2013). Groynes or breakwaters can also alter the sedimentary system in ways that compromise the flood-protection properties of natural habitats as well as their ability to adapt to sea level rise (Cooper and McKenna 2008; Temmerman et al. 2013; Whelchel and Beck 2016).

Such tradeoffs imply that the net economic benefits of coastal adaptation depend on benefits and costs related to (a) changes in the conservation or protection of coastal assets, and (b) changes in other ecosystem services affected directly or indirectly. Evaluating these benefits and costs requires approaches that can disentangle values related to the assets that are protected and the methods through which protection is achieved. The biophysical dynamics of coastal systems imply that unambiguous Pareto improvements (win–win scenarios) are rare; actions to increase outcomes valued by some beneficiaries (e.g. protection for built assets) often require tradeoffs (e.g. increased adaptation costs, diminished ecosystem services valued by the same or other beneficiaries). An ecosystem services framework provides a structure through which these tradeoffs can be evaluated. Such evaluations require attention to the development of conceptual models linking adaptation actions to outcomes to values, and to the identification of appropriate measures of final ecosystem goods and services (Fisher, Turner, and Morling 2009; Bateman et al. 2011; Johnston and Russell 2011; Boyd and Krupnick 2013). It also requires theory and methods sufficient to estimate well-defined measures of economic value.

This article illustrates paired theoretical and empirical models designed to quantify ecosystem service values related to the methods and outcomes of coastal adaptation. Particular attention is given to valuation challenges associated with dual outcomes that influence human welfare both directly and indirectly. An illustrative empirical application is drawn from a stated preference, discrete choice experiment implemented in the coastal communities of Waterford and Old Saybrook, Connecticut, United States, grounded in storm and flooding scenarios developed for the Coastal Resilience decision-support platform (Beck et al. 2013; Hoover and Whelchel 2015). Model results enable estimation of households' willingness to pay (WTP[2]) for disparate outcomes such as the reduction of flood risk for coastal homes and the protection of services from coastal marshes and beaches. These estimates enable the evaluation of tradeoffs in social value related to the use of alternative strategies for coastal adaptation. Comparison of results across communities illustrates how differences in adaptation context can lead to variations in both ecosystem service values and optimal adaptation tradeoffs.

Frameworks for ecosystem service valuation and coastal adaptation

There is a mature literature on frameworks and methods for ecosystem service valuation (e.g. Daily 1997; Millennium Ecosystem Assessment 2005; Brown, Bergstrom, and Loomis 2007; Fisher and Turner 2008; Fisher, Turner, and Morling 2009; US EPA 2009; Gómez-Baggethun et al. 2010; Holland, Sanchirico, and Johnston 2010; Bateman et al. 2011; Wainger and Mazzotta 2011; NESP 2016; Olander et al. 2015, 2017), including applications to coastal adaptation and flood defense (e.g. Birol et al. 2009; Luisetti et al. 2011; Yue and Swallow 2014; Johnston and Abdulrahman 2017). As economic valuation is inherently reductionist (conveying effects using a monetary metric), these analyses are most useful when interpreted in combination with input from other natural and social sciences. Caveats such as these aside, economic valuation provides a rigorous, internally consistent and widely accepted means to evaluate, and compare the benefits and costs of different coastal management outcomes using the same monetary metric.

Among the distinguishing aspects of ecosystem service valuation is attention to the ways in which an action or policy change propagates through an ecosystem to affect structure and function, the provision of ecosystem services, and social benefits. This requires identification of the multiple direct and indirect ways in which any ecological outcome might influence social welfare. The challenge of identifying and disentangling these effects can be exacerbated by the prevalence of "dual outcomes" that influence welfare both directly and indirectly through effects on other valued outcomes (Boyd and Krupnick 2013; Boyd et al. 2016). Both hardened defenses and NNBF may be valued *directly* (positively or negatively), for example due to aesthetic or nonuse values associated with the existence of the adaptation measure itself.[3] They may also be valued *indirectly* for effects on other coastal assets or ecosystem services. Causal relationships such as these can be characterized graphically using causal chains or mathematically using structural, utility-theoretic models of ecological production and human welfare (Bateman et al. 2011; Boyd and Krupnick 2013; Johnston et al. 2013, 2017b; Olander et al. 2015; NESP 2016). These models can help ensure that valuation (a) targets final rather than intermediate services,[4] (b) considers the full set of causal pathways through which decisions or actions influence human welfare, and (c) avoids double counting (Fisher, Turner, and Morling 2009; Johnston and Russell 2011; Boyd and Krupnick 2013; Johnston et al. 2013, 2017b).

Failure to consider the full set of direct and indirect pathways (or at least those with potentially important effects) can lead to incomplete estimates of social value (Schultz et al. 2012). Given the cost and difficulty of high-quality empirical valuation research, most ecosystem service valuation targets only a subset of the services that are potentially relevant to the public. However, systematic inattention to significant values affected directly or indirectly by changes in the same coastal conditions (e.g. new hardened defenses) can lead to incomplete conclusions regarding the effects of these changes on social welfare. Past applications of ecosystem services valuation to flood adaptation have often omitted potentially relevant direct or indirect effects, leading to an inability to evaluate potentially relevant tradeoffs.

For example, Luisetti et al. (2011) calculate WTP for recreation and amenity benefits of flood adaptation, but do not consider the possibility that changes in hardened defenses could be a direct source of (possibly negative) amenity value. Yue and Swallow (2014) value ecosystem services associated with different coastal defenses (e.g. seawalls vs. living shorelines), but do not consider differences in the extent and type of flood protection offered by these alternatives. Imamura et al. (2016) derive WTP estimates for changes in seawall height and selected ecosystem services, but do not consider potential changes in built asset protection. Other studies focus on tradeoffs related to flood risks (e.g. prevalence, depth) and costs, with little attention to other ecosystem services (e.g. Brouwer and Schaafsma 2013), or evaluate values for all flood defense outcomes as an indivisible whole (e.g. Bateman, Willis, and Garrod 1994).[5,6]

Although informative, studies such as these are unable to evaluate at least some potentially important tradeoffs involving flood protection and other benefits provided by adaptation. Of particular emphasis here is the ability to disentangle the direct and indirect welfare effects of hardened versus natural defense strategies. For example, would residents of a coastal community be willing to give up a certain amount of coastal flood protection (e.g. placing more built assets at risk) if this enables flood protection to be provided primarily via natural defenses with concomitant increases in other ecosystem services? Works in the current adaptation literature rarely enable such tradeoffs to be evaluated.

The current application is grounded in an ecosystem services framework that – although necessarily simplified – explicitly recognizes (a) that methods for flood defense may represent dual outcomes with both direct and indirect effects on social welfare, and (b) that there may be tradeoffs between the asset protection provided by adaptation methods and the provision of other ecosystem services. The goal of the analysis is to demonstrate the capacity of ecosystem services valuation to characterize the benefits and tradeoffs associated with coastal adaptation.

An illustrative application to coastal adaptation tradeoffs

We illustrate the model using a case study application in Waterford and Old Saybrook, Connecticut. Waterford is a coastal community of 19,517 residents (2010 US Census) with approximately 26 miles of tidal shoreline along Long Island Sound and adjoining rivers. Old Saybrook is a nearby community of 10,367 residents, with approximately 50 miles of tidal shoreline (Makriyannis 2017). Primary adaptation concerns for these communities include flood risks facing built assets such as homes, along with the resilience of natural assets such as beaches and coastal marshes (Pardo and Whelchel 2013a, 2013b; Town of Old Saybrook 2015; Whelchel and Ryan 2015). Although located in the same area of the state, these two communities have different risk profiles, with built assets in Old Saybrook more vulnerable to flooding. The communities also have different endowments of natural assets such as coastal marshes.[7]

Economic values are quantified using households' WTP for changes in ecosystem services and other adaptation outcomes, estimated using a stated preference choice experiment. Stated preference methods are survey-based approaches for nonmarket valuation, enabling the estimation of both use and nonuse values (Johnston et al. 2017a). A choice experiment questionnaire asks respondents to choose among a set of hypothetical but realistic policy options, similar to a public referendum with two or more choice options. Each option is described by multiple attributes, often including indicators of ecosystem service changes and the monetary cost to the household required to implement each option. Data consisting of choices over many sets of multi-attribute options enables WTP estimation (Adamowicz et al. 1998; Bateman et al. 2002).

Theoretical model

The choice experiment is grounded in a random utility framework in which the utility of household n is determined by the choice of a multi-attribute adaptation plan from a set of j alternatives ($j = A, B, N$). These include two adaptation options (A, B), and a status quo option (N) with no adaptation and zero cost. The household's utility, $U_{nj}(\cdot)$, is split into a deterministic and stochastic component. The deterministic component, $V_{nj}(\cdot)$, includes observable attributes X_{nj} and C_{nj}, where X_{nj} is a vector of final outcomes and methods of adaptation that influence utility directly, and C_{nj} is monetary cost. The utility function is given by

$$U_{nj}(\cdot) = U_{nj}(X_{nj}, C_{nj}) = V_{nj}(X_{nj}, C_{nj}) + \varepsilon_{nj}, \tag{1}$$

where ε_{nj} represents the stochastic component of utility, modeled as a random error. Equation (1) is estimated using an additively separable, linear-in-the-parameters function

$$U_{nj}(X_{nj}, C_{nj}) = \beta_1 X_{nj} + \beta_2 C_{nj} + \varepsilon_{nj} \tag{2}$$

where β_1 is a conforming vector of parameters on utility-relevant adaptation outcomes and methods, and β_2 is the parameter on household cost. When making a choice between policy alternatives ($j = \text{A, B, N}$) the household is assumed to choose the alternative that provides the greatest anticipated utility. Utility parameters are estimated using maximum likelihood models for discrete dependent variables (here, mixed logit), allowing for preference heterogeneity among respondents (Train 2009).

A model of this type can accommodate biophysical causality by specifying X_{nj} as a biophysical function of another vector of intermediate outcomes, $X_{nj} = f(Z_{nj})$, where Z_{nj} are outcomes that influence utility only through causal influences on final outcomes X_{nj} (Johnston et al. 2013, 2017b). As detailed below, the associated choice experiment scenarios need only present information on the final arguments that affect utility directly (X_{nj} and C_{nj}), including dual outcomes that affect utility both directly and indirectly.

Attribute selection and choice experiment design

Grounded in this theoretical structure, the choice experiment questionnaire was developed over two years in a process involving collaborative efforts of economists and natural scientists; meetings with town planners, engineers and stakeholder groups; and 13 focus groups with community residents. Focus group participants were recruited by a marketing firm using random sampling over phone listings for the two communities, with respondents paid to participate. Focus groups were implemented (design, moderation, and analysis) using ethnographic focus group methods for stated preference design (Johnston et al. 1995). The data used to inform choice scenarios were obtained from sources including Columbia University's Center for Climate Systems Research, NASA's Goddard Institute for Space Studies, The Nature Conservancy (TNC), and the National Oceanic and Atmospheric Administration (NOAA), as reflected in coastal flooding scenarios for TNC's Coastal Resilience decision-support platform (www.coastal-resilience.org). The specification of X_{nj} (i.e. attributes in scenarios) was grounded in a conceptual model combining input from focus groups; scientists with expertise in sea level rise and coastal resilience; coastal flooding scenarios for each community; and municipal officials and stakeholders. This conceptual model specified the linkages between adaptation outcomes and methods, and identified the outcomes and methods with the most salient (potential) final effects on utility.

As shown by Johnston et al. (2013), unbiased WTP estimation requires that choice scenarios provide information on all affected direct (or final) utility arguments (X_{nj}), regardless of whether any of these have causal or dual effects on other direct utility arguments. Purely intermediate outcomes (Z_{nj}) are excluded (i.e. respondents do not require information on intermediate outcomes Z_{nj} to make choices that reveal their value for adaptation outcomes). Scenarios designed in this way provide all information required for respondents to correctly evaluate anticipated utility change and make well-informed choices. The resulting WTP estimates for all final outcomes (including dual outcomes) are interpreted as the value of changes in these outcomes alone, *holding other outcomes constant*. This provides a clean estimate of the direct value of each outcome, holding all indirect (causal) effects constant, and thereby allows WTP for coastal adaptation methods and outcomes to be disentangled (Johnston et al. 2013).

Consider a stylized example of flood protection provided by coastal marshes, assuming that both marsh extent (e.g. acres of marsh remaining) and flood protection (e.g. homes expected to flood in a typical storm scenario) are valued directly by the public. Here, marsh extent is a dual outcome, because marsh acres are valued directly (e.g. for aesthetic properties), as well as indirectly because marshes prevent home flooding. A choice experiment scenario for this example would present respondents with information characterizing final effects on both wetland acres and expected home flooding. Once information on these final (and dual) outcomes is provided, there is no need to provide additional functional information on the biophysical effectiveness of wetlands at preventing flooding – the relevant information is already embedded in the presented information on wetland extent and homes expected to flood.[8]

Grounded in this theoretical structure, among the most important choices facing choice experiment design is the set of attributes that will be used to communicate changes in final ecosystem services (Zhao, Johnston, and Schultz 2013; Boyd et al. 2016). Respondents' cognitive limitations restrict the number of attributes that can be included in scenarios (DeShazo and Fermo 2002). Hence, attributes are limited to those with the greatest potential salience for respondents' welfare, based on the decision-making context. Here, these include the set of utility-relevant adaptation outcomes and methods (including ecosystem service impacts) that both (a) are of greatest potential importance to residents, as reflected by focus group responses in each community, and (b) would vary significantly across adaptation alternatives.

In the present case study, the primary ecosystem service attributes meeting criteria (a) and (b) were those directly related to the existence and quantity of beaches/dunes and coastal marshes in the two communities. Focus group results suggested that beach acres are valued directly due to the recreation and aesthetic benefits they provide. Hence, the number of beach acres remaining was selected as the most relevant indicator of these services. Ecosystem services from coastal marshes were similarly quantified in the choice experiment using the number of marsh acres. This reflects a situation in which the most salient ecosystem services from coastal marshes to residents – as identified by focus groups – are aesthetic and nonuse services related to the quantity of marsh remaining. Although focus group participants recognized that marshes provide other services such as fishery production, these were not a primary motivation for their choices over adaptation alternatives, nor would these related services vary significantly and predictably across adaptation alternatives.[9] Given this emphasis on existence and aesthetic services, the most suitable indicator was judged to be the quantity of marsh remaining.

The choice experiment design process summarized above led to a set of six final and dual attributes, including indicators of ecosystem services, natural/built assets, and adaptation methods with direct influences on utility. These included: (1) the percentage and number of homes expected to flood in a Category 3 storm, (2) marsh acreage lost, (3) beach and dune acreage lost, (4) the length of coastline that would be hard-armored, (5) the general emphasis of adaptation efforts (whether there would be additional emphasis on hardened coastal defenses), and (6) unavoidable household cost.[10] All outcomes are forecast as of the mid 2020s. Following the general approach of Johnston et al. (2012), each attribute is presented in relative (percentage) terms with regard to upper and lower reference conditions (i.e. best and worst possible). Scenarios also present the cardinal basis for relative levels where applicable.[11]

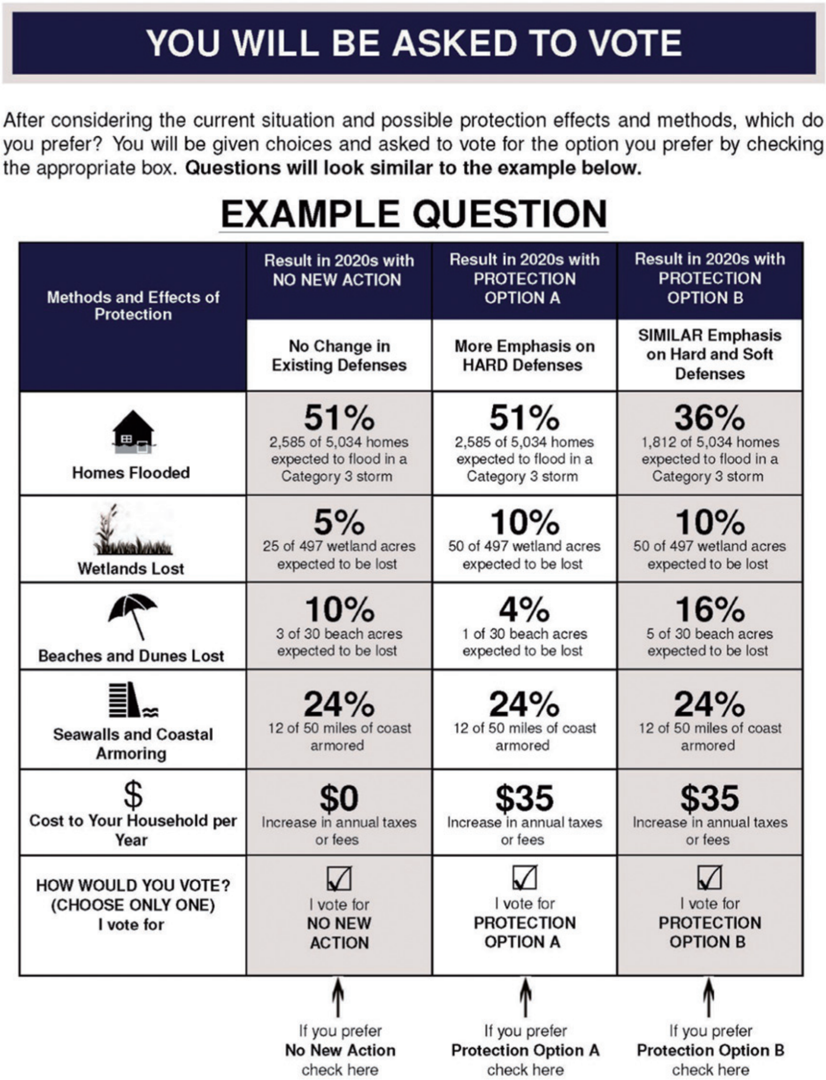
Table 1 provides summary statistics, definitions, and possible levels for each attribute. These levels were chosen based on feasible adaptation outcomes for each community, identified using data sources discussed above. Based on these attribute levels, a fractional factorial experimental design was generated using a D-efficiency criterion (Ferrini and Scarpa 2007; Sándor and Wedel 2001, 2002; Scarpa and Rose 2008) for main effects and two-way interactions, yielding 72 profiles blocked in 24 booklets. Each respondent was provided with three choice questions and was instructed to consider each as independent and nonadditive. A sample question is shown in Figure 1. Prior to administration of choice questions, the questionnaire described tradeoffs associated with alternative approaches to coastal adaptation and illustrated projected inundation scenarios with no new adaptation actions. This and other information was conveyed via a combination of text, graphics including geographic information system (GIS) maps and photographs. Detailed instructions were also provided. Survey language and graphics were subject to extensive pretesting in focus groups (Johnston et al. 1995).

The choice experiment was implemented during May–June 2014 over a random sample of Old Saybrook and Waterford households. The questionnaire was distributed via

Table 1. Attribute definitions and levels.

Variable	Definition	Attribute levels – Old Saybrook	Attribute levels – Waterford
Homes	Number of homes expected to flood in a Category 3 storm in the mid 2020s; presented as a percentage of the total number of homes in each community (Range 0–100%).	36% (1,812 out of 5,034 homes expected to flood) 43% (2,165 out of 5,034 homes expected to flood) 51% (2,585 out of 5,034 homes expected to flood)[a] 59% (2,970 out of 5,034 homes expected to flood)	2% (169 out of 8,460 homes expected to flood) 4% (338 out of 8,460 homes expected to flood) 7% (566 out of 8,460 homes expected to flood)[a] 10% (846 out of 8,460 homes expected to flood)
Wetlands	Number of acres of wetlands expected to be lost by the mid 2020s due to flooding or erosion; presented as a percentage of current coastal marsh acres in each community (Range 0–100%).	2% (10 out of 497 acres expected to be lost) 5% (25 out of 497 acres expected to be lost)[a] 10% (50 out of 497 acres expected to be lost)	5% (4 out of 77 acres expected to be lost) 12% (9 out of 77 acres expected to be lost)[a] 19% (15 out of 77 acres expected to be lost)
Beaches	Number of acres of beaches and dunes expected to be lost by the mid 2020s due to flooding or erosion; presented as a percentage of current beach and dune acres in each community (Range 0–100%).	4% (1 out of 30 acres expected to be lost) 10% (3 out of 30 acres expected to be lost)[a] 16% (5 out of 30 acres expected to be lost)	4% (1 out of 36 acres expected to be lost) 10% (4 out of 36 acres expected to be lost)[a] 16% (6 out of 36 acres expected to be lost)
Seawalls	Miles of the town coast shielded by hard defenses by the mid 2020s; presented as a percentage of the total miles of the town coastline (Range 0–100%).	15% (8 out of 50 miles) 24% (12 out of 50 miles)[a] 35% (18 out of 50 miles)	40% (10 out of 26 miles) 50% (13 out of 26 miles)[a] 60% (16 out of 26 miles)
Hard	Binary (dummy) variable indicating whether the protection plan places more emphasis on hard defenses relative to the status quo.	0 (no emphasis on hard defenses)[a] 1 (emphasis on hard defenses)	0 (no emphasis on hard defenses)[a] 1 (emphasis on hard defenses)
Neither	Alternative specific constant (ASC) that takes a value of 1 for the status quo option (a choice of neither adaptation plan), and 0 otherwise.	0 (status quo)[a] 1 (not status quo)	0 (status quo)[a] 1 (not status quo)
Cost	Household annual cost, presented as unavoidable increase in taxes and fees required to implement the coastal protection plan. A choice of neither protection plan is associated with zero cost (Range $0–$155).	$0 (cost to household per year)[a] $35 (cost to household per year) $65 (cost to household per year) $95 (cost to household per year) $125 (cost to household per year) $155 (cost to household per year)	$0 (cost to household per year)[a] $35 (cost to household per year) $65 (cost to household per year) $95 (cost to household per year) $125 (cost to household per year) $155 (cost to household per year)

[a]Status quo value.



YOU WILL BE ASKED TO VOTE

After considering the current situation and possible protection effects and methods, which do you prefer? You will be given choices and asked to vote for the option you prefer by checking the appropriate box. **Questions will look similar to the example below.**

EXAMPLE QUESTION

Methods and Effects of Protection	Result in 2020s with NO NEW ACTION	Result in 2020s with PROTECTION OPTION A	Result in 2020s with PROTECTION OPTION B
	No Change in Existing Defenses	More Emphasis on HARD Defenses	SIMILAR Emphasis on Hard and Soft Defenses
Homes Flooded	51% 2,585 of 5,034 homes expected to flood in a Category 3 storm	51% 2,585 of 5,034 homes expected to flood in a Category 3 storm	36% 1,812 of 5,034 homes expected to flood in a Category 3 storm
Wetlands Lost	5% 25 of 497 wetland acres expected to be lost	10% 50 of 497 wetland acres expected to be lost	10% 50 of 497 wetland acres expected to be lost
Beaches and Dunes Lost	10% 3 of 30 beach acres expected to be lost	4% 1 of 30 beach acres expected to be lost	16% 5 of 30 beach acres expected to be lost
Seawalls and Coastal Armoring	24% 12 of 50 miles of coast armored	24% 12 of 50 miles of coast armored	24% 12 of 50 miles of coast armored
$ Cost to Your Household per Year	$0 Increase in annual taxes or fees	$35 Increase in annual taxes or fees	$35 Increase in annual taxes or fees
HOW WOULD YOU VOTE? (CHOOSE ONLY ONE) I vote for	☑ I vote for NO NEW ACTION	☑ I vote for PROTECTION OPTION A	☑ I vote for PROTECTION OPTION B

If you prefer **No New Action** check here

If you prefer **Protection Option A** check here

If you prefer **Protection Option B** check here

Figure 1. Example choice question from Old Saybrook questionnaire.

U.S. mail, with follow-up mailings to increase response rates (Dillman, Smyth, and Christian 2009). Of 2513 deliverable questionnaires, 808 were returned for a response rate of 32.2%.[12]

Model and value estimation

Independent random-utility models are estimated for each community. Models are estimated in WTP-space (Train and Weeks 2005; Scarpa, Thiene, and Train 2008), so that coefficients represent direct estimates of annual per household WTP (implicit prices). Coefficients on *neither*, *beaches*, and *homes* are specified as random with independent normal distributions to allow for WTP heterogeneity across households. Other attributes are assumed to have nonrandom coefficients.[13] We assume an underlying lognormally

Table 2. WTP-space mixed logit results: Waterford and Old Saybrook.

Attribute	Waterford Coefficient (SE)	Old Saybrook Coefficient (SE)
Hard	–5.368	–69.979**
	(22.705)	(29.750)
Wetlands	–9.646***	–6.372**
	(2.231)	(2.918)
Beaches	–6.779***	–5.606**
	(2.591)	(2.297)
Seawalls	2.095	–0.351
	(1.651)	(1.974)
Neither	–175.822***	–160.130***
	(53.574)	(56.712)
Homes	–1.289	–9.333***
	(4.786)	(2.703)
	Estimated Std. Dev	**Estimated Std. Dev**
Std_neither	392.497 (104.410)***	327.382 (141.467)**
Std_homes	21.920 (8.023)***	11.198 (3.557)***
Std_beaches	7.162 (7.141)	10.459 (3.775)***
Number of observations	407	408
-2LnL χ^2 (prob $> \chi^2$)	183.84 (11 *df*) / 0.0001	186.657 (11 *df*) / 0.0001
Pseudo – R^2	0.206	0.208
WTP per cardinal unit		
Wetlands (per acre)	$12.53***	$1.28**
Beaches (per acre)	$18.83***	$18.69**
Seawalls (per mile)	–$8.06	$0.70
Homes (per home)	$0.02	$0.19***

*$p < .10$, **$p < .05$, ***$p < .01$.

distributed cost coefficient to ensure a positive marginal utility of income. The model is estimated using simulated likelihood mixed logit with 500 mixed Halton and pseudo-random draws. Alternative specifications were tested to assess robustness and convergence; these suggest that the presented results are robust.

Results

Results are shown in Table 2. Both models are significant at $p < .0001$, with pseudo-R^2 estimates of 0.206 and 0.208 for Waterford and Old Saybrook. Signs of estimated coefficients match prior expectations, where expectations exist.[14] For attributes on continuous variables (*homes, beaches, wetlands, seawalls*), coefficients reflect mean annual implicit prices (marginal WTP) per percentage point increase in each attribute, along with the estimated standard deviation (SD) of these implicit prices (for random parameters). For example, an estimated coefficient of –9.333 for *homes* in Old Saybrook implies that the average sampled household in Old Saybrook is willing to pay $9.33 annually (in increased taxes and fees) to prevent the expected flooding of 50.34 homes (1% of the total homes in the community) during typical Category 3 hurricanes occurring in the mid 2020s, *ceteris paribus*. For the two binary attributes (*hard*, *neither*), the estimates reflect mean per household WTP for the presence of the attribute relative to its absence. For example, according to survey responses, an average Old Saybrook household would be willing to pay $69.98 to *avoid* an adaptation plan that places additional emphasis on hardened shoreline.[15]

The estimates discussed above reflect mean WTP across the sample. Estimated WTP SD for the random parameters *neither*, *beaches*, and *homes* (Table 2) characterize heterogeneity – whether WTP differs across sampled households. Results suggest that values for adaptation outcomes are often, but not always heterogeneous. For example, the SD of 21.920 for *homes* in Waterford is statistically significant at $p < .01$, implying that WTP for home protection varies to a statistically significant degree across households. This result implies that some Waterford households value home protection highly, even though mean WTP for home protection across the entire community is not statistically different from zero. Similar heterogeneity is found for other attributes (such as *beaches* in Old Saybrook).

The final three rows of Table 2 transform mean WTP estimates for continuous attributes into values per cardinal unit. For example, responses imply that the average Waterford household would be willing to pay $6.78 to prevent the loss of 1% of the town's current beaches, which corresponds to $18.83 per beach acre protected. As noted above, these WTP estimates are interpreted as the monetary value of changes in each attribute alone, apart from any causally related effects on other attributes. This independence facilitates the evaluation of tradeoffs associated with different adaptation methods and outcomes. Consider a stylized example in which Old Saybrook restores one acre of marsh in such a way that 20 fewer homes are expected to flood in a typical Category 3 storm. Results imply that the average Old Saybrook household would be willing to pay $5.08 ($\$1.28 + 20 \times \$0.19$) per year for these combined outcomes, accounting for all causal relationships between them.

Implications for coastal adaptation values and tradeoffs

Model results provide insight into the effect of adaptation tradeoffs on ecosystem service and other values realized by community residents, and the extent to which different adaptation methods and outcomes are valued in each community. As noted above, results also enable estimation of the extent to which residents would be willing to tradeoff different types of adaptation outcomes – for example the protection of homes versus ecosystem services of various types. Finally, they reveal differences in coastal adaptation values across the two communities.

Residents of both Old Saybrook and Waterford hold statistically significant WTP for the protection of *wetlands* and *beaches*, reflecting the direct ecosystem service value of these assets (Table 2). Beach protection provides similar value across the two communities ($18.83 vs. $18.69 per household, per acre in Waterford and Old Saybrook). In contrast, protection of marshes is valued more highly in Waterford ($12.53 vs. $1.28 per household, per acre in Waterford and Old Saybrook). This variation comports with theory-based expectations, given that coastal marshes are scarcer in Waterford, and are hence more valued on a per acre basis.

Unlike *beaches* and *wetlands* (which are valued by residents of both communities), the protection of *homes* is only associated with statistically significant mean WTP in Old Saybrook. Model results indicate that average Waterford residents place no statistically significant value on public actions that protect additional community homes from flooding. Even in Old Saybrook, per unit WTP for home protection (*homes*) is relatively low compared to that for *wetlands* and *beaches*. For example, the WTP for *homes* versus *beaches* among Old Saybrook residents implies that protection of an additional ∼101 homes (i.e. 18.69/0.19) would be required to offset the value sacrificed due to the loss of a single acre of beach in the community.

The relatively low value of home protection might seem surprising given the attention given to the protection of built assets by the literature (e.g. Kirshen, Knee, and Ruth 2008; Barbier et al. 2013; Bridges et al. 2015). However, these values are intuitive when viewed from the perspective of community-wide values, and public versus private goods. Recall, the choice experiment estimates the WTP of average residents for community-wide outcomes. Although residents may hold high values for the protection of their *own homes* from flooding, a large proportion of residents do not live in homes that are at risk of flooding. Moreover, focus group results suggested that residents often view the protection of private homes as the responsibility of homeowners – not a valued outcome for which (not-at-risk) residents are willing to pay. That is, typical residents hold low values for the protection of *other people's homes* – leading to low average WTP for home protection community-wide. These residents make tradeoffs suggesting that greater value is provided by public actions that protect ecosystem services.

Results also suggest that residents hold no statistically significant, direct WTP for marginal changes in hardened shoreline (*seawalls*) alone. Hence, the only value associated with changes in *seawalls* would be related to indirect effects on other assets and ecosystem services. This result, however, must be interpreted within the context of other model results. For example, Old Saybrook results suggest a relatively large negative value associated with community-wide adaptation plans that emphasize the use of hardened shoreline, *ceteris paribus*. Hence, while additional miles of seawalls are value-neutral in isolation, large-scale hardening of the shoreline would be associated with negative WTP in Old Saybrook. Similarly, any long-term loss of coastal habitats (*beaches* or *wetlands*) caused by the construction of hardened shoreline would cause a statistically significant loss of economic value to residents. Results such as these demonstrate the capacity of approaches such as this to decouple direct values associated with adaptation methods and indirect values associated with the resulting adaptation outcomes.

Illustrative adaptation scenarios

Results such as these may be linked to biophysical adaptation scenarios to project combined implications for adaptation benefits. These scenarios may either be assumed or generated via biophysical simulations. For example, consider a choice between two illustrative coastal adaptation scenarios in Waterford, both of which would apply to exposure zone C (Figure 2). Plan 1 would prevent the expected flooding of 50 homes during a Category 3 hurricane using NNBF, such that two acres of coastal marsh is added to the community. Plan 2 would prevent expected flooding of 150 homes, but would do so via the construction of one mile of hardened shoreline, with an attendant loss of one acre of coastal marsh. Based on the point estimates of value in Table 2, each community household would be willing to pay an average of $25.82 per year to obtain the adaptation outcomes in Plan 1. Assuming an approximately representative sample (Johnston and Abdulrahman 2017), this aggregates to a total of $194,734 per year, over all 7,542 Waterford households (as of the 2010 US Census).[16]

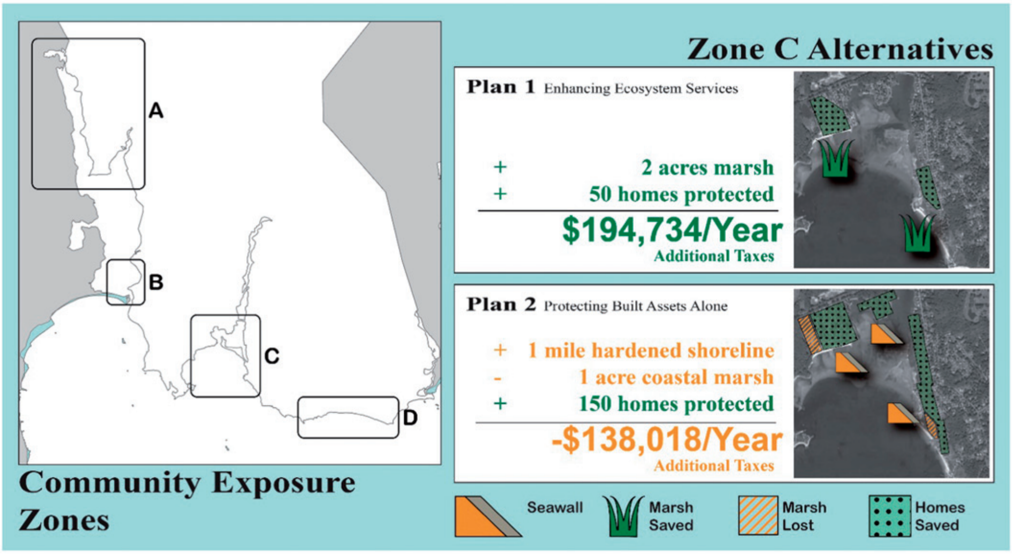


Figure 2. Biophysical adaptation and economic WTP tradeoffs between two illustrative coastal adaptation scenarios for an exposure zone in Waterford, Connecticut.

In contrast, each household would be willing to pay an average of *negative* $18.30 per year for Plan 2, or negative $138,018 per year aggregated over all community households. That is, average residents would *have to be paid* in order to voluntarily accept Plan 2, because the negative aspects of the plan outweigh the positive aspects. This negative value persists despite the greater protection of homes within Plan 2. Results such as these demonstrate the community-wide economic losses that can be associated with adaptation plans that sacrifice ecosystem services in order to protect built assets alone, and show the importance of considering both direct and indirect effects when evaluating adaptation benefits. Note that these illustrative estimates represent the (positive or negative) benefits of adaptation – they do not include the costs of adaptation. Evaluation of adaptation costs is beyond the scope of the present analysis.

Conclusion

Coastal adaptation can influence many different types of economic values realized by residents and nonresidents of coastal communities. Moreover, adaptation may require tradeoffs between the protection of built and natural assets. Economic valuation provides a means to reconcile and compare these "apples to oranges" tradeoffs in terms of their effects on social welfare, enabling identification of adaptation strategies that provide the greatest benefit to the public.

This article illustrates the use of economic choice experiments to estimate the WTP of households in two communities for coastal adaptation outcomes. The results can be used to characterize effects on social benefits that would occur when making adaptation tradeoffs. For example, results can be used to evaluate whether society would benefit from actions that provide additional protection for homes via hardened infrastructure, with a concomitant loss of ecosystem services (Figure 2). Results suggest that such strategies can reduce social welfare, even before considering the direct costs of hardened infrastructure (i.e. the benefits alone are negative). More generally, results demonstrate that the protection of built assets may have lower value to residents than is typically assumed, whereas the protection of ecosystem services is valued more highly. Comparison of results across the two communities further illustrates that "one size does not fit all" when considering coastal adaptation – the same actions that increase benefits (i.e. aggregate WTP) in one community may diminish benefits in other communities.

From a methodological perspective, results such as these highlight the importance of disentangling values for the methods and outcomes of adaptation. For example, results demonstrate that marginal changes in hardened shoreline have no significant *direct* impact on residents' welfare, but may have large *indirect* effects – for example if additional hardened shoreline causes attendant losses in valued ecosystem services. The capacity to disentangle direct and indirect utility effects in this way requires that choice experiment design be grounded in a formal, utility theoretic structure for ecosystem service valuation.

All empirical results pertain to our case study and might differ across contexts. Despite substantial attention to survey design, it is also possible that some valued adaptation outcomes might have been omitted – leading to incomplete value estimates. Given the many market and nonmarket economic benefits associated with ocean and coastal resources, no single valuation method can typically measure all aspects of value (Johnston et al. 2002). As a result, economic analysis often combines results from different valuation methods. These and other caveats aside, the present application illustrates ways that ecosystem service valuation may be used to provide otherwise unavailable insight into the effect of coastal adaptation tradeoffs on social welfare.

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Notes

1. As noted by Bridges et al. (2015, 239), "there are numerous uncertainties regarding the performance, timing, and scale of NNBF needed to provide flood risk reduction and decrease storm damages. NNBF are typically more responsive to storms, and the risk reduction services provided often depend on local conditions." Urban areas, for example, may sometimes require hardened defenses to provide flood protection (Temmerman et al. 2013).
2. WTP is a commonly used measure of monetary value in economics, reflecting the maximum amount of money that an individual or group would be willing to give up in exchange for more of something else. WTP is bounded by income (or ability-to-pay), and hence is conditional on the current distribution of income. This has led to critiques related to the equity implications of welfare analysis. Hence, "while economic welfare analysis should be viewed as an integral step in policy formulation and evaluation, it is not the only component'" (Just, Hueth, and Schmitz 2004, 11). Other issues, such as equity and social justice, are relevant. In some cases one can also estimate willingness to accept (WTA) measures of welfare, which are not bounded by income (Freeman, Herriges, and Kling 2014).

3. For example, residents may hold positive values for the aesthetic properties of nearby coastal marshes (Johnston et al. 2005), or negative values for the aesthetic properties of hardened defenses such as concrete seawalls.
4. Intermediate services are ecological conditions or processes that only benefit humans through effects on other final services. They hence may be viewed as inputs into the production of final services. As such, all social value related to intermediate services is derived through the production of final services (Johnston et al. 2017b).
5. There are also a large number of studies that estimate values associated with a single economic aspect of flood risk or adaptation, for example the effect of flood risk on property values (e.g. Bin and Polasky 2004; Troy and Romm 2004; Bin et al. 2008; Daniel, Florax, and Rietveld 2009).
6. The assumed baselines for valuation also influence the relevance of value estimates for adaptation decisions. For example, multiple studies have evaluated the flood attenuation services provided by marshes, generally relative to a baseline in which these NNBF are not present (e.g. Barbier et al. 2011; Shepard, Crain, and Beck 2011). However, a more relevant basis for comparison in many instances is an alternative in which hardened defenses are applied.
7. For example, land cover data in Coastal Resilience indicates that Old Saybrook has approximately 477 acres of coastal marsh remaining today, whereas Waterford has approximately 77 acres remaining (Makriyannis 2017).
8. Considering another example, a WTP estimate for an attribute on seawall length would reflect the direct WTP for aesthetic and other immediate properties of seawalls alone, apart from any value related to the causal, indirect effects of seawalls on utility (e.g. due to coastal asset protection, effects on natural habitats, etc.). WTP for these other, causally related outcomes would be captured by other direct outcomes included in the choice experiment scenario (e.g. attributes reflecting effects on valued habitats and protection of built assets from flooding).
9. For example, marginal changes in salt marsh acreage within Waterford – of the type likely given changes in current adaptation planning – are unlikely to cause measurable change in neighboring Long Island Sound water quality or fish populations, despite the fact that wetlands provide fish production and water filtration services.
10. As discussed above, outcomes with purely intermediate effects on welfare are excluded. For example, scenarios do not include an attribute on residential zoning regulations in flood-prone areas, because the primary way that these regulations affect welfare (for most residents) is through an intermediate effect on homes expected to flood.
11. For example, the attribute representing the number of homes expected to flood in a Category 3 storm (*homes*) is presented both as a cardinal number and as a percentage relative to the total number of homes in each town.
12. An analysis of sample representativeness is provided by Johnston and Abdulrahman (2017).
13. This assumption is required to ensure model convergence.
14. For example, focus group results and prior research by the authors (e.g. Johnston et al. 2005; Johnston, Ramachandran, and Parsons 2015) suggest that the public values beach and marsh protection, corresponding to negative coefficients for beach and marsh loss.
15. Because these estimates reflect ongoing annual WTP they are not discounted values. Assumptions regarding the discount rate are required when aggregating such estimates over time (Johnston et al. 2017a). Egan, Corrigan, and Dwyer (2015) discuss the role of discounting in the design if stated preference scenarios.
16. Following Adamowicz et al. (1998), this estimate does *not* include WTP associated with the alternative specific constant (*neither*). If included, this would result in an additional WTP of $175.82 per household, per year, associated with any nonstatus quo adaptation plan.

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