Impacts of Terrestrial and Shoreline Stressors on Eelgrass in Puget Sound: An Expert Elicitation

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We used expert elicitation to examine potential responses of eelgrass to several restoration strategies in Puget Sound. Restoration strategies included shoreline armor removal and modification, removal and modification of overwater structures, and efforts to improve water clarity via reductions in anthropogenic nutrient and sediment loadings. Expert responses indicated a general belief that reducing stressors would increase eelgrass cover; however, responses varied greatly among stressors. Our analyses revealed that removal of overwater structures, nutrient loading and shoreline armor will have significantly larger effects on eelgrass recovery than would removal of sediment loading, with removal of overwater structures having the largest effect. We then used a probabilistic model to estimate what actions, singularly or in combination, could yield a large increase in eelgrass cover. Reducing single stressors could, in theory, result in recovery of eelgrass in Puget Sound; however, the magnitude of actions required would be so great that it is likely not practical. In contrast, we identified combinations of smaller reductions of stressors that could achieve significant eelgrass recovery. For example, a 40% reduction in overwater structures, combined with 20% reductions in shoreline armor, and nutrient and sediment loadings, was predicted to be one of the more feasible combinations of actions for meeting the target. The importance of eelgrass to Puget requires prompt input of scientific advice, and this work fills an important knowledge gap in the face of rapidly approaching legislative deadlines. While coded expert opinion of the sort we use here is a weak substitute for data, our work clarifies the current extent of scientific uncertainty that can guide management action in the near term and scientific research in the long term.

Keywords eelgrass, expert elicitation probabilistic model, recovery target, Zostera marina

Introduction

The central importance of habitat extent and quality in marine conservation and resource management is widely recognized (Mangel, Levin, and Patil 2006) and is codified in
such seminal legislation as the U.S. Endangered Species Act (ESA) and the Sustainable Fisheries Act (SFA). A great deal of effort has been directed toward understanding what habitats might be “essential” or “critical” for a range of species of management concern (Levin and Stunz 2005; Able 1999). Indeed, setting aside habitat in reserves or easements, restoring habitat in locations where it once existed, and creating new habitat in areas where it never existed are now standard instruments in marine conservation.

In many locations, research has highlighted the underlying causes of habitat degradation or loss. As examples: (1) On a global scale, degradation of coral reefs has occurred as a result of overfishing, nutrient and sediment inputs, and ocean warming and acidification (Pandolfi et al. 2003); (2) subsidence of salt marshes along the Atlantic Coast of North America has been caused by diking to establish permanent water supplies and waterways, thereby restricting natural seawater flow (Portnoy and Giblin 1997; Teal and Howes 2002); (3) degradation of kelp forests has occurred due to direct and indirect effects of overfishing, kelp harvesting, pollution, invasive species, and climate change (Steneck et al. 2002).

Identifying critical habitats and their threats is necessary but not sufficient for the restoration of habitats and their ecological function. Effective restoration also requires an understanding of the types of management actions that can be taken and the required magnitude of those actions. Understanding the functional relationship between stressors and habitat quality or quantity gives us the ability to predict ecosystem responses, provides a road map for how to reach desired outcomes, and, in some cases, allows us to offset the types and magnitudes of habitat degradation that have previously occurred (Duarte 2002).

Puget Sound, Washington, USA, is among the most productive and biologically diverse habitats in North America (Ricketts et al. 1999), but has also suffered extensive degradation of nearshore habitats and processes (Ruckelshaus and McClure 2007) (Figure 1). Eelgrass (Zostera marina) in Puget Sound currently covers ca. 21,500 hectares (200 km²) (Dowty, Berry, and Gaeckle 2010). Although long term eelgrass change estimates are hampered by limitations in the historical record, total losses have been documented in several major embayments, and since 2000 eelgrass cover has decreased in more than twice as many continuously monitored sites than it has increased (Thom and Hallum 1990; Gaeckle et al. 2011). The structure created by eelgrass provides spawning substrate, grazing opportunities and predation refugia for a number of species, including economically important species such as Pacific salmon, Dungeness crab, and Pacific herring (Mumford 2006). It may also be a significant contributor to food webs, via detritus formed when eelgrass blades slough off (McConnaughey and McRoy 1979; Simenstad and Wissmar 1985).

Eelgrass is at risk from any activity that reduces light or disturbs the sediment where it grows. The link between water quality and the cover and distribution of eelgrass beds is well established (Thom et al. 2003; Kentula and DeWitt 2003; Hauxwell et al. 2001). For example, macroalgal blooms associated with nutrient loading reduce light penetration and have been linked to seagrass declines (Hauxwell and Valiela 2004). Additionally, overwater structures, such as marinas, piers, docks, and floats shade benthic habitats, reducing plant density, vigor, and leaf size (Fresh et al. 2006; Nightingale and Simenstad 2001). Under certain conditions, shoreline armoring may also hinder eelgrass beds by reducing or eliminating the supply of sediments from feeder bluffs, thereby reducing appropriate habitat for seagrass (Stevens and Lacy 2012; de Boer 2007; Simenstad et al. 2008). In Puget Sound alone, approximately 27% of 2500 shoreline miles have been armored (defined as shoreline that has been “hardened” in a variety of ways including, rip rap, bulkheads, and seawalls), while almost 9000 overwater structures (e.g., docks, piers)
Figure 1. Maps of Puget Sound within the larger Salish Sea marine ecosystem, including much of its comprising watersheds (areas inside dashed line). (Left map) Hardened shorelines. Red segments indicate presence of armoring along ShoreZone-defined shoreline (WDNR 2001; Anchor QEA 2009; PSNERP 2010). Data were compiled from local county data, photo interpretation, and field observations. Overwater structures are not differentiated due to their small relative scale, but are concentrated near urban centers. (Right map) Eelgrass (Zostera marina) distributions and impervious surfaces. Dark green segments of shoreline are “continuous” eelgrass beds while light green lines are “patchy” beds (WDNR 2001). Imperviousness values range from 100% (darkest red) to 0% (white) and are a general proxy for degree of urbanization (Fry et al. 2011).

Because eelgrass supports a number of ecologically and economically important species, declining eelgrass cover has garnered attention by management agencies within Puget Sound. The Puget Sound Partnership (PSP), a state management agency overseeing the restoration of Puget Sound, has set a 2020 recovery target for eelgrass that calls for a 20% increase in seagrass cover (Puget Sound Partnership 2011). However, it is unclear what management actions, singularly or in combination, will most efficiently accomplish this objective. In this article we estimated functional relationships between the type and magnitude of stressors and seagrass cover using expert elicitation. Specifically, we generated relationships between changes in shoreline armoring, overwater structures, sediment loading and nutrient loading and changes in seagrass cover. We then used these relationships as the basis of analyses that identify how single or multiple management actions influence the cover of eelgrass in Puget Sound.
Methods

We focused on management actions aimed at addressing three eelgrass stressors: (1) shoreline armor removal or modification; (2) overwater structure removal or modification; and (3) improvements in water clarity via reductions in artificial nutrient and sediment loadings. Modifications include project design changes that allow natural processes to persist; for example, overwater structure modification, which may include incorporating dock grating, which allows light passage whereas shoreline armoring modifications may incorporate native substrates and more gradual slopes (soft-armoring). We chose to analyze these types of management actions because of their importance to the Puget Sound ecosystem (Mumford 2006; Simenstad et al. 2006; Thom et al. 2011; Clancy et al. 2009).

Elicitation of Seagrass Experts

We used a stratified chain referral approach (Bernard 2006) to identify individuals with specialized knowledge of seagrass Puget Sound. Following standard practice (Morgan and Henrion 1992), our aim was to identify >10 individuals with expert knowledge. Our search yielded 20 regional experts, and 19 of these experts participated in our study. Beliefs were elicited on-line using a Google Docs questionnaire (see Appendix 1 in online supplemental material) about relationships between eelgrass and six potential management actions: (1) change in extent of shoreline armor; (2) change in extent of shoreline armor that is modified (i.e., the extent of armor that is altered to allow natural processes to persist, as opposed to being removed entirely); (3) change in area of overwater structures; (4) change in the area of overwater structures that is modified (i.e., the area of structures that is altered to allow natural processes to persist, as opposed to being removed entirely); (5) change in nutrient loading; and (6) change in sediment loading. Two of the experts did not answer the questions regarding nutrient or sediment loading.

In our elicitation we asked respondents to indicate how eelgrass cover would respond to incremental changes (“magnitude” of the management action), ranging from complete removal to doubling, of each of four stressors (shoreline armor, overwater structure, nutrients, and sediments), and ranging from 20% to doubling, of two modifications (the shoreline armor and overwater structures modifications apply only in the positive direction). Experts characterized the resulting change in eelgrass cover using five categories: (1) large decrease, (2) small decrease, (3) no change, (4) small increase, and (5) large increase. Experts were also asked to indicate their associated confidence in their estimate (on a scale of 1 to 10, corresponding from high uncertainty (1) to absolute certainty (10)). Thus, experts scored each Action x Magnitude scenario separately (from 1 to 5), but each expert evaluated all scenarios. Initially, we left the interpretation of “large” and “small” increase/decrease to respondents; however, we remained in e-mail and phone contact with them and post hoc, we asked respondents to report how they interpreted this language. On average, experts (n = 10) stated that a change of >10% (SD 5%) would be a “large” change. The consequences and limitations of using nominal categories are discussed below.

Statistical Analysis to Predict Relative Impacts of Actions

We conducted two analyses on the expert-derived data. To estimate the relative effect of the different management actions on eelgrass recovery, we performed ordinal regressions, and then used a probabilistic model analysis to predict which management actions, singularly
or in combination, would most efficiently accomplish a restoration objective. We detail these two approaches below.

Because response scores were ordered categories describing predicted response of eelgrass to management actions, an ordinal regression with a logit-link (also known as a cumulative logit model or a proportional odds model) is the most appropriate analytical method (McCullagh 1980). Cumulative logit models give the probability that a response will be in a given category or less (i.e., a score of 4 or less). Thus, unlike more familiar approaches like analysis of variance, these models do not give a simple mean prediction. Instead, they produce a cumulative probability curve. Here, response ranks were: (1) large decrease, (2) small decrease, (3) no change, (4) small increase, and (5) large increase. The respondents’ uncertainty scores were scaled to 0–1 and included as weights in the model. The factors in the model were: “Action” (e.g., nutrient loading, overwater structures; described above) and “Magnitude” of the management action (e.g., –20% to –100%). “Respondent” was included as a random effect because individuals gave scores for each Action x Magnitude scenario. We fit a series of nested, candidate models and used Akaike’s Information Criterion (AIC) weights ($w_i$) to select the best-fit model (Burnham and Anderson 1998). We used the “clmm” procedure in the R package “ordinal” to conduct analyses (Christensen 2012; R Development Core Team 2012).

**Probabilistic Model Analysis**

We carried out a probabilistic model analysis based on Bayesian network techniques. A probabilistic model, or Bayesian network, consists of a graphical structure and a probabilistic description of the relationships among variables in a system (Borsuk, Stow, and Reckhow 2004). Variables, in this case eelgrass cover and its associated stressors, are represented as nodes. Causal relationships, the stressor–eelgrass cover relationships in this instance, are represented as directed links between the nodes and are specified by conditional probability distributions.

The Bayesian networks that related each management action to eelgrass cover were designed using the graphical network building program, Netica (Norsys Software Corp. 2010). Conditional probabilities between management actions and eelgrass cover were derived as follows. First, we used the frequency of expert responses (weighted by reported level of confidence as described above) to generate probabilities, which were then inserted into the conditional probability table of each network. The expected change in eelgrass cover of each project is calculated by multiplying the probability of each possible outcome by its value and then summing these values.

**Results**

**Distributions of Expert Responses**

Survey respondents had an average of 23 years (range: 5–49 years) research experience in Puget Sound; all cited some form of biology or ecology in their background, with a wide spectrum of specific expertise (e.g., marine benthic ecology to phycology). The distributions of expert responses revealed a range of beliefs about the effects of management actions on eelgrass (Figure 2). As expected, responses indicated a general belief that removing threats would increase eelgrass cover, while expanding threats would reduce eelgrass cover. However, these patterns were complex, with predicted effects on eelgrass varying by management action and magnitude (Figure 2).
Nearly all (18/19) experts believed that eelgrass cover would show a small increase in cover if 20% or 40% of overwater structures were removed (Figure 2). If 60% of overwater structures were removed, three experts believed that eelgrass cover would experience a large increase, and this increased to 8 and 10 experts under the 80% and 100% overwater structure removal scenarios, respectively. The pattern for shoreline armor removal was quite different than that for removal of overwater structures (Figure 2). For instance, if 20% of shoreline armor in Puget Sound could be removed, 10 of 19 experts believed that
eelgrass cover would experience a small increase and 9 that it would not experience any change. A 60% removal of shoreline armor was needed before 18 of the 19 experts believed an increase in eelgrass would occur. For both shoreline armor and overwater structures, experts believed that modification of the armor or overwater structure was less effective than removal (Figure 2).

In general, experts believed that small changes in nutrient or sediment loading would have smaller impacts on eelgrass than would small changes to shoreline armor or overwater structures. For instance, more than 50% of respondents believed a 20% change in nutrient or sediment loading would have no impact on seagrass cover. In contrast, experts believed large (i.e., 100% reductions) management efforts targeting sediment or nutrient loading would have larger impacts than efforts of similar magnitude targeting shoreline armor or overwater structures (Figure 2).

Experts reported a range of confidence for their assessment of the effects of management actions on eelgrass cover. We found significant differences in the mean level of expert-assigned confidence among the levels of change in management actions (ANOVA: $df = 9$, $F = 7.535$, $p < .001$). There was also a significant relationship between the type of management action and the level of expert-assigned confidence (ANOVA: $df = 4$, $F = 40.755$, $p < .001$) (Figure 3). Experts had greater confidence in their assessments at higher magnitude levels of change (i.e., complete removal or doubling of the management action, than at lower magnitudes of change). Expert confidence was significantly higher for overwater structures and overwater structures modification than that for the other actions. Interestingly, expert confidence (included in the ordinal regression as a weighting factor) appeared to be

**Figure 3.** Mean levels of expert confidence assigned to changes in management actions. Error bars indicate ±1.0 s.e.
negatively related with experience: mean certainty scores never exceeded 6 for experts with >25 y of experience ($n = 6$), whereas 6 out of 13 experts with <25 y of experience had mean certainty scores greater than 6.

**Predicted Relative Impacts of Actions**

Ordinal regression analyses to estimate the relative effect of the different management actions on eelgrass recovery revealed that a model that included “Action,” “Magnitude,” and “Respondent” (Model 5), gave the best fit to the data (Table 1). Model 2 performed similarly to model 5, but was more complex and carried slightly less weight (Table 1). Therefore, of the eight models we explored, we chose model 5 as the best-fit model. Importantly, removal of random effect “Respondent” strongly decreased the model fit suggesting that variation among individuals was important (e.g., compare model 5 and model 6).

Respondents predicted that reduced sediment loading and modification of shoreline armoring would have the smallest effect on recovery of eelgrass (Figure 4, Table 2). Experts predicted that removal of overwater structures, changes in nutrient loading and removal of shoreline armoring would have significantly larger effects on eelgrass recovery than would sediment loading (Table 2), with removal of overwater structures having the largest effect. Modification of overwater structures showed a non-significant trend towards a larger effect that sediment loading (Table 2).

**Predicted Strategies for Achieving a Large Increase in Eelgrass**

Our probabilistic model revealed that several strategies could be used to achieve “large” increases in the cover of eelgrass. Major changes in single stressors would be sufficient to achieve a large increase. For instance, our models predict that removal of 100% of overwater structures, 100% removal of shoreline armoring, and 100% reduction of nutrients would each have an approximately 50% probability of achieving a large increase in eelgrass cover;

### Table 1

Results of model fitting of selected cumulative logit models examining the predicted response of management actions (“A”) and the magnitude (“M”) of those actions on the predicted recovery of eelgrass

| Model | Log-likelihood | AIC | $\Delta_i$ | $w_i$ |
|-------|----------------|-----|-----------|------|
| (1) $A + M | R + R*A + R*M$ | -188.23 | 411.65 | 2.54 | 0.116 |
| (2) $A + M | R + R*A + R*M$ | -188.23 | 409.65 | 0.54 | 0.316 |
| (3) $A + M | R + R*A$ | -334.29 | 698.58 | 289.47 | $5.7 \times 10^{-64}$ |
| (4) $A + M | R + R*M$ | -190.56 | 411.11 | 2.00 | 0.15 |
| (5) $A + M | R$ | -190.56 | 409.11 | 0.00 | 0.41 |
| (6) $A + M$ | -207.46 | 440.93 | 31.82 | $5.1 \times 10^{-8}$ |
| (7) $A + M + A*M | R$ | -185.13 | 438.26 | 29.15 | $1.9 \times 10^{-7}$ |
| (8) $A + M + A*M$ | -202.59 | 471.18 | 62.07 | $1.4 \times 10^{-14}$ |

*Note: A and M were fixed effects in the model. Respondent (“R”) was a random effect. AIC = Akaike’s Information Criterion; $\Delta_i = \text{AIC}_i - \text{AIC}_{\text{min}}$; $w_i$ = AIC weight for each model. The symbol “$|$” separates fixed and random effects.*
The expected value generated from each of the distributions would likely give a large increase in eelgrass cover. When combinations of management actions are taken, more modest levels of intervention are required to achieve large increases in eelgrass (Figure 5). For example, our models suggest that removal of 40% of each of the four stressors we examine would have a 20% probability of achieving a large increase and a 60–90% probability of a small
Table 2
Parameter estimates for ordinal regression (cumulative link model) with logit link for the management actions

| Action type                           | Model coefficients (s.e.) | p-value |
|---------------------------------------|---------------------------|---------|
| Nutrient loading                      | 1.19 (0.54)               | 0.027   |
| Overwater structures                  | 1.58 (0.49)               | 0.001   |
| Overwater structures modification     | 0.85 (0.49)               | 0.085   |
| Sediment loading                      | 0.00                      |         |
| Shoreline armoring                    | 1.12 (0.53)               | 0.033   |
| Shoreline armoring modification       | 0.30 (0.57)               | 0.575   |
| −20% modification                     | −4.39 (0.57)              | < 0.001 |
| −40% modification                     | −3.27 (0.47)              | < 0.001 |
| −60% modification                     | −0.74 (0.41)              | 0.071   |
| −80% modification                     | −0.96 (0.38)              | 0.011   |
| −100% modification                    | 0.00                      |         |

Note: P-values are for Wald tests and indicate whether the parameter estimate differed from zero. In context, this indicates whether the effect size for an action differed from that of sediment loading or for a −100% change, respectively. Respondent (added variance = 1.067) was included as random effects in the model.

Discussion

Seagrass is an important biogenic habitat in Puget Sound, as it is in many estuarine systems around the world (Fonseca 1998; Jackson et al. 2001; Mumford 2006; Phillips 1984; Waycott et al. 2009), and its extent can influence broad community dynamics that affect an array of ecosystem services (Plummer et al. 2013). Considering the challenges wrought by coastal human population pressures on seagrass systems worldwide (Orth et al. 2006; Waycott et al. 2009), any approach for successfully protecting, monitoring, managing and restoring these habitats, particularly in urbanized estuaries, requires data describing the functional relationship between anthropogenic stressors and the cover and condition of eelgrass (Fonseca 1998; Short et al. 2002). Here, we used an expert elicitation to generate functional responses of eelgrass to changes in four primary stressors throughout Puget Sound (Thom et al. 2011), and used this information to parameterize a probabilistic restoration model. Our analyses suggest that mitigation of any of the stressors we examined would improve the status of eelgrass in Puget Sound; however, the strongest response occurred with the removal of overwater structures. That is, even small changes in overwater structures were predicted to have more positive effects on eelgrass relative to the three other stressors we analyzed.

Grech and colleagues (2012) noted that the development of coastal infrastructure is the most serious threat to seagrass beds in the North Pacific Bioregion, which includes Puget Sound. Recent reports highlighting specific stressors to eelgrass in Puget Sound also suggest that overwater structures and other direct disruptions from human activity (e.g., construction, dredging/filling) represent priority threats with known, long-term effects on eelgrass (Thom et al. 2011); in most cases, these stressors are spatially clustered near
Figure 5. Twenty-four combinations of management actions predicted to achieve a “large” increase in eelgrass cover. Shaded-in areas refer to the action and its magnitude (of reduction) undertaken in a particular scenario.

centers of population growth and are also subjected to an array of other human pressures (Fresh et al. 2006; Thom et al. 2011; Thom, Williams, and Dieferderfer 2005). Thus, our result emphasizing the importance of overwater structures not only reflects the degree of certainty associated with related impacts to eelgrass (primarily borne by light limitation), but also hints at the importance of a diverse range of threats in the Puget Sound Region.

Our results, however, generally differ from work elsewhere that emphasizes the indirect impacts of water quality degradation from increased nutrient additions and sediment runoff from human-altered watersheds (Grech et al. 2012; Orth et al. 2006; Short and Wyllie-Echeverria 1996). In fact, increased eutrophication is considered the primary cause of seagrass declines in much of Europe (Cardoso et al. 2004), parts of Belize (Short et al. 2006), and the U.S. East and Gulf coasts (e.g., Waquoit Bay, Chesapeake Bay, Florida Bay). The relative importance of overwater structures as an eelgrass stressor in Puget Sound (e.g., Fresh et al. 2006; Nightingale and Simenstad 2001; Thom et al. 2005) may be a function of the region’s geomorphology. Puget Sound is a fjord-like estuary with deep basins and steep bathymetry (Burns 1985), and eelgrass here is often limited to a narrow fringe of shallow habitat along the edges. Shallow embayments with large eelgrass meadows which typify a number of other geographic regions, are much less common regionally, and consequently a larger proportion of eelgrass
cover may be found in fringing beds that co-occur with locations of overwater structures. Furthermore, large-scale eutrophication of the system by anthropogenic nutrient inputs is generally considered unlikely because of high ambient nitrogenous nutrient concentrations and rapid water exchange (Mackas and Harrison 1997).

The Puget Sound Partnership has called for an increase of 20% increase in eelgrass cover in Puget Sound (Puget Sound Partnership 2012). While a range of protection and restoration measures have been proposed, the PSP has not identified the most efficient mix of strategies that could be used to achieve this target. Participants in our expert elicitation reported that a “large” increase in eelgrass would approach the PSP 20%-increase target. Our probabilistic models clearly showed that relying on a single management action to achieve “large” increases in eelgrass cover is unrealistic. For instance, to generate this level of change an 80–100% reduction of shoreline armor or nutrients would be required. Given the economic and social costs of such actions (Holl and Howarth 2000), these approaches seem unlikely. However, our model revealed that modest actions across several threats could achieve a “large” increase in eelgrass. This agrees with the bulk of seagrass studies conducted in our region, which recommend using a mix of approaches tailored to the local political, social and biological environment; and intervening in land management approaches, watershed management, and coastal planning (Grech et al. 2012; Waycott et al. 2009). Unfortunately, because we used nominal categories (e.g., “large increase”) in our questionnaire, it is impossible to directly link our results to the specific 20% target of the PSP. Nonetheless, respondents clearly indicated that a 20% increase is “large” (see Methods), and the general conclusion of our work should be robust to the vagary of the response format.

Management decisions often occur at a time frame that is much faster than the response time of scientific study. Thus, scientists are frequently called to inform management decisions, but may not have the temporal luxury to conduct empirical analyses. In these cases, expert elicitation can be a useful stop-gap measure (Runge, Converse, and Lyons 2011; Morgan, Pitelka, and Shevliakova 2001). While informative and a practical necessity in a number of circumstances, analyses such as the work we report, here should be considered an interim product, not a replacement for quantitative empirical data. The substitution or over-interpretation of coded expert opinion for empirical data is an insidious problem that has hampered progress in a number of conservation and restoration settings since it can relax the pressure to monitor ecological indicators (Ruckelshaus et al. 2002).

Our work is meant to provide strategic guidance about restoration and conservation of eelgrass. By strategic, we are referring to decisions that are linked to policy goals and are generally long-range, broadly based and inherently adaptable (FAO 2008). In contrast, our results should not be used to inform tactical decisions (i.e., short-term decisions linked to operational objectives in the form of rigid guidelines). Furthermore, our work was focused on Puget Sound as a whole, and not any specific location; the management actions included here address key priority stressors only (i.e., are not comprehensive); the relative spatial scale of the stressors was not specified; and we assumed effects due to multiple actions were additive (because the nature of interactions was unknown) and may not represent the true nature of the system. As a consequence, our work can be used to inform what suites of actions and their general magnitude are needed to restore Puget Sound eelgrass. However, our work is insufficient to provide tactical advice on the exact location or magnitude of management actions. Such tactical advice must await detailed, place-based observations, experiments, and models.
The recognized environmental and economic importance of eelgrass to Puget Sound (Plummer et al. 2013; Thayer and Phillips 1977) requires prompt input of scientific advice into management and policy arenas (Lester et al. 2010). Our work fills an important knowledge gap in the face of rapidly approaching legislative deadlines. Efforts to characterize the true costs of management actions, including the distribution of costs among citizens, is a difficult enterprise (Holl and Howarth 2000) and primarily has been addressed in the context of restoration damage assessment (Fonseca, Julius, and Kenworthy 2000). Further work quantifying the economic and social costs of different strategies in different locations would be a useful next step allowing for a more thoughtful prioritization of management actions. It is obviously crucial that the management and policy institutions pursue targeted ecological and socioeconomic research that could inform tactical decisions. In the meantime, by adhering to the principles of adaptive management (Holling 1978) it is clear that eelgrass restoration can proceed without waiting for “perfect information” (e.g., Tallis et al. 2012).

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Supplemental Material
Supplemental data for this article can be accessed on the publisher’s website.

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