Combining ecosystem services assessment with structured decision making to support ecological restoration planning

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Abstract

Accounting for ecosystem services in environmental decision making is an emerging research topic. Modern frameworks for ecosystem services assessment emphasize evaluating the social benefits of ecosystems, in terms of who benefits and by how much, to aid in comparing multiple courses of action. Structured methods to evaluate social benefits using decision analytic approaches are emerging for the practice of ecological restoration. In this article, we combine ecosystem services assessment with structured decision making to estimate and evaluate measures of the potential benefits of ecological restoration with a case study in the Woonasquatucket River watershed, Rhode Island, USA. We partnered with a local watershed management organization to analyze dozens of candidate wetland restoration sites for their abilities to supply five ecosystem services – flood water retention, scenic landscapes, learning opportunities, recreational opportunities, and birds. We developed 22 benefit indicators related to the ecosystem services as well indicators for social equity and reliability that benefits will sustain in the future. We applied conceptual modeling and spatial analysis to estimate indicator values for each candidate restoration site. Lastly, we developed a decision support tool to quantitatively aggregate the values and help the organization screen the restoration sites. Results show that restoration sites in urban areas can provide greater social benefits than sites in less urban areas. Our research approach is general and can be used to investigate other restoration planning studies that perform ecosystem services assessment and fit into a decision-focused process.

Keywords

Introduction

Ecosystem services, regarded as the natural characteristics of ecosystems that benefit people, is a concept with increasing potential to influence global environmental management,

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particularly as it is incorporated into real world decisions that are focused on sustaining human health and the environment, protecting and conserving wildlife, and assisting in the recovery of damaged or degraded ecosystems (Balvanera et al. 2001; Aronson et al. 2010; Perrings et al. 2011; Griggs et al. 2013). Assessing ecosystem services requires making explicit connections between how environmental management affects ecosystems, i.e., ecosystem response models; how ecosystem services are produced from management actions, i.e., ecological production function models; and what the monetary and non-monetary values of those ecosystem services are, i.e., valuation models (Wainger and Mazzotta 2011).

In this article, we combine modern approaches to ecosystem services assessment with structured approaches to decision making for ecological restoration planning. In the past, ecological restoration decisions generally utilized physicochemical and biological criteria as proxies for maximizing the ecological functioning of ecosystems. Federal regulatory programs, such as the U.S. Clean Water Act, use physical, chemical, and biological metrics to set permitting and decision priorities for restoration (King et al. 2000; Doyle and Shields 2012). In a similar manner, restoration practitioners have been guided by a vision to conserve critical species and ecological integrity (McDonald et al. 2016). Acting on this vision often requires a focus on recovering acceptable structural and functional components of ecosystems as targets and objectives; for example, nutrient concentration, biological diversity, plant or animal biomass, habitat type and size, or hydrologic flow rates.

As the ecosystem services concept has evolved, modern viewpoints emphasize the need to make decisions based on the social benefits of ecological restoration (Wainger et al. 2010; Martin 2017). Ecological restoration in an urban area, for example, is unlikely to achieve the full functioning of a pristine ecosystem, yet restoration in urban areas is likely to benefit a large number of people, particularly those who may not have ready access to the direct benefits of ecosystems, including clean air and water, recreation, education, and scenic and wildlife viewing. Accordingly, frameworks that equip researchers with concepts and tools to analyze a full range of ecosystem services and their potential benefits, in terms of who benefits and by how much, are emerging so that management alternatives are more comparable across different landscapes (Wainger and Mazzotta 2011; Olander et al. 2018).

Our objective with this research is to test whether a social benefits approach to ecological restoration is useful to groups who plan and implement restoration in a real-world decision making context. Previous work focused on concepts and approaches for developing benefit indicators, defined as natural or proxy representations of impacts of restoration on social welfare, that can be measured without monetary units but still reflect what is valued by people (Wainger et al. 2010; Mazzotta et al. 2016; Olander et al. 2018). However, research is limited on how to use this approach in a decision-focused manner, where benefit indicators can be combined with other management objectives and used to support decision making. To do this, we incorporate the approaches of structured decision making, or SDM (Fig. 1; Gregory et al. 2012). SDM is a stepwise framework for researchers to investigate complex environmental problems. The SDM framework guides decision makers in clarifying goals and objectives, identifying decision alternatives, estimating tradeoffs in the consequences of those alternatives, and choosing preferred alternatives for implementation and monitoring.

By breaking down the various parts of the decision making process, SDM provides a format that is transparent, documentable, and amenable to good decision making. As such, SDM has been used to analyze wildlife and landscape management problems with case studies in the United States, Canada, and Australia (e.g., Ogden and Innes 2009; Martin et al. 2009; for a review, see Schwartz et al. 2017).

The general SDM process has been proposed for ecological restoration elsewhere (Kozak and Piazza 2014; Guerrero et al. 2017), but our approach is specifically useful for incorporating objectives that require benefit indicators to be evaluated. To guide that process, we use the ecosystem services cascade (Potschin and Haines-Young 2011), which provides context for selecting benefit indicators (step 2; Fig. 1). Likewise, we incorporate the analytical approaches of multi-criteria decision analysis (MCDA) to evaluate tradeoffs in non-monetary benefit indicator values (step 5; Fig. 1). Methods for MCDA are well known for their ability to aggregate monetary and non-monetary ecosystem service tradeoffs into a comparable metric, without requiring them to be in common units (e.g., dollars) or even in numerical terms, for decision making purposes (Langemeyer et al. 2016; Saarikoski et al. 2016). Yet, MCDA has not been fully embraced for making decisions based on ecosystem services assessment for several reasons: i) prioritizing management alternatives is not a commonly-used step of ecosystem services assessment (Martinez-Harms et al. 2015), ii) the mathematical concepts of MCDA are generally unfamiliar to ecosystem service analysts (Olander et al. 2016), and iii) many of the current MCDA approaches that have been used for ecosystem services assessment do not make explicit important assumptions that imply different decision maker preferences for evaluating ecosystem service outcomes and therefore affect how management alternatives are prioritized (Martin and Mazzotta 2018).

Study area

In southern New England, the Woonasquatucket River flows southeast through northern Rhode Island into the city of Providence, the state's capital (Fig. 2). The river is approximately 31 kilometers long and its watershed covers 132 square kilometers. It flows from mixed suburban and agricultural settings in its northern headwaters into highly urbanized neighborhoods downstream. The Woonasquatucket River is one of fourteen recognized by the American Heritage Rivers Protection Program for the role it played in the Industrial Revolution, which consequently degraded and destroyed large amounts of wetlands in the watershed. The interspersion of urban and semi-rural land-use types in the watershed provide an opportunity to compare wetland restoration alternatives across highly populated to less populated areas.

We worked with the Woonasquatucket River Watershed Council (WRWC) to plan for wetland restoration in the watershed. The WRWC is a non-profit organization, whose mission is to support and promote restoration and preservation in the watershed. Among other initiatives, the WRWC is seeking to implement wetland restoration in the watershed. However, funding is a significantly limiting factor for this organization; in general, funding for implementing restoration in urban areas is scarce. The WRWC does not currently use a structured process for analyzing the potential for wetland restoration in the watershed

but relies on a more ad-hoc approach based on funding availability, opportunities for partnerships with other organizations, and expressed interests of community groups.

Earlier work to investigate wetland restoration in the watershed focused on the potential to recover the ecological functioning of previously destroyed wetlands (Miller and Golet 2001; Golet et al. 2003). These studies evaluated dozens of candidate wetland restoration sites based on site-specific functional assessments, resulting in qualitative measures of the *opportunity* that a wetland had to perform a function and the *effectiveness* of a wetland to perform a function. The measures were modified based on wetland size and on the perceived presence of social significance of a function (Golet et al. 2003).

Reflecting on these functional assessments, local groups conducting restoration in Rhode Island, including the WRWC, found that considering ecological function alone often made it hard to get funding for restoration in urban areas of the watershed, because the urban sites typically scored lower using functional assessments. This was particularly true for areas in the city of Providence that are densely populated, where people may not have access to ecosystem services or are more vulnerable to environmental disasters. Urban ecosystems have the potential to provide people with greater social benefits because natural ecosystems are more scarce in an urban setting and the proximity of these ecosystems to more people means more people can benefit from ecosystem services (Gobster 2001; Hubacek and Kronenberg 2013; Larondelle and Haase 2013; Kremer et al. 2015). For these reasons, we developed a suite of research methods on the social benefits of wetland restoration in urbanizing areas (Bousquin et al. 2015; Mazzotta et al. 2016), and we partnered with the WRWC to apply them in the watershed using an SDM approach (Fig. 1).

The WRWC identified two fundamental objectives for selecting restoration sites: maximizing ecosystem service benefits, and maximizing social equity. For ecosystem services, the WRWC desired a better estimate of how wetland restoration could benefit people than previous functional assessments. While they would have preferred to have monetary measures of ecosystem service benefits, developing these measures was not feasible because the organization did not have the skills or resources required to estimate monetary measures. For this reason, we supported the WRWC in developing non-monetary benefit indicators associated with several ecosystem services to allow restoration alternatives at smaller, more urban locations to be compared more adequately against larger, less urban sites. For social equity, the WRWC wanted to know whether there are groups of people that are particularly socially vulnerable who might benefit from ecosystem services provided by wetland restoration.

Methods

The inventory of potential restoration sites to serve as management alternatives was not comprehensive. The WRWC chose to focus on the 77 candidate restoration sites that were previously analyzed in the functional studies in the watershed (Miller and Golet 2001; Golet et al. 2003). We screened those sites for their restoration feasibility based on current site features in imagery from Bing and Google Earth in May, 2017. Twelve sites are fully developed and therefore not feasible, and 11 site polygons were re-drawn to exclude recently

developed lands. As a result, we determined that 65 candidate restoration sites could serve as feasible alternatives for the analysis.

Benefit indicators

Identifying appropriate performance metrics for the objectives centered on identifying which ecosystem services were influential for wetland restoration planning in the watershed and developing benefit indicators for them. This process took several years of research, including stakeholder and public engagement (Hychka and Druschke, 2017), and resulted in developing a rapid benefits indicator method (Mazzotta et al. 2016). Five wetland ecosystem services/benefits were identified for assessment in the watershed – flood water retention/ flood risk reduction, scenic landscapes/scenic views, learning opportunities/environmental education, recreational opportunities/recreation, and birds/bird watching. The ecosystem services are not comprehensive; rather, they were chosen in part based on available information and preferences of restoration managers in Rhode Island and the WRWC.

We performed literature reviews and conceptual modeling to develop 22 benefit indicators based on several site characteristics: i) potential beneficiaries and access near the site, ii) site quality, iii) complementary ecosystems or built infrastructure near the site that potentially enhance benefits, and iv) other wetlands or infrastructure near the site that may substitute for the same benefit (Table 1). Many frameworks and models for ecosystem services assessment aim to predict outcomes in terms of the provision of ecosystem services (Bagstad et al. 2013). In contrast, focusing on beneficiaries, access, site quality, complementarity, and substitution characteristics emphasizes uses of those services in terms of who could potentially benefit from restoration and by how much. Restoration managers in Rhode Island reasoned that these considerations were important to planning restoration (Hychka and Druschke 2017). More detailed explanations of the process of selecting and measuring these characteristics are available elsewhere (Wainger and Mazzotta 2011; Mazzotta et al. 2016).

For beneficiaries, we aimed to identify people within areas with access to each of the five ecosystem services (e.g., "servicesheds;" Tallis et al. 2015). The more people who are likely to benefit, for example, the more homes in a floodplain and downstream from a restoration site that could benefit from a site's water retention capacity, the greater the total potential benefits of restoration. In addition to estimating the number of potential beneficiaries, we developed indicators based on the ability to access ecosystem services at a site. Recreation, for example, requires access points and often requires infrastructure, such as roads, trails, or bus stops. For these reasons, we developed several indicators for scenic views, recreation, and bird watching based on presence/absence of roads, hiking trails, bike trails, and bus stops near candidate restoration sites (Table 1).

The site quality, complementarity, and substitution characteristics modify the expected magnitude of ecosystem service benefits relative to other sites. This perspective on ecosystem services assessment is limited in current decision support tools (Healy and Secchi 2016). For site quality, we assumed that there were characteristic ecological features of a candidate restoration site that would enhance some benefits. Based on this reasoning, we developed indicators for flood risk and recreation based on the size of the candidate restoration site. For complementarity, we developed an indicator for scenic views based on

the variety of natural land use types nearby, which may enhance the potential scenic view benefits of a candidate restoration site. For substitution, we developed indicators for flood risk, scenic views, education, and recreation based on whether there were wetlands or other green space already available in the vicinity that could potentially diminish the benefits of a candidate restoration site relative to others.

Two additional benefit indicators were identified to evaluate sites, based on their potential to improve social equity in the distribution of benefits and on their reliability in terms of ability to sustain benefits into the future (Table 1). For social equity, we assumed that restoration in locations that are vulnerable to environmental pressures could provide more benefits than restoration in less vulnerable locations. For reliability, we assumed that benefits from sites with fewer developmental pressures are more likely to persist over time, and therefore those sites will be more valuable to restore than others.

Spatial analysis

A range of methods have been proposed to estimate benefit indicator measures based on the beneficiaries, access, site quality, complementarity, and substitution characteristics, to allow for different approaches to measurement (e.g., site visits, desktop methods; Mazzotta et al. 2016). We determined that many of the indicator values for our study watershed could be estimated based on readily-available spatial data, using geographic information systems (GIS). For this reason, Python-based spatial analysis tools for ArcGIS were developed to streamline the calculation of indicator values.[1]

The spatial tools require a variety of different user inputs depending on which indicators are assessed and available data. Where possible, our analysis used data that are available on a national scale to ensure better transferability of methods to other places, but in some instances more detailed state-scale data were required or preferred. All state and national data inputs that we used for the case study are listed in Table S.1 in the Electronic Supplementary Material.

Although describing the methods and assumptions for each spatial relationship is beyond the scope of this article and described elsewhere, some assumptions and methods are pertinent for the case study and its transferability. To determine a buffer zone for estimating flood risk beneficiaries (Table 1), flood models were calibrated with input from past streamflow and rainfall data as well as models of storm and restoration scenarios in the watershed (Bousquin et al. 2015). These data were used to develop the downstream distance from a restoration site in the watershed where the change in flood flows reaches <1% (Bousquin et al. 2015). This buffer zone value was equal to 4 km for our case study and was used to calculate flood risk benefits (Table 1).

We used the U.S. Geological Survey National Hydrography Dataset to identify addresses in catchments downstream from the candidate restoration sites. The dataset includes locations in the site catchment as well as downstream catchments that may include addresses in tributaries that are upstream of Woonasquatucket River streamflow. For this reason, we

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[1]These tools are available for download at https://www.epa.gov/water-research/rapid-benefit-indicators-rbi-approach

manually inspected and modified the results of the spatial analysis tool for flood risk beneficiaries, removing any identified addresses or dams that were not actually downstream from a candidate restoration site. Given the prevalence of dams in the Woonasquatucket River watershed, we counted and presented the number of dams downstream, rather than simply presenting presence or absence of downstream dams.

Another important relationship is estimating the number of beneficiaries within a specific radius of a candidate restoration site who receive scenic view benefits. The probability that someone receives scenic view benefits and the quality of those benefits decreases with distance, but people may still receive scenic view benefits beyond, for example, 50 m. As a result, we developed multiple indicators for beneficiaries based on how many people live between, for example, 50 and 100 m of the sites. To avoid double counting the people in the first range, only the people outside that range were counted for the second range, and so on. This logic is similar to how we identified distance ranges for recreation benefits (Table 1).

An important spatial relationship was percent coverage of substitute land covers within a distance range. The potential benefit of a restoration site is lower for a site with more substitutes, relative to sites with few or no substitutes, such as other wetlands for flood risk benefits or other green space for recreation benefits. Ideally, the substitution possibilities would be determined for each person who benefits, but this is not practically feasible, so we determined substitutes for each candidate restoration site. To do so, we doubled the radius of the area selected to define beneficiaries and evaluated the potential substitutes within this area. For example, for education benefits, the number of beneficiaries is defined as number of educational institutions within 400 m of the restoration site (Table 1). Because a school within this radius may benefit from an existing wetland at a similar distance in the other direction, we evaluated substitutes by determining the percent of the area within 800 m of the candidate restoration site that is existing wetlands (Table 1).

To evaluate social equity, we used the 2010 Social Vulnerability Index by Census Tracts provided by the National Oceanic and Atmospheric Administration.[2] The Social Vulnerability index combines a set of statistically-relevant demographic variables summarizing information on race, class, wealth, age, ethnicity, and other factors into an index (Cutter et al. 2003). Social vulnerability scores within 4 km of a candidate restoration site were classified into percentages based on the percent of the total area falling into each of three vulnerability categories: low, medium, and high. We used the percent area that fell into the medium category as the benefit indicator value. It did not make sense to include all three categories as separate indicators in the analysis. Since sites with more high values tended to have a reciprocal effect with low medium and more low values, we determined that using high values could be misleading.

We used the spatial analysis tools to calculate quantitative and qualitative non-monetary indicator metrics, resulting in a large multivariate data set, characterized by all 24 benefit indicator values for each of the 65 candidate restoration sites. These data represented the projected consequences of restoring each candidate restoration site (Fig. 1).

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[2] http://artsandsciences.sc.edu/geog/hvri/sovi%C2%AE-0

Decision support tool

We developed a spreadsheet decision support tool in Microsoft Excel, using Visual Basic macros, to simplify and automate the process of evaluating the consequences of the 65 candidate restoration sites (step 5; Fig. 1; Electronic Supplementary Material). The decision support tool requires the user to specify their preferences as numerical weights for each of the ecosystem service benefits as well as the social equity and reliability indicators. It then calculates aggregated scores for each candidate restoration site and displays the ten highest ranking sites along with their aggregated scores, short descriptions of the sites, and URL links to their locations in Google Maps (Fig. 3).

Analytically, the tool uses a simple additive aggregation function for MCDA called compromise programming (Zeleny 1973) that performs the following steps: i) it uses a linear function to transform all benefit indicator values into a comparable performance value between 0–1; ii) it calculates squared (Euclidean) distances between the transformed benefit indicator values and the ideal indicator value, which is equal to one; iii) it distributes the seven benefit category weights given by the user (five benefits, social equity, and reliability) equally among their corresponding indicators and squares them; iv) it performs a weighted aggregation by multiplying the distance measurements of each indicator at each site by the squared preference weights for each indicator; and v) it scales and ranks the aggregated distance scores for each candidate restoration site with respect to all other sites.

The WRWC chose the distance measurement technique rather than other approaches; that is, they preferred to evaluate the candidate restoration sites based on which sites had the potential to provide the most balanced impact on the indicators versus ones that had the potential to provide the greatest impact on indicators (for a description of method assumptions relevant to decision making, see Martin and Mazzotta 2018).

The decision support tool is used by entering preferences based on the relative importance of each of the benefit categories under different planning scenarios. To edit a scenario, the user divides 100 points among the selected benefit categories that s/he wishes to investigate. For example, the WRWC was interested in preparing a proposal for a restoration grant from the U.S. Federal Emergency Management Agency, and they chose to evaluate the candidate restoration sites by emphasizing flood risk benefits and providing those benefits to persons that are socially vulnerable to flood hazards, all other indicators having zero weight in the scenario ("Scenario 2;" Fig. 3).

Results

In a series of meetings, we worked with the WRWC to implement the decision support tool. We applied many alternative planning scenarios to rank the candidate restoration sites based on their aggregated benefits. Each scenario incorporated a different weighting scheme for the ecosystem service benefits and social equity and reliability indicators. This process allowed our partners to identify a short-list of preferred restoration sites under various

Electronic Supplementary Material
The online version of this article contains supplementary material, which is available at: URL

planning scenarios. Four sites were among the highest ranking sites in most of the scenarios that we screened (Fig. 4).

We and the WRWC investigated the benefit indicator performance values of each of the four top ranking sites. The tradeoffs in performance values among the four sites were very close, leading to the conclusion that one site was not clearly better than the others. Because sites C and D are nearly adjacent, their indicator values are extremely similar (Fig. 4; Electronic Supplementary Material). Sites A and B, although not located near each other, provide similar flood risk and social equity benefits. These sites have similar access, site quality, natural substitutes, and social equity values, as measured by site size, area of wetlands within a 4 km buffer zone, and high level of social vulnerability to environmental hazards, respectively. With regard to beneficiaries, site A has the potential to impact more downstream addresses than site B, which is off the mainstem of the Woonasquatucket River (662 vs. 423). However, site B has no dams (built substitutes) while site A has three dams providing substitute flood protection within the 4 km buffer zone downstream of the site (Electronic Supplementary Material). In terms of overall ratings, these two sites are very similar. Looking at the values and tradeoffs across different indicators helped the WRWC to understand how these four top ranking sites received similar overall ratings but differed somewhat in terms of specific indicator values, although not enough to make any one site clearly superior.

Discussion

Multiple courses of action and the existence of tradeoffs makes good decision making extremely challenging and requires systematic, structured, and sometimes analytical models (Keeney 2004). An advantage of using MCDA is that they apply mathematics to compare numerous environmental management alternatives and make choices based on user-defined preferences. Using our decision support tool, benefit indicator values for each of the 65 candidate restoration sites were easily combined to rank choices. As a result, the WRWC was able to identify four candidate restoration sites (Fig. 4) as priority sites to put forward and discuss with other Rhode Island stakeholders if and when funding becomes available to implement wetland restoration. The indicator values and aggregated scores can be used in funding requests to demonstrate potential ecosystem service benefits and social equity concerns.

A key outcome of implementing the decision support tool with the WRWC is our finding that larger sites in the upper watershed scored lower than smaller sites in the more urban lower watershed; whereas with the functional assessment, the larger upper watershed sites scored higher (Golet et al. 2003). As one would expect, differences that were particularly large included the beneficiaries, access, and substitution indicator values. Because the magnitude of indicator values for the urban sites was much larger than that of the more rural sites, we and the WRWC were unable to conclude that the low ranking upper watershed sites were equivalent to or better than the high ranking urban sites. This post-hoc evaluation of the decision support tool further solidified the WRWC conclusion that high ranking sites based on social values are good first choices if and when funding opportunities are available to implement wetland restoration in the watershed.

It is important to note that our research provides an alternative way of evaluating the candidate restoration sites to that of the previous functional assessment of sites in the watershed (Miller and Golet 2001; Golet et al. 2003), and therefore site rankings differ. We and the WRWC tested this assertion by running multiple iterations of the decision support tool with various combinations of weights that resembled the single function and multiple function scoring assessments in the previous functional assessment. In the functional assessment, sites ranking high for a single function (flood abatement, visual accessibility, recreation, wildlife) were large in site area, distant from urban areas, or both, whereas sites that ranked high for multiple functions were all located in rural areas of the upper watershed.

Attempting to mimic these functional assessments with related weights for the objectives used in this study (i.e., ecosystem services, social equity), we showed that high ranking sites based on benefit indicators were almost always located in more urban areas of the watershed with a range of site sizes. These results illustrate the differences between focusing on functional assessment versus the social benefits of ecological restoration and support our previous hypothesis that smaller restoration sites in more urban areas can provide large ecosystem service benefits.

Typically, monitoring and evaluation are important parts of the SDM process to assure that actions taken are achieving the objectives (Fig. 1; Lyons et al. 2008). It is important to note the difficulty of monitoring in this context due to the challenge of measuring success in ecosystem service benefits received after implementing restoration. Traditional monitoring to evaluate restoration performance use biophysical metrics (SER 2004). By using social benefit indicators and metrics (e.g., number of potential beneficiaries), however, we lack information about how to measure effectiveness of restoration based on how ecosystem services are "used" or "enjoyed" by people. Future work on the case study will attempt to develop concepts and methods to support monitoring for the social benefits of ecological restoration in terms of who benefits and by how much.

Concluding Remarks

In this article, we combined previously developed concepts and methods for ecosystem services assessment with an SDM approach to illustrate methods for ecological restoration planning. Our approach is limited by the need to interact directly with user groups and the availability of funding and time to analyze ecosystem services at multiple candidate restoration sites and relevant scales. Despite these limitations, our approach is reproducible, even with limited resources or software capabilities, in most ecological restoration contexts with ecosystem service objectives and groups who desire to investigate multiple courses of actions among multiple sites in terms of how they meet the objectives.

This study tested whether new concepts and methods for ecosystem services assessment could provide useful information to support decision making. The incorporation of these methods into an SDM framework proved useful to all parties involved, particularly because the SDM framework is systematic and allows for rigor and transparency of each step in the process (Martinez-Harms et al. 2015). The WRWC participated and co-developed all parts of this application of SDM such that no key objectives or ecosystem services were left out

of the analysis. Based on this, we suggest that SDM could be a useful approach to provide salience, credibility, and legitimacy of information related to ecosystem service outcomes in a similar way that other science-policy frameworks aim to do (Posner et al. 2016).

Our approach proved useful to our case study partners. The WRWC greatly benefited from having the decision support tool to screen the 65 candidate restoration sites and deliberate on the potential benefits of the four highly ranked sites (Fig. 4). The WRWC is currently working with the Rhode Island Department of Transportation to consider drafting restoration designs around the sites identified as A, C, and D (Fig. 4) based on their flood risk and social equity values as well as site location preferences and priorities of the two organizations. This collaboration will help offset some of the costs of restoration for the sites and surrounding greenways. If information on the potential social benefits of restoration at the sites had not been available to the WRWC, this discussion may not have occurred and the offset in cost would not have been possible.

In sum, our approach was valuable for restoration planning in the watershed, and the WRWC and other Rhode Island groups are interested in using the approach for other types of environmental management, including green infrastructure planning. Based on feedback from our partners and giving many external presentations and webinars on the subject, the approach is currently being considered by other restoration, conservation, and green infrastructure groups in Rhode Island, Alabama, Florida, Massachusetts, and Puerto Rico. It is important to note that this project was limited in size, scope, and type of engagement. Larger projects, however, will be subject to significant stakeholder engagement, and concepts and lessons learned from implementing other frameworks for ecosystem services assessments (e.g., Verutes et al. 2017) are valuable to further develop and apply our methods.

Scientists, from both natural and social science fields, play a critical role in implementing a decision-focused approach to ecological restoration, especially if objectives are translated into ecosystem service performance metrics. We caution that it is important for scientists to maintain a clear separation of roles so that normative preferences are not inserted into actual decision making processes (Lackey 2016). We performed research to support a planning context; our study was not aimed at accomplishing restoration. If we take this into consideration, research using the ecosystem services concept can continue to provide a critical context for ecological restoration and support management decisions. Our approach can be useful in that endeavor.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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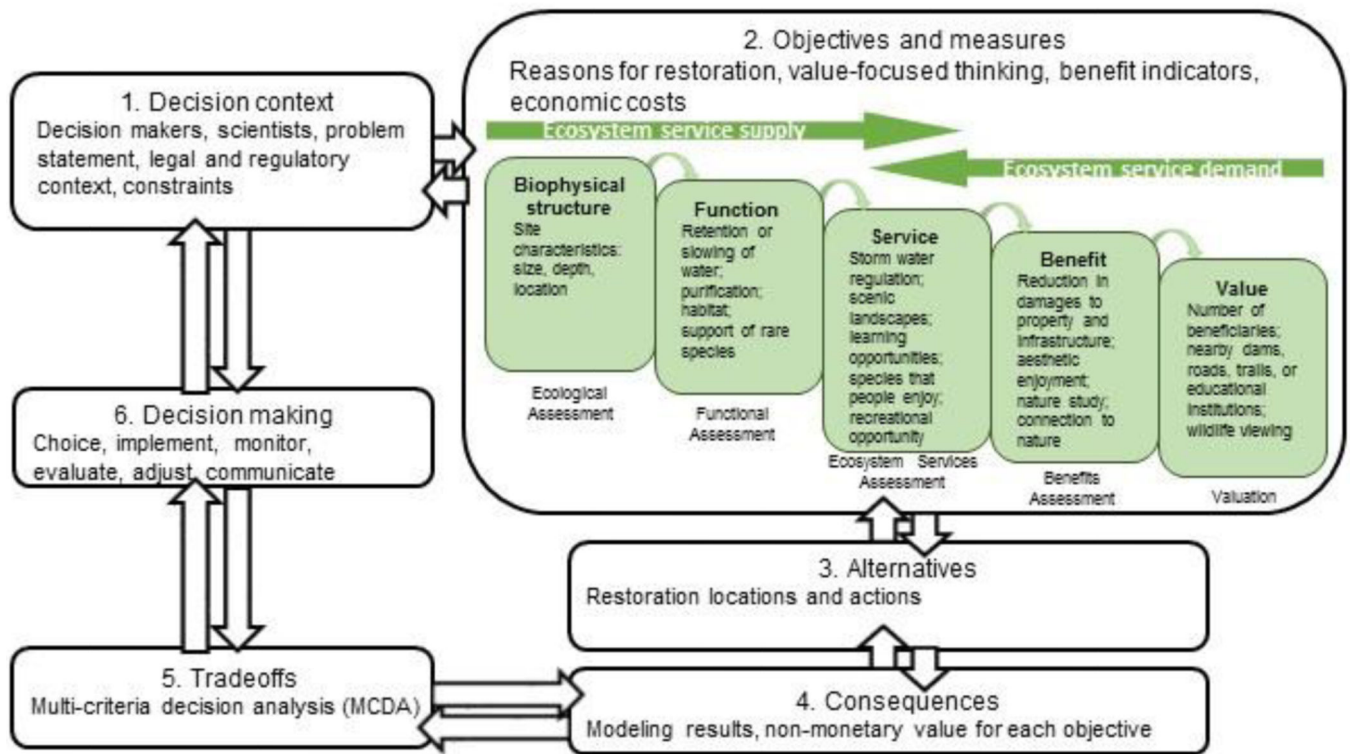
Fig. 1.

A modified structured decision making approach for estimating and evaluating the social benefits of ecological restoration (adapted from Potschin and Haines-Young 2011; Gregory et al. 2012). Ecosystem services associated with flood water retention, education, and recreation are used as example in the objectives and measures step.
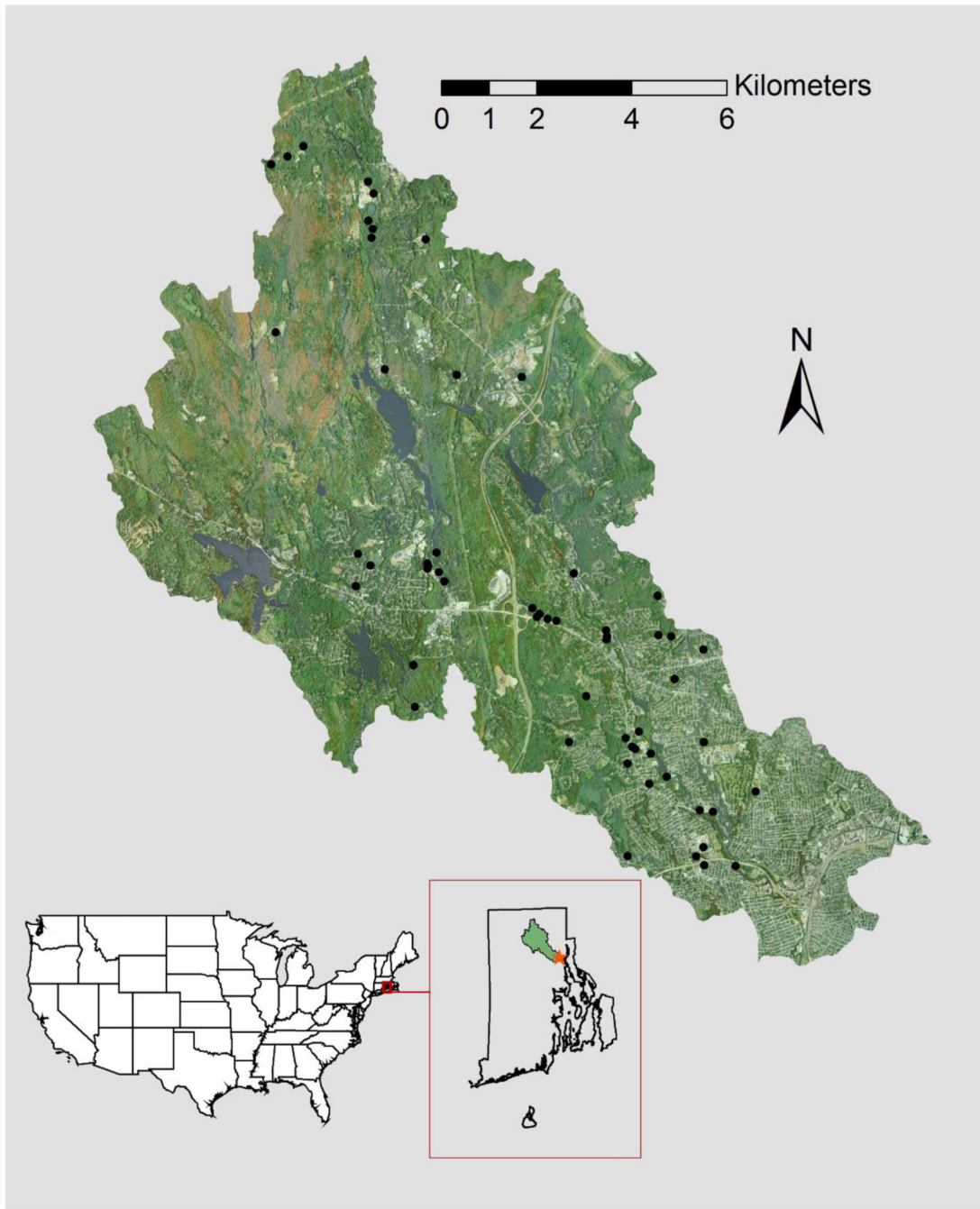
Fig. 2.
Woonasquatucket River watershed. The 65 candidate restoration sites we analyzed are shown as black points.

Fig. 3.

Screen shot of the spreadsheet decision support tool. The tool and data set are provided in the Electronic Supplementary Material.

Fig. 4.
Four highest ranking candidate restoration sites in numerous scenario iterations of the decision support tool. In the upper left panel, locations of the sites are identified by letter, the Woonasquatucket River is identified by a blue line, and locations of other candidate restoration sites in the vicinity are shown as black points. Polygons for each site are shown in the other panels. Site A is located on a former mill complex along the river and near an urban park and residential development. Site B is located near a tributary to the river; it is on public school property and near a golf course and residential development. Sites C and D are located along the river and adjacent to urban commercial and residential development. In addition to other scenarios, these sites are the highest ranking sites in terms of preferences for flood risk and social equity benefits (site A refers to site 455; site B refers to site 430; site C refers to site 425; site D refers to site 256; Scenario 2; Fig. 3).

Table 1.

Benefit indicators and measurements for the evaluation of candidate restoration sites

Category	Benefit indicator	Description/Metric	Goal	References
Flood risk	FR1 beneficiaries	Number of addresses in the floodplain within 4km radius and downstream from site	Maximize	Bousquin et al. (2015)
	FR2 quality	Size (hectares) of site	Maximize	
	FR3 substitution	Number of dams/levees 4km radius and downstream from site	Minimize	
	FR4 substitution	Percent area of wetlands within 4km radius of site	Minimize	
Scenic views	SV1 beneficiaries	Number of addresses within 50m of site	Maximize	Mazzotta et al. (2016)
	SV2 beneficiaries	Number of addresses between 50–100m of site	Maximize	
	SV3 access	Roads or trails within 100m of site	Yes	
	SV4 complementarity	Number of natural land use types within 200m of site	Maximize	
	SV5 substitution	Percent area of wetlands within 200m of site	Minimize	
Education	E1 beneficiaries	Number of educational institutions within 400m radius of site	Maximize	Mazzotta et al. (2016)
	E2 substitution	Percent area of wetlands within 800m of site	Minimize	
Recreation	R1 beneficiaries (walking)	Number of addresses within 536m of site	Maximize	Mazzotta et al. (2016)
	R2 beneficiaries (driving)	Number of addresses between 536–805m of site	Maximize	
	R3 beneficiaries (driving)	Number of addresses between 805m-10km of site	Maximize	
	R4 access	Bike trails within 536m of site	Yes	
	R5 access	Bus stops within 536m of site	Yes	
	R6 quality	Size (hectares) of site and adjacent green space	Maximize	
	R7 substitution	Percent area of green space within 1.1km but not adjacent to site	Minimize	
	R8 substitution	Percent area of green space between 1.1–1.6km but not adjacent to site	Minimize	
	R9 substitution	Percent area of green space between 1.6–19km but not adjacent to site	Minimize	
Bird watching	BW1 beneficiaries	Number of addresses within 322m of site	Maximize	Mazzotta et al. (2016)
	BW2 access	Roads or trails within 322m of site	Yes	
Social equity	S1 social vulnerability	Proximity-based percent of total area of social vulnerability to environmental hazards within 4km of site; based on demographics (e.g., race, class, wealth, age, ethnicity, employment) and other factors	Maximize[a]	Cutter et al. (2003); NOAA[b]
Reliability	RE1 reliability	Assurance that a site will continue to provide benefits over time, in the face of development stressors. Measured percentage of lands within 152m of site designated as conservation, parks & open space, reserve, or water land use categories	Maximize	Mazzotta et al. (2016)

[a]Highly vulnerable areas are less able to recover from environmental hazards because they are in lower income, ethnically diverse areas. We assume that a high social vulnerability index is preferable because the site provides access to people who are less able to access benefits otherwise.

[b]National Oceanic and Atmospheric Administration. Information at https://coast.noaa.gov/dataregistry/search/collection/info/sovi