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Capturing twenty years of change in ecosystem services provided by coastal Massachusetts habitats

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ABSTRACT

Accounting for ecosystem services across expansive and diverse landscapes presents unique challenges to managers tasked with navigating and synthesizing the social-ecological dynamics of varied stakeholder interests and ecological functions. One approach to this challenge is through expert based matrices that provide valuations for specific service-habitat combinations. In this study, we combine a literature review with local expert input to build an ecosystem service capacity matrix for the Massachusetts Bays National Estuary Partnership (MassBays). We then apply this matrix to a custom conglomerate land cover data set and a habitat connectivity analysis to assess the spatial and temporal dynamics in select ecosystem services of coastal habitats across MassBays from 1996 to 2016. In 1996, saltmarsh was the primary provider of coastal ecosystem services, representing roughly 60% of the total service capacity. More specifically, high elevation saltmarsh was top-ranked, followed by tidal flats, seagrass, low elevation saltmarsh and unclassified saltmarsh. This distribution of service provisioning varied considerably among the five regions of MassBays, reflecting the unique habitat mixes and local expert valuations of each. Although saltmarsh dominated the overall production of services, seagrass and tidal flats drove 97% of the service changes that occurred from one year to the next. From 1996 to 2016, MassBays lost 50% of its seagrass cover and gained 20% more tidal flats, resulting in a 5% overall loss in ecosystem services. Again, this varied among the five regions, with Cape Cod losing as much as 12% of a given service while the Upper North Shore gained 4% in services overall. We bootstrapped the analysis to provide a range of probable outcomes. We also mapped the changes in service production for each of the sixty-eight embayments. This analysis will aid local managers in accounting for ecosystem services as they develop management plans for their represented stakeholders.

1. Introduction

Landscape scale ecosystem management, or ecosystem-based management (EBM), is a methodology increasingly applied to maintaining and improving the condition of extensive landscapes that encompass a diverse array of habitats, species, and stakeholders (Arkema et al., 2006; Curtin and Prellezo, 2010). EBM aims to maintain ecosystems in a healthy and resilient condition while providing the services that humans want and need (Rosenberg and McLeod, 2005). Despite the theoretical and empirical success of this management strategy, implementation is often hindered by complexities associated with balancing the varied needs of different stakeholders, characterizing and monitoring the biological condition of expansive environments, and assessing the tradeoffs associated when managing multiple ecosystems (Arkema et al., 2006; Granek et al., 2010). The concept of ecosystem services in EBM is integral to connecting ecosystems with the wants and needs of society (O'Higgins et al., 2020). Identifying and quantifying the value of ecosystem services demonstrates the relevance of ecosystems to the public and decision-makers and is fundamental to understanding the capacity of ecosystems to produce benefits to people (DeWitt et al.,

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2020).

One of the fundamental challenges in any ecosystem services assessment is establishing an appropriate service valuation mechanism across the project area. Approaches to ecosystem service valuation are often in the form of monetary (e.g. $/ha yr) or ecological function metrics (e.g. tons N/ha yr), but other representations of value, both quantitative and qualitative, are also used (Kelemen et al., 2014; Kenter, 2016). However, drawbacks to many of these approaches often result in an inability to effectively translate ecosystem services into policy or action (Brown et al., 2014). Monetary valuations, despite their direct transfer to economic decision making, are often evaded for their anthropocentric approach and their inability to account for or the underestimation of non-monetary value (Chee, 2004; Ludwig, 2000; Rapport and Singh, 2006). Ecological function metrics, useful for their straightforward representation of service production values, are sometimes de-coupled from social use, benefit, economic value, and thus ultimately to service value (Peterson et al., 2010; Spangenberg et al., 2014). Other quantitative representations of ecosystem service value often rely on abstract social or ecological principals (e.g. emergy, Q-methodology, time-use studies etc.) (Kelemen et al., 2014; Kenter, 2016), making them less transmittable to stakeholders or policy makers. One method that avoids many of the above stated issues relies on matrices informed by local experts that directly link an ecosystem or habitat to a quantitative value of service capacity.

The ecosystem service matrix approach is a relatively simple means of establishing comparable values representing ecosystem service capacity per unit area for different ecosystems or habitats (Campagne et al., 2020; Jacobs et al., 2015). This approach ties each ecosystem to each service via expert knowledge, which may come in the form of estimates based on expert understanding, or actual in-situ measurements or studies that provide the relevant information (Jacobs et al., 2015). In either case, a "score" is provided that allows for the direct comparison to all other values in the matrix and thus the ability to boil down the tradeoffs and complexities associated with landscape scale ecosystem service analyses. Additionally, variability in expert scoring can be used to provide uncertainty estimates in final projections (Hattam et al., 2021). When paired with spatial data of ecosystem cover, the matrix approach can be used to analyze and map ecosystem services tradeoffs across both space and time and across a complex landscape of ecosystems (Jacobs et al., 2015). Although examples of both the matrix approach and spatial mapping of ecosystem services have been well documented in the scientific literature, it is also critical to ensure these methods are optimal for use in decision making, management action, and policy.

One critical review of ecosystem service assessments as they pertain to efficacy in decision making and policy implementation suggests that only matrices and spatial mapping facilitate instrumental decision making by presenting normalized indicators that allow for more straightforward comparisons across services and ecosystems (Wright et al., 2017). This same study and others also suggest that decision makers should play a more central role in scientific assessments, which allows for a sense of ownership and facilitates the interpretation of the often-complex results and conclusions of such studies (Guerry et al., 2015; Saarikoski et al., 2018). Further, a number of critiques point out the importance of including uncertainty in the analysis and emphasize that decision makers find it essential to reaching robust conclusions (Hamel and Bryant, 2017; Stritih et al., 2019). Thus, through the matrix approach, informed by local decision makers and combined with spatial mapping and the inclusion of uncertainty, ecosystem service assessments can be more effectively translated into management and policy action. Additional nuances, such as the inclusion of habitat connectivity analysis, may provide further confidence and efficacy in applying such assessments.

Landscape connectivity has long been hypothesized to play an important role in ecosystem service production, as many ecosystem services are dependent upon the transfer of organisms and materials between patches and between different habitats (Mitchell et al., 2013). Pollination is a clear example of this (Kremen et al., 2007), however the same is true for other services. Water quality depends upon the ability of water to move through different ecosystems, each performing a particular role in removing contaminants (Brauman et al., 2007). Recreation is bolstered when people and focal species can move easily across the landscape and between different ecosystems (van der Zee, 1990). Commercial fishing has also been shown to be enhanced when surrounding coastal ecosystems are more connected (Meynecke et al., 2008). Greater resilience from natural disasters, such as flooding, has also been connected to increased landscape connectivity (Su et al., 2018). As a result, several studies have incorporated connectivity into analyses of landscape ecosystem services capacity and production (Cicchetti and Greening, 2011; Frank et al., 2012; Grêt-Regamey et al., 2014; Peng et al., 2015).

This study brings together the above understanding of EBM, ecosystem service capacity matrices, and landscape connectivity to evaluate recent historical changes in ecosystem services for the Massachusetts Bays (MassBays) National Estuary Partnership (NEP). Using consultations with the partnership's regional coordinators and a conglomeration of national and local spatial datasets, our analysis explores the ways in which previous land use and land cover change influenced the capacity of ecosystem services across the NEP. Our study is meant as a conceptual demonstration for MassBays, to visualize current ecosystem services capacity in the context of what was possible historically, to help support target setting to restore or maintain desired levels of services, to weigh trade-offs in ecosystem services at multiple spatial scales, and to establish an approach by which to evaluate the potential effects of restoration and ecosystem management scenarios on ecosystem services. As this partnership and others within the National Estuary Program develop their Comprehensive Conservation and Management Plan (CCMP), an assessment of ecosystem services capacity can allow for more informed decisions that consider and communicate potential socio-economic benefits of restoration, and ultimately more effective implementation of plans.

2. Methods

2.1. Study location

The MassBays NEP is a partnership between sixty-eight local communities and the state and federal governments of the United States. Its mission is to protect, restore, and enhance the estuarine ecosystems of Ipswich, Massachusetts, and Cape Cod Bays in the state of Massachusetts. The partnership's jurisdiction encompasses over 1600 km of continuous coastline from Provincetown at the tip of Cape Cod, to the state's northern border with New Hampshire at Salisbury Beach. The member communities of the partnership represent an area of approximately 1775 km^2 and 1.7 million people (MassBays National Estuary Program, 2015). These communities are grouped into five geographic regions of Cape Cod, South Shore, Metro Boston, Lower North Shore, and Upper North Shore (Fig. 1). Each region and each community within the MassBays coastline represent a unique collection of stakeholders and interests concerning the use and management of the corresponding estuarine landscapes. Each of these regions is overseen by a regional coordinator, who works to bring together the region's various stakeholders and represent the NEP in matters of estuarine ecosystem management.

MassBays NEP has worked recently to update their comprehensive management plan (MassBays National Estuary Program, 2015) to include restoration targets for saltmarsh, seagrass, and tidal flats, based in part on examining historical trends in acres of habitat loss or gain (Cicchetti and Greening, 2011). A parallel understanding of the potential ecosystem services lost or gained historically can help to communicate potential benefits of restoration and inform management and restoration decisions to help achieve desired levels of benefits (Yee et al.,

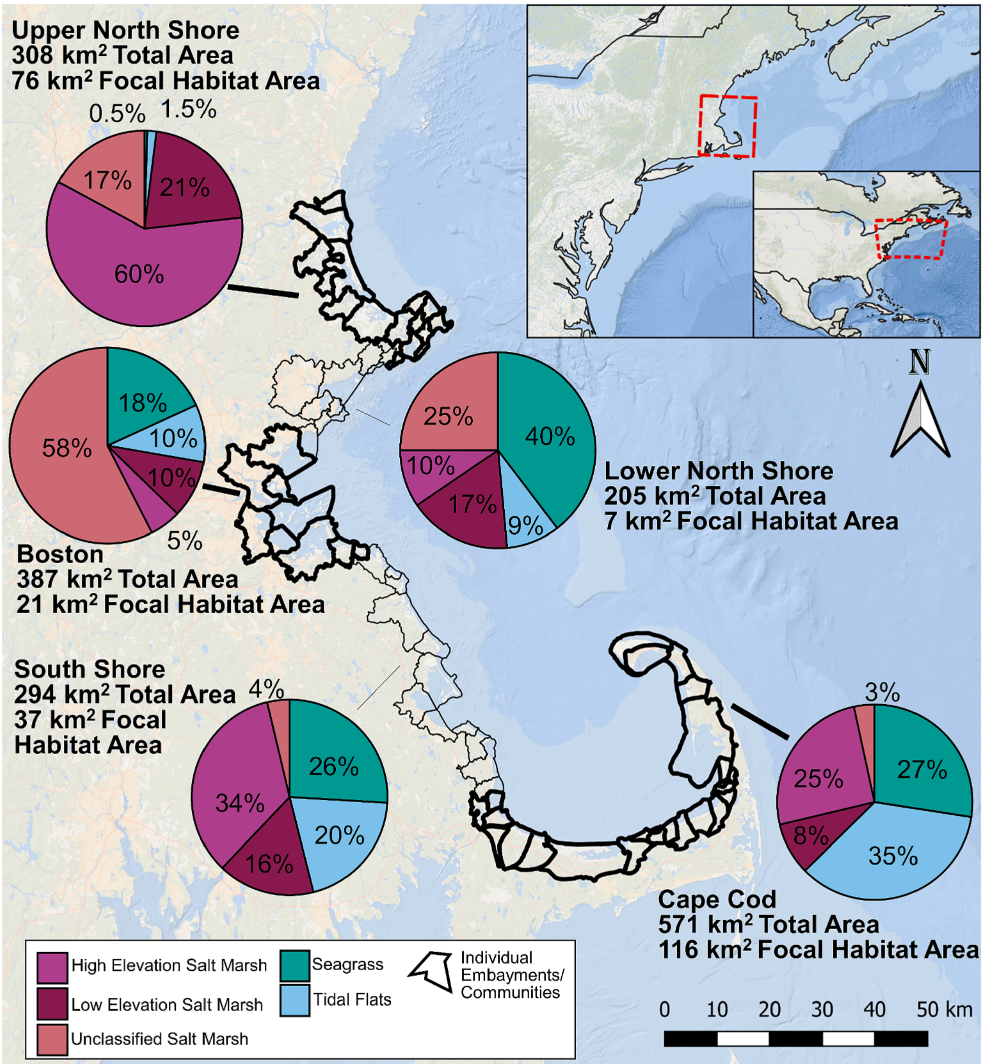

Fig. 1. The MassBays National Estuary Program encompasses 1770 km of coastline and is coordinated by five separate regions, each with a distinct coastal resource distribution (shown here as 1996 values) and stakeholder priorities. A regional coordinator from each region provided relative provisioning capacity scores for each focal coastal ecosystem (high and low elevation salt marsh, unclassified salt marsh, tidal flats, and seagrass), which was combined with the coverage and connectivity of those ecosystems to produce an overall ecosystem service provisioning value. Pie chart percentages are the percent of specific habitats that make up the total focal habitat area for each region.

2020).

2.2. Approach

We used a combination of spatial analysis, a literature review, and regional coordinator participation to quantify changes in ecosystem services capacity across various spatial scales and over time within MassBays (Fig. 2). For an ecosystem service capacity matrix, we combined an aggregate matrix from a literature review of available matrices, with a separate set of matrices informed by local experts. This expert input came from the MassBays regional coordinators and was used to refine the literature-derived matrices scoring each ecosystem according to its relative capacity per unit area to provide each service. The matrices were then combined with the land cover analysis to compute the total capacity for each service at various spatial scales (individual patches, embayments, regions, and NEP-wide) and for each year of spatial data availability. These values were then aggregated and analyzed across space and time to reveal temporal changes in ecosystem services capacity and to explore trade-offs in this capacity associated with past changes in habitat cover. This was done primarily with three focal ecosystems identified by the NEP: saltmarsh, seagrass, and tidal flats. As described later, saltmarsh was further divided into high-elevation, low-elevation, and unclassified components to better capture the different service capacities of these distinct hydro-geographic regimes. R scripts for the entire analysis, including the matrix synthesis, spatial dataset conglomeration, spatial analysis, and results reporting, can be found at the Open Science Framework repository https://osf.io/umrb3/.

3. Literature review of ecosystem services capacity

We conducted a literature review to collect values from published studies that used a matrix approach to evaluate the relative service capacity of various ecosystems per unit area (Table 1 and Supplementary Materials). As a previous study had already recently done this (Campagne et al., 2020), we used their listed studies and found any additional matrices using the same search terms and studies published since then (between 2020 and 2022). We searched in Google Scholar for the term (matrices OR matrix OR “look-up table”) AND (ecosystem) AND (service). From these combined matrices, we further refined the results to thirty-nine by eliminating studies that focused only on limited or very specific habitats (urban, benthic, agricultural etc.) or one type of ecosystem service (habitat, cultural, food etc.). A list of all studies considered in the literature review and reasons for exclusion can be found in the supplementary materials. We then standardized the terminology in all of the matrices to use a consistent set of terms for both habitats and ecosystem services, as described below.

For ecosystem service terminology, we chose the United States Environmental Protection Agency’s National Ecosystem Services Classification System (NESCS-Plus) (https://www.epa.gov/eco-research/na

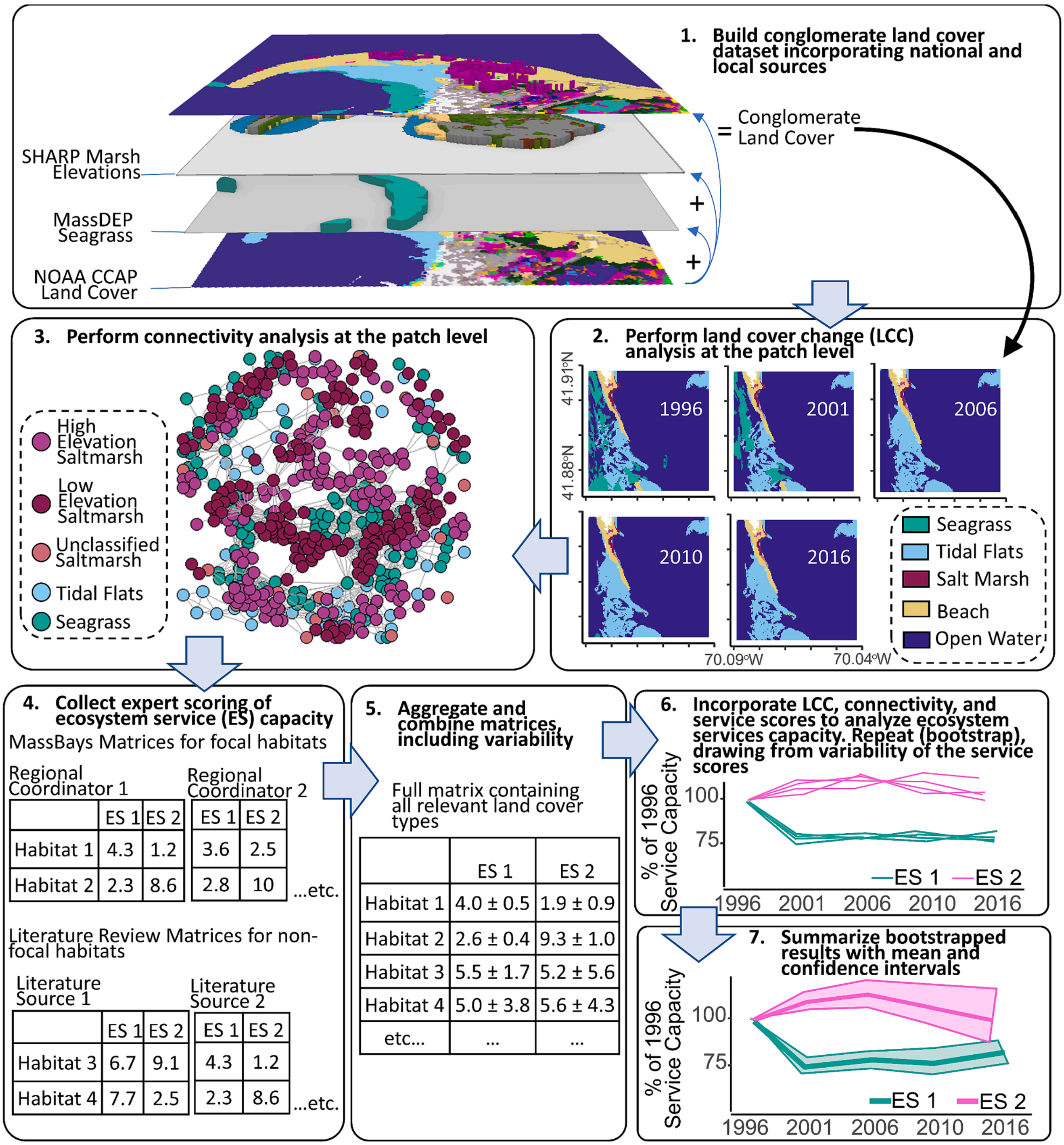

Fig. 2. A general overview of the approach to quantify ecosystem services capacity, which begins with a national dataset of land cover spanning twenty years from 1996 to 2016 and is modified to incorporate local datasets of focal ecosystems. This new conglomerate dataset is then used to conduct land cover change analysis and patch connectivity analysis, which are combined with expert scoring matrices of ecosystem service capacity to analyze spatial and temporal gradients in ecosystem services aggregated from patch to landscape scales.

tional-ecosystem-services-classification-system-nescs-plus) as a reference. NESCS Plus provides a structure of classes and sub-classes for identifying and comprehensively listing ecosystem services of direct relevance to the people who use them (Newcomer-Johnson et al., 2020). We used the fifty-six ecosystem attributes for ecological end products as our ecosystem services. These can be found in the "Services Classification Key" in the supplementary materials (S1). The regional coordinators were especially interested in a small subset of these services, which were edible or commercially important fauna (a combination of edible fauna and commercially important fauna), fauna community, water quality and water quantity. Additionally, the regional coordinators were interested in evaluating the services of climate change impacts mitigation and protection from extreme events and flooding, which were not explicitly stated in NESCS Plus and were thus added to this analysis. For ecosystems, we used the land cover classes provided by the Coastal Change Analysis Program (CCAP) (Office for Coastal Management, 2022), which includes the three primary focal habitats (saltmarsh, seagrass, tidal flats) as well as twenty-one additional classes. For this analysis, we refer to the CCAP classes of estuarine emergent wetland, estuarine aquatic beds, and unconsolidated shore as saltmarsh, seagrass, and tidal flats, respectively. The elevation varieties of saltmarsh were further distinguished as described below in the spatial

Table 1
Mean standardized ecosystem service capacity scores from various scientific literature sources for a selection of the non-focal habitats. Values represent the mean and standard deviation of the relative capacity of each habitat to provide the corresponding ecosystem service, with 0 being the lowest and 10 being the highest. These scores were used when one of these habitats was involved in a conversion with one of the focal habitats, whereas the focal habitats were scored using values from Table2. A full table with all habitats is provided in the supplementary materials S1.

	Climate Change Impacts Mitigation	Edible or Commercially Important Fauna	Fauna Community	Protection from Extreme Weather Events and Flooding	Water Quality	Water Quantity
Developed, High Intensity	0.6 ± 1.6	0.1 ± 0.5	0.5 ± 0.7	0.6 ± 1.2	0.7 ± 1	0.7 ± 1
Developed, Medium Intensity	0.5 ± 1.5	0.2 ± 0.8	0.9 ± 1.5	0.2 ± 0.6	0.4 ± 1.5	0.3 ± 1.3
Developed, Low Intensity	0.7 ± 1.2	0.4 ± 0.9	0.6 ± 1	0.8 ± 1.1	0.8 ± 1.4	0.5 ± 0.9
Developed, Open Space	1.7 ± 2.1	0.3 ± 0.9	1 ± 1.5	1.1 ± 1.3	1.1 ± 1.7	1.1 ± 1.7
Cultivated Crops	3.2 ± 2.6	1.5 ± 2.7	2.2 ± 2	2 ± 2.1	2 ± 2.3	2.2 ± 2.3
Pasture/Hay	4.1 ± 2.9	3 ± 3.8	1.5 ± 2.4	2.6 ± 2.7	2.9 ± 3	3.1 ± 3.1
Grassland/Herbaceous	4 ± 2.3	3 ± 3.6	3.7 ± 2.9	2.6 ± 2.2	3.9 ± 3.3	2.7 ± 2.7
Deciduous Forest	6.9 ± 3.5	3.3 ± 4	4.5 ± 3.1	5 ± 3.2	4.5 ± 4.2	3.8 ± 3.9
Evergreen Forest	6.5 ± 3.4	2.5 ± 3.5	3.2 ± 2.5	5.5 ± 2.9	4.2 ± 3.8	3.4 ± 3.6
Mixed Forest	6.8 ± 3.6	2.4 ± 3.5	6.7 ± 3.6	5.2 ± 3	5.4 ± 4.1	4.5 ± 4.1
Scrub/Shrub	4.8 ± 2.6	2.3 ± 3	4.5 ± 3.4	3.7 ± 2.7	4 ± 3.4	2.9 ± 3.1
Palustrine Forested Wetland	6.1 ± 3.7	1.9 ± 2.9	6.5 ± 3.2	4.4 ± 4.6	5.8 ± 3.5	5.1 ± 3.8
Palustrine Emergent Wetland	4 ± 3.4	1.2 ± 2.3	3 ± 3.6	3.8 ± 3.5	2.8 ± 3.5	3 ± 3.4
Estuarine Forested Wetland	6.1 ± 4.3	4.8 ± 4	5.7 ± 4.2	4.3 ± 4	3.7 ± 3.3	3.7 ± 3.7
Estuarine Scrub/Shrub Wetland	6.9 ± 2.9	4.8 ± 2.8	6.8 ± 3.1	6.7 ± 1.8	8.5 ± 1.7	7.7 ± 2.4
Barren Land	1.1 ± 1.7	0.4 ± 1.1	1.9 ± 1.9	2.6 ± 3.3	1 ± 1.5	1 ± 1.4
Open Water	3.5 ± 3.1	2.9 ± 3.6	4.3 ± 3.5	3.4 ± 3	3.6 ± 3.8	4.2 ± 4.3

analysis.

Each habitat and ecosystem service from the literature review was matched to its corresponding term from NESCS Plus and from the 24 NOAA CCAP land cover classes. Keys that cross-reference all original terminology for both habitat and ecosystem service with the above described classification systems can be found in the supplementary materials (S1). The resulting matrices provided at least one and up to 190 different values for nearly every habitat-ecosystem service pairing. The only habitat that was not accounted for in these matrices was tundra. For all study matrices, values were normalized and rescaled to a range of 0–10 through the equation

$$C_{hs} = 10\frac{C_o - MinC}{MaxC - MinC}$$

in which C_{hs} is the normalized, re-scaled service capacity score for a given combination of habitat (h) and service (s), C_o is the original value from the source matrix, and $MinC$ and $MaxC$ are the minimum and maximum values of the source matrix. All re-scaled values for each habitat-ecosystem service combination were averaged across the studies to produce the mean and standard deviation of the ecosystem services capacity value. The full literature review matrix, including mean, median, standard deviations, and number of observations for all habitat-service combinations can be found in the supplementary materials S1. This full literature review matrix was then used to supplement matrices developed by the regional coordinators, in which only a subset of coastal systems and relevant ecosystems services were evaluated.

3.1. Local evaluation of ecosystem services capacity

Because the five regions within MassBays represent distinct collectives of communities, stakeholders, and perspectives, as well as variability in coastal ecosystems, we invited each of the regions' coordinators to fill in their own version of a smaller, more focal matrix that reflects their local knowledge and priorities. First, MassBays NEP partners and regional coordinators narrowed down the full list of fifty-six ecosystem services classes to a subset of six of primary interest to MassBays. These were edible or commercially important fauna (a combination of edible fauna and commercially important fauna), fauna community, water quality and water quantity, climate change impacts mitigation, and protection from extreme events and flooding. Second, regional coordinators were invited to provide alternative matrices of scores representing local knowledge of the relative capacity to provide different services per unit area of each of the five focal habitats, including saltmarsh further classified by elevation (Table 2). Coordinators were asked to use a scale of zero to ten, with zero representing no capacity and ten representing the maximum capacity relative to the other ecosystems. Numbers could be repeated between ecosystems or a given service, meaning those ecosystems had the same capacity to provide that service. If a coordinator was unsure of any combination of ecosystem and service, they were asked to leave that score blank. We met with each coordinator to explain the process and to clarify any ambiguities that may have arisen. We used an example matrix to demonstrate and avoided discussing MassBays ecosystems in detail during the process, with the intention of allowing the regional coordinators to reflect their own knowledge of their region's systems when filling in the matrices. Coordinators were not able to see the scores from the other coordinators, again with the intention of allowing them to develop their own score based on their regions' unique qualities. The five regional scores for each habitat-service combination were averaged to create a single matrix of scores representing coastwide assessments from the combined input of the coordinators. This single regional coordinator vector (Table 2, rightmost column) was used to score areas of the focal habitats and the focal services of interest, while the other habitats involved in land cover change with focal habitats were scored from the matrix representing the literature review (Table 1 and supplementary materials). To account for the variability in scores from both the regional coordinators as well as the literature review, we repeated the analysis 100 times, each time drawing capacity scores randomly from normal distributions described by the means and standard deviations of the habitat-service combinations (Tables 1 and 2). The details regarding this spatial analysis are described in the following section.

Table 2
Ecosystem service provisioning capacity scores provided by the regional coordinators of the MassBays NEP. Coordinators were asked to provide the relative provisioning capacity for each service by each of the focal habitats, with 0 being the lowest and 10 being the highest. Each regional coordinator provided values for their region only, but these were averaged to provide a single service-habitat score and its standard deviation that was used in the analysis. Blank cells represent values that regional coordinators were not comfortable providing. This table's values were used to score services capacity in the focal habitats only, with all other habitats using the values from Table 1.

Ecosystem Service	Focal Habitat	Upper North Shore	Lower North Shore	Boston	South Shore	Cape Cod	Aver. ± Stand. Dev.
Climate change impacts mitigation	High Elevation Saltmarsh	8	8	8.5	9	10	8.7 ± 0.8
	Low Elevation Saltmarsh	5	4	7.5	8	9	6.7 ± 2.1
	Unclassified Saltmarsh	6.5	6	8	8.5	9.5	7.7 ± 1.4
	Seagrass	7	10	8.5	8	1	6.9 ± 3.5
	Tidal Flats	4	2	1	6	10	4.6 ± 3.6
Edible or Commercially important fauna	High Elevation Saltmarsh	5	6	1	2	9	4.6 ± 3.2
	Low Elevation Saltmarsh	8	7	5	4	10	6.8 ± 2.4
	Unclassified Saltmarsh	6.5	6.5	3	3	9.5	5.7 ± 2.8
	Seagrass	7	8	5	7	10	7.4 ± 1.8
	Tidal Flats	6	10	2	8	10	7.2 ± 3.3
Fauna community	High Elevation Saltmarsh	9	8	1.5	4	10	6.5 ± 3.6
	Low Elevation Saltmarsh	7	8	2.5	6	10	6.7 ± 2.8
	Unclassified Saltmarsh	8	8	2	5	10	6.6 ± 3.1
	Seagrass	6	10	5	7	10	7.6 ± 2.3
	Tidal Flats	5	5	2.5	7	10	5.9 ± 2.8
Protection From Extreme weather events and Flooding	High Elevation Saltmarsh	4	10	7.5	9	10	8.1 ± 2.5
	Low Elevation Saltmarsh	4	8	5	8	10	7 ± 2.4
	Unclassified Saltmarsh	4	9	6.25	8.5	10	7.6 ± 2.4
	Seagrass	6	5	2.5	2	1	3.3 ± 2.1
	Tidal Flats	2	2	1.5	7	9	4.3 ± 3.5
Water quality	High Elevation Saltmarsh	7	8	2.5	9	9	7.1 ± 2.7
	Low Elevation Saltmarsh	3	4	3.5	8	10	5.7 ± 3.1
	Unclassified Saltmarsh	5	6	3	8.5	9.5	6.4 ± 2.6
	Seagrass	6	4	5	6	5	5.2 ± 0.8
	Tidal Flats	3	6	0	5	5	3.8 ± 2.4
Water quantity	High Elevation Saltmarsh	2		1.5	1		1.5 ± 0.5
	Low Elevation Saltmarsh	2		2	1		1.7 ± 0.6
	Unclassified Saltmarsh	2		1.75	1		1.6 ± 0.5
	Seagrass	2		2.5	1		1.8 ± 0.8
	Tidal Flats	2		7.5	1		3.5 ± 3.5

3.2. *Spatial analysis*

All spatial analyses were carried out in R (R Core Team, 2021). To characterize the land use and land cover across the MassBays landscape and to quantify the changes in this landscape across space and time, we used a variety of local and national spatial datasets (Table 3). As described in more detail below, these datasets were merged within their corresponding timeframes to create unique combinations that optimized both the areal coverage and the classification accuracy, especially of the five focal ecosystems. As a base layer, we used CCAP GeoTIFFs, 30 m horizontal resolution land cover images with twenty-four classes covering a twenty-year timespan represented by 1996, 2001, 2006, 2010, and 2016. This dataset was used both for its temporal and spatial coverage, as well as for its representation of distinct coastal ecosystems, including estuarine aquatic beds (seagrass), unconsolidated shoreline (tidal flats), and estuarine emergent wetlands (saltmarsh). However, to improve the accuracy of these classifications, the CCAP layers were updated with additional datasets for seagrass and saltmarsh classifications.

Because submerged aquatic vegetation is often difficult to classify

Table 3
Spatial data used in the analysis to classify focal coastal habitats throughout the study area.

Dataset	Description	Spatial Resolution	Temporal Coverage	Source
NOAA CCAP	Land use and land cover represented by 26 classes	30 × 30 m	1996,2001,2006,2010 and 2016	(Office for Coastal Management, 2022)
Seagrass Cover	Seagrass cover through in-situ surveys	NA	1995, 2001, 2006–2007, 2010–2013, 2015–2017	(Costello & Kenworthy, 2011; MassGIS, 2020)
SHARP	Saltmarsh elevation classes	3 × 3 m	2014	(SHARP, 2017)

through remote sensing (Rowan and Kalacska, 2021), all seagrass classified pixels within the CCAP dataset were updated with the Massachusetts Department of Environmental Protection's (MassDEP) Eelgrass mapping project, which covers roughly the same years as CCAP (Costello and Kenworthy, 2011; MassGIS, 2020). This project uses both aerial imagery and on-site surveys to identify both the dominant species, eelgrass (*Zostera marina*), as well as the more elusive species, widgeon grass (*Ruppia maritima*). Original seagrass bed delineations were carried out via human identification from aerial photos for years 1994, 1995, 1996, 2000, 2001 and 2002, as well as digital imagery for years 2002, 2006, 2007, 2010, 2012, 2013, 2015, 2016 and 2017. On site surveys were also carried out to rectify questionable or otherwise non-visible areas and to provide accuracy assessments throughout the remotely sensed delineations. These were done using submerged video equipment towed from small boats. Although delineations for a given year do not always cover the entire study area, multiple contiguous years do for all years except 2006 and 2007. The 2006–2007 surveys did not cover all of the Cape Cod or Lower North Shore Regions. We therefore removed seagrass from the analysis and results for these regions in the 2006 analysis. In all other years, we used multiple seagrass surveys to represent one year of CCAP data, ensuring a complete coastal survey for each. Delineations for 1995, 2001 and 2006 were used to replace CCAP data from 1996, 2001, and 2006, respectively. Surveys from 2010 to 2013 were used to represent the CCAP data from 2010, and those from 2015 to 2017 were used to represent CCAP data from 2016. To do so, we assumed that any seagrass classified pixels from CCAP were either truly seagrass, in which case the MassDEP data would validate, or they were areas of shallow unvegetated open water that were erroneously identified as seagrass. We therefore converted all seagrass pixels to the open water classification and then used the *mask* function from the raster package in R (Hijmans, 2016) to update all pixels covered by the MassDEP delineations.

For saltmarsh, although there were no known issues with the CCAP classifications, input from the regional coordinators suggested there should be a distinction between high and low elevation marshes, which have different service capacities and different vulnerabilities. To accomplish this, we used marsh zonation maps produced by the saltmarsh Habitat and Avian Research Program (SHARP) (SHARP, 2017). These images utilize local tidal gauges in combination with digital elevation models (DEM) (when available) and existing saltmarsh delineations to characterize saltmarsh according to its probable tidal inundation characteristics. High marshes are those that are only flooded by average or larger than average tides, roughly every week or month. Low marshes are those flooded daily by regular tides. In addition, this dataset also classifies areas as salt pools, terrestrial borders, mudflats, open water, non-marsh uplands, and finally stands of the non-native *Phragmites australis*. We updated the existing CCAP data with the SHARP data by first vectorizing both via the *st_as_stars* function from the stars package in R (Pebesma, 2020). We then updated the CCAP marsh areas with the classification from the SHARP data via the *st_intersection* function from the sf package. In many cases, the same CCAP marsh area intersected with classifications that came from SHARP data that was generated with and without a DEM. In those cases, only the DEM classification was used. The resulting dataset represented the original CCAP marsh delineations but was now split into the different classifications represented by the SHARP data, or remained unclassified if no SHARP data coincided with a CCAP marsh. For this study, the resulting classifications were aggregated into high marsh, low marsh, and unclassified marsh. High marsh represented the SHARP classifications of high elevation marsh, upland, and terrestrial borders. Low marsh represented the classifications of low elevations marsh, mudflat, salt pool, and open water, while unclassified marsh remained as a pre-existing CCAP marsh area that did not overlap with the SHARP data. In this way, although the SHARP data may have identified an area as non-marsh (upland, open water, mudflat etc.) we retained the CCAP marsh classification but modified its elevation accordingly. Further, because the SHARP data only represented one year (2014), we assumed there were negligible changes in marsh elevations from 1996 to 2014 and from 2014 to 2016, and thus repeated this process for all years of the CCAP data.

The result of the above were five raster images, one from each of the CCAP years, representing mostly the original CCAP data but with the seagrass pixels reflecting only the MassDEP data and most of the saltmarsh pixels being reclassified as either high or low elevation. These five datasets were then vectorized via the *st_as_stars* function and contiguous habitat patches were given unique identifiers. Contiguous patches in this case were defined as those sharing any edge or corner. These final datasets were used to track changes in ecosystem cover and patch connectivity across space and time. We did this at the individual patch level, which allowed us to scale up to embayment, regional, and NEP levels if needed, but also to identify specific areas of interest within individual embayments at which potential management could be focused.

To track ecosystem cover with time, we used the *exact_extract* function from the exactextractr package in R (Baston, 2021). Using the 1996 patch delineations, we calculated the percentage of all other land cover classes within these original 1996 areas for every subsequent year (2001, 2006, 2010 and 2016). This provided a timeseries of any land cover conversion occurring within each individual patch, and this timeseries could then be scaled up to the community, regional, and NEP levels. We also calculated patch connectivity at the embayment level via the lconnect package (Mestre and Silva, 2019). The *con_metric* function was used to calculate the integral index of connectivity (IIC) for all patches of the same habitat within or intersecting with a given embayment (Pascual-Hortal and Saura, 2006). While many studies define connectivity differently, the (IIC) is a graph-theory based approach that has been demonstrated by Pascual-Hortal and Saura (2006) to be a robust single measure of landscape connectivity. We chose a patch distance threshold, the distance at which patches are no longer connected, as 500 m based on previous studies (Du et al., 2018; Engelhard et al., 2017).

4. Ecosystem service capacity

We evaluated ecosystem services capacity at the patch level as a combination of coverage area, intra-habitat connectivity, and literature-based and expert-derived capacity scores for each ecosystem and each service. As a base capacity value, we multiplied each patch area of a habitat by its capacity score per unit area (Tables 1,2). In this way, service capacity increases linearly with both habitat coverage and the capacity score. To incorporate connectivity, we added a percent increase in the base capacity value proportional to the connectivity index (IIC) for a given habitat in each community. Thus, a habitat with an IIC of 0.5 increases its capacity by 50% over the base value. This is represented by the following equation:

$$ESP_{hs} = A_h C_{hs} + A_h C_{hs} I_{he} \tag{1}$$

where h is a given habitat, s is a given service, e is a given embayment,ESP is the ecosystem service capacity, A is the coverage area of a patch, C is the matrix capacity score per unit area, and I is the IIC index. Further, the corresponding change in service capacity from one year ($t1$) to the next ($t2$) can be represented by the following equations:

$$\Delta ESP_{hs} = ESP_{hs_{t2}} - ESP_{hs_{t1}} \tag{2}$$

$$\Delta ESP_{hs} = (A_{h_{t2}} C_{hs} + A_{h_{t2}} C_{hs} I_{he_{t2}}) - (A_{h_{t1}} C_{hs} + A_{h_{t1}} C_{hs} I_{he_{t1}}) \tag{3}$$

$$\Delta ESP_{hs} = (A_{h_{t1}} + \Delta A_h) C_{hs} + (A_{h_{t1}} + \Delta A_h) C_{hs} I_{he_{t2}} - A_{h_{t1}} C_{hs} - A_{h_{t1}} C_{hs} I_{he_{t1}} \tag{4}$$

$$\Delta ESP_{hs} = \Delta A_h C_{hs} + A_{h_{t1}} C_{hs} (I_{hc_{t2}} - I_{he_{t2}}) + \Delta A_h C_{hs} I_{he_{t2}} \tag{5}$$

$$\Delta ESP_{hs} = (\Delta A_h C_{hs}) + (A_{h_{t1}} C_{hs} \Delta I_{he} + \Delta A_h C_{hs} I_{he_{t2}}) \tag{6}$$

Equation (6) can be further reduced to distinguish the change in service capacity due to the change in areal coverage (left most pair of parentheses) and that due to the change in connectivity (right-most pair of parentheses). Equation (6) describes the total change in service capacity from $t1$ to $t2$ for habitat h. It is also possible to expand equation (6) into the respective portions of capacity change attributed to each of the habitats in which h was converted to, such that:

$$\Delta ESP_{hs} = \sum_{i=1}^{n} \Delta ESP_{is} \tag{7}$$

where i is one of the n separate habitats to which h was converted from times $t1$ to $t2$. However, while the term ΔA_h in equation (6) can be made to represent only the land converted from h to i, there is no way to determine exactly how much of the term ΔI_{he} (the change in connectivity for habitat h in embayment e from $t1$ to $t2$) is attributed to any given replacement habitat i. Therefore, we approximated this partial change in connectivity as being equivalent to the proportion of total land converted from habitat h to habitat i. Thus, if the conversion from habitat h to habitat i represented 40% of the total land converted from habitat h, the portion of ΔI_{he} attributed to the conversion to habitat i was also 40%. Here, there is a distinction between gross and net land conversion. Whereas net conversion of habitat h could be zero, resulting in no ability to calculate the proportions of ΔI_{he}, using the gross land conversion avoids this issue. As an example: of ten km^2 of seagrass lost from 1996 to 2001, seven km^2 were converted to open water, three km^2 were converted to tidal flats, one km^2 was converted to salt marsh and another one km^2 of saltmarsh was converted to seagrass. As a result, a combined 12 km^2 of seagrass was involved in a conversion and the IIC for seagrass fell from 0.4 to 0.3. Thus, the ΔI_{he} of -0.1 for seagrass can be attributed to each of the conversion habitats as:$0.1\left(\frac{-7}{12}\right)+0.1\left(\frac{-3}{12}\right)+0.1\left(\frac{-1}{12}\right)-0.1\left(\frac{-1}{12}\right)$, for open water, tidal flats, the loss to salt marsh, and the gain from salt marsh, respectively. Using the gross conversion to and from salt marsh allows for the calculation of the individual contribution to ΔI_{he}, even though the net conversion for this habitat was 0.

We used equation (6) to track how service capacity changed for each habitat in each community throughout the study period, as well as the relative amounts of these changes attributed to both changes in coverage area as well as connectivity. We also used equation (7) to track where and how much of service capacity was distributed when habitat conversions took place. To account for the variability in service capacity scores from the literature review, we bootstrapped the analysis by drawing random values for the service capacity scores from normal distributions with the same mean and standard deviations presented in Table 1. We assigned these values to each pixel of a given habitat patch and then calculated the resulting sum in service capacity from equation (6). We repeated this process 100 times to produce a distribution of values and then report on the mean and standard deviation of this distribution. Because our raw values have no units by which to make comparisons with other metrics of ecosystem services, we report only the relative differences in the values between habitats, regions, and years. Thus, if a different metric of service value is otherwise known, these relative differences can be used to estimate how such a metric might change as a result. Finally, to determine the overall trends in area-to-service conversion rates, we fit simple linear regression models of the change in area versus the resulting change in service production for each habitat-to-habitat conversion. We report the slopes of these models as an indication of the overall proportional gain or loss in services for a given conversion in habitat area in any given location of the MassBays NEP. Only models with p-values less than 0.05 are reported.

5. Results

5.1. Changes in MassBays coastal habitats

In 1996, the embayments of MassBays contained a combined 258 km^2 of saltmarsh, seagrass, and tidal flats. Saltmarsh represented 61 % of this total area at 157 km^2, which was further divided into 89 km^2 of high elevation, 35 km^2 of low elevation, and 32 km^2 of unclassified saltmarsh. Tidal flats occupied 52 km^2 of this combined area and seagrass occupied 49 km^2. By region, Cape Cod, the Upper North Shore, and the South Shore held 45 %, 29 %, and 14 % of all focal ecosystem area, respectively, while Boston held 8 %, and the Lower North Shore held 3 %. For marsh, the Upper North Shore, Cape Cod, South Shore, and Boston area held 47 %, 28 %, 13 %, and 10 % of the total area, respectively, with the Lower North Shore holding the remaining 2 %. For tidal flats, Cape Cod and the South Shore held 78 % and 14 %, respectively, with the Upper North Shore, Boston and the Lower North Shore holding 2 %, 4 %, and 1 %, respectively. In seagrass, Cape Cod, the South Shore and Boston held 66 %, 20 %, and 8 %, respectively, with the Lower North Shore and Upper North Shore holding 6 % and 1 %, respectively.

From 1996 to 2016, the coverage of all focal ecosystems declined 5 % across the NEP area, however, the change in habitat cover varied by habitat and region (Fig. 3). Seagrass lost 50 % of its 1996 area, while all marsh classes lost 0.1 % and tidal flats gained 20 %. The greatest percent decline in total focal ecosystem cover occurred in the South Shore, which lost 13 % of its focal ecosystem cover attributed almost entirely to a 5 km^2 (54 %) loss of seagrass. Cape Cod, however, experienced the greatest total loss in focal ecosystem cover, about 12 km^2, due primarily to a 57 % (18.2 km^2) decline in seagrass that was partially offset by a 6.2 km^2 (15 %) gain in tidal flats. The Lower North Shore and the Boston area each lost around 4–5 % in total coverage, again attributed primarily to a 12 % and 14 % loss in seagrass, respectively. In contrast, the Upper North Shore experienced a 5 % gain in focal habitat cover, driven primarily by a 330 % increase in tidal flats (4 km^2).

For all regions, the majority of focal ecosystem area converted from 1996 to 2016 was to a non-focal ecosystem, primarily open water, which accounted for 77 % (25 km^2) of the conversions across the study area (Fig. 3). This was followed by tidal flats, which accounted for another 16 % (5 km^2) of focal ecosystem conversions. By region, open water accounted for 92%, 90%, 84%, 73% and 33% of the focal ecosystem conversions for the Lower North Shore, South Shore, Boston, Cape Cod, and the Upper North Shore, respectively. The largest conversions from 1996 to 2016 occurred as seagrass to open water, with 25 km^2 converted in total across the NEP and 16 km^2, 6 km^2, 2 km^2 in Cape Cod, the South Shore, and Boston, respectively. Across the study area, developed land accounted for less than 0.2 % of focal ecosystem conversions, with the highest percentage occurring in Boston (0.9 %) and the Lower North Shore (0.8 %).

5.2. Ecosystem services capacity across MassBays

Across the NEP in 1996, the relative provisioning of all combined services by the focal habitats reflected their distribution across the landscape. The results in this section reflect the means of the 100 bootstrap iterations performed using the distributions of ecosystem service scores from both the regional coordinators and the literature review. Bootstrapped ES capacity values were almost always within 10% of mean value; for more details on variability among bootstrapped iterations (minimum, maximum, standard deviation) see Supplementary Materials S1. The most abundant habitat, high elevation saltmarsh, ranked the highest (36.3 %) in providing all combined ecosystem services. This was followed by tidal flats (19.8 %), then seagrass (18.1 %), low elevation saltmarsh (13.4 %), and unclassified saltmarsh (12.4 %) (Fig. 4). For individual services, however, there was some trading ranks among the habitats. High elevation saltmarsh was the primary provider of all individual services except water quantity and edible or commercially important fauna, in which tidal flats ranked highest. In turn, seagrass was second in providing all other services except water quantity, edible or commercially important fauna, and protection from extreme weather events and flooding. Tidal flats followed as the third highest capable habitat for half of the services except edible or commercially important fauna, protection from extreme weather events

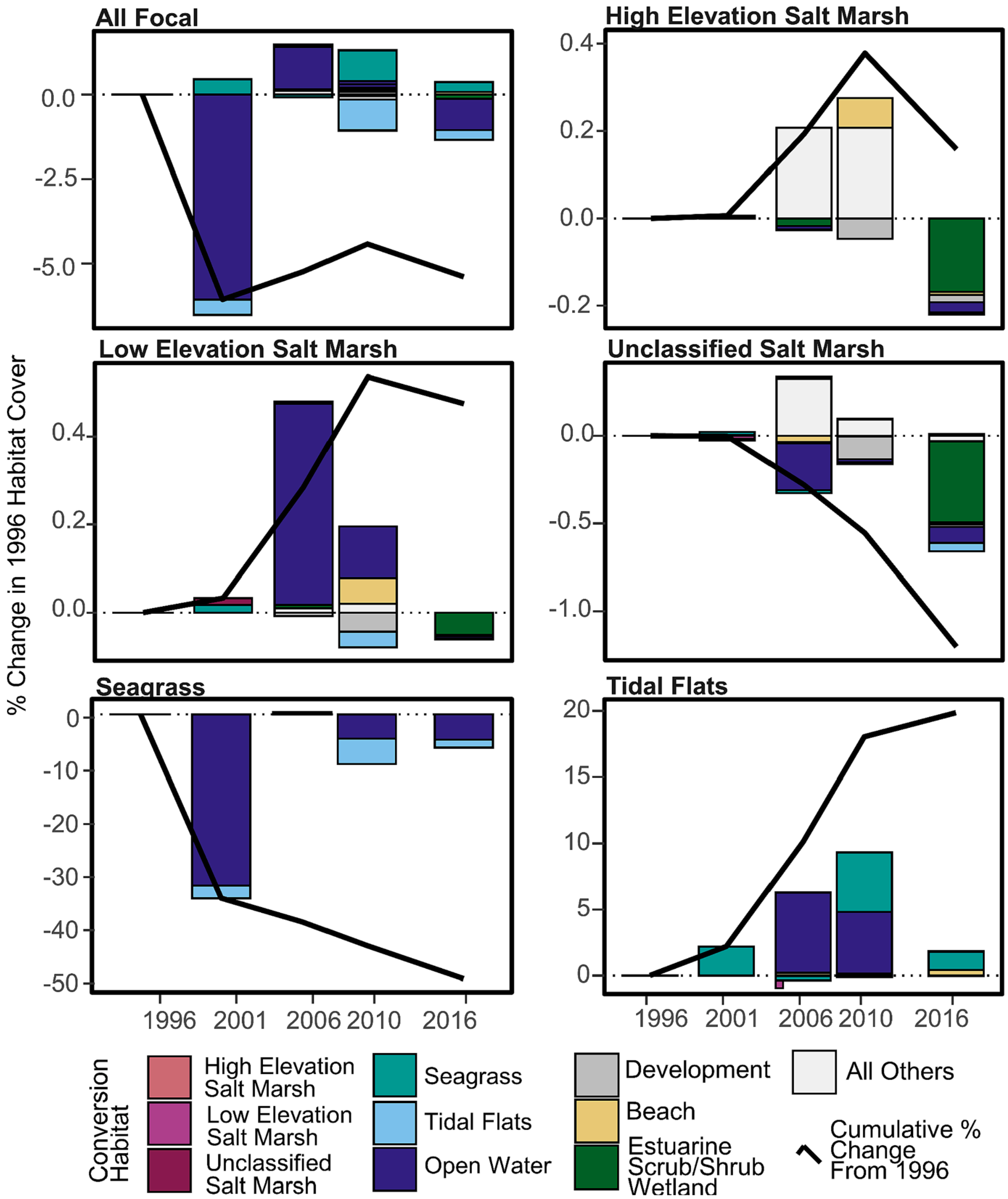

Fig. 3. Land use land cover change among all combined focal habitats across the five regions of the MassBays NEP. Bars represent single time-step conversions relative to the original 1996 values. Positive bars signify an increase in focal habitats from the conversion habitat. Negative bars are a loss in focal habitats to the conversion habitat. Stacked bars are additive and the total net height (positive plus negative) is the total conversion of the focal habitat for that year. Lines represent the cumulative change in focal habitats relative to the original 1996 coverage.

and flooding, and water quantity. Low elevation saltmarsh and unclassified saltmarsh traded places for the fourth and fifth most capable habitats for all individual services except protection from extreme weather events and flooding, in which seagrass was the last ranked habitat.

The above rankings changed considerably within individual regions. In 1996 in the Upper North Shore, all services were provided almost exclusively by saltmarsh (98 %), with high elevation marsh providing 60 % of a given service. In the Lower North Shore, seagrass provided 40 % of all combined ecosystem services and was top ranked for all services except protection from extreme weather and flooding, in which unclassified saltmarsh ranked highest. In Boston, unclassified saltmarsh provided the most (59 %) of all combined services and was also top-ranked in all individual services, providing 58 %. Seagrass came in second (18 %) in all services combined as well as in all individual services (18 %), except protection from extreme weather and flooding, in which it was ranked third (10 %. Low elevation saltmarsh was the third most provider of all combined services (10 %) and many of the individual services (9 %). Tidal flats were fourth in all combined services (8 %) in combined and individual (9 %) services, while high elevation saltmarsh was last (5 %) in combined and many of the individual services (5 %).

In the South Shore, high elevation saltmarsh was top-ranked in all combined services provisioning (37 %) as well as in all individual services except water quantity and edible or commercially fauna, which were provided primarily by tidal flats and seagrass, respectively.

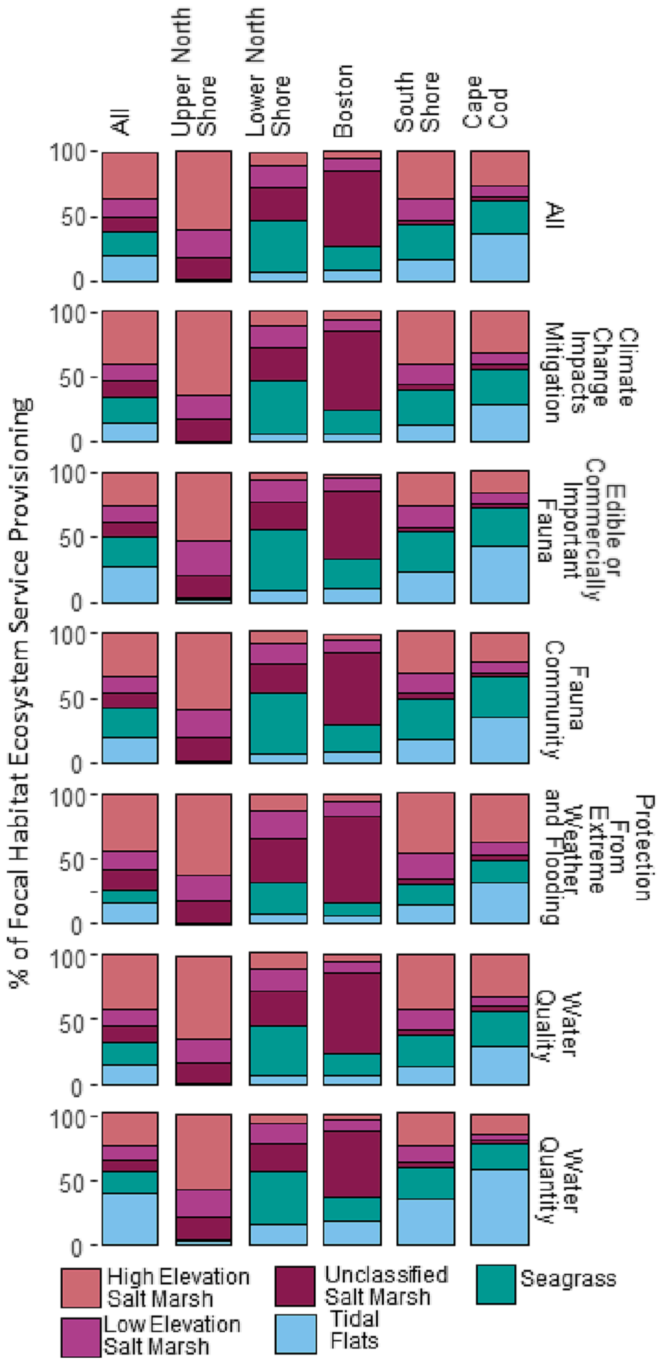

Fig. 4. 1996 distribution of ecosystem services provisioning among the focal habitats of the MassBays NEP regions based on the mean service capacity scores provided by the local experts. Saltmarsh provided the majority of services across the NEP.

Seagrass ranked second (26 %) in all combined services, as well as in water quantity (25 %), climate change impacts mitigation (27 %) and fauna community (31 %), as well as highest in edible or commercially important fauna (32 %). Tidal flats in the South Shore ranked third (17 %) in all combined services provisioning, as well as in edible or commercially important fauna (23 %) and fauna community (18 %).

Finally, in Cape Cod, tidal flats ranked highest (36 %) in providing all combined services, as well as in edible or commercially important fauna (43 %), water quantity (58 %) and fauna community (35 %). High elevation saltmarsh was second (27 %) in all combined services provisioning, and was first in water quality (29 %), climate change impacts mitigation (28 %) and protection from extreme weather events and flooding (32 %).

6. Loss or gain in ecosystem services due to habitat conversions

The highest exchange rates, those that produced the greatest net increase in service capacity for each % of the collective total land area converted, often involved a conversion to tidal flats (Table 4). For edible and commercially important fauna, conversions from beach to tidal flats produced 4.9 times more capacity for every % of the area converted. For climate change impacts mitigation, the conversion rate was 2.5 times more services for the 1 km^2 of beaches converted to tidal flats across the NEP. Substantially more seagrass was converted to tidal flats (12 km^2) and these conversions also often resulted in high exchange rates. For climate change impacts mitigation and protection from extreme weather events, the exchange rates of 0.4 and 1.3 resulted in 4.8 and 15.6 times more overall capacity for these converted patches, respectively. On the other hand, the 59 km^2, gross, of open water collectively converted to seagrass across the NEP, had a conversion rate of 1.6, resulting in 94.4 times more edible or commercially important fauna capacity. The reverse of these conversions, when applicable, also resulted in net losses of services provisioning equal to the negative of the above conversion rates.

From 1996 to 2016, there was a 5 % decline in focal ecosystem services capacity across all combined regions and services (Figs. 5 and 6). This loss was primarily a result of a 49 % loss in the services provided by seagrass, which was partially compensated by a 21 % gain in those of tidal flats and further buffered by saltmarsh, which changed by less than 1% from 1996 to 2016 (Fig. 6). However, there were more substantial changes within regions and in specific services (Supplementary S2, Fig. 1). In the Upper North Shore, combined services in focal ecosystems rose by 4 %, with all services increasing by 2 %. Most of these gains were made from 2001 to 2006 by an increase in tidal flats. In the Lower North Shore, all combined services dropped 2 %, with the greatest loss of 2 % occurring in the service of fauna community. These losses were primarily a result of seagrass losses in 2001, which were only partially regained in 2010. In the Boston metropolitan area, focal ecosystem services capacity in 2016 was 3 % less than that in 1996, with the greatest loss of 3 % again occurring in the service of fauna community. These losses were again primarily a result of seagrass losses in 2001, which were partially regained in 2006, but also of saltmarsh losses in 2010. In the South Shore, all combined services fell 14 % from 1996 to 2016. The services of edible or commercially important fauna and fauna community fell by 17 % and 16 %, respectively. These losses primarily occurred in the 2016 along with a 45 % loss in seagrass cover. Finally, the combined services in Cape Cod fell by 9 % from 1996 to 2016, with the greatest losses of 11 %, 12 %, 10 %, and 10 % occurring in climate change impacts mitigation, fauna community, water quality, and edible and commercially important fauna, respectively. These changes were primarily a result of conversions between seagrass and tidal flats throughout the study period.

7. Discussion

Ecosystem service capacity reflects, among other measures, not only the areal coverage of an ecosystem, but also its connectivity across the landscape as well as its relative capacity to provide a specific service compared to other ecosystems. When combined with the social complexities of balancing stakeholder interests, these interacting influences can make managing and evaluating ecosystem services intractable, especially across large landscapes that span multiple social boundaries. Ecosystem services capacity matrices and mapping have been proposed as a tool to aid in this management process, as they allow for the visualization and comparison of services production across space and time with changing land cover. In this study, the literature-derived matrix is broadly transferable to any location where NOAA CCAP land cover data

Table 4
Conversion rates for combinations of land cover conversions and all combined ecosystem services. The conversion rate is the % change in combined ecosystem services capacity from the two habitats over the % of the combined area converted. The mean conversion rates are calculated as the slope of a statistically significant model between the % of area converted and the % change in service provisioning from all conversions across the NEP from 1996 to 2016. The mean slope from the bootstrap iterations is provided first, followed by the model significance (* p<0.05;** p<0.01;*** p<0.001). Positive values greater than one indicate a higher percent increase in service capacity than the corresponding % change in habitat cover. Values between zero and one indicate an increase in service capacity, but less than the corresponding % of land converted. Values less than 0 indicate a loss of ecosystem services. A more detailed table with specific services and bootstrapped ranges is provided in the supplementary materials S1.

Conversion Habitat	Original Habitat				
	High Elevation Salt Marsh	**Low Elevation Salt Marsh**	**Unclassified Salt Marsh**	**Seagrass**	**Tidal Flats**
High Elevation Salt Marsh	–	0.01**	0.06**	–	−0.78**
Low Elevation Salt Marsh	−0.01**	–	–	−1.03**	−13.14**
Unclassified Salt Marsh	−0.06**	–	–	0.87**	2.08**
Tidal Flats	0.78**	13.14**	−2.08**	0.63**	–
Seagrass	–	1.03 **	−0.87**	–	−0.63**
Open Water	−0.5**	−0.48**	−3.69*	−0.42**	−0.41***
Beach	−3.37**	−2.48**	−0.99**	–	−2.56**
Estuarine Scrub/Shrub Wetland	−0.59***	0.69**	0.91*	–	–
Development	−4.71**	−3.06**	−6.75**	–	−13.53**

is available, and covers a breadth of ecosystem services by being directly linked to the NESCS Plus Classification system, thus allowing interoperability among a suite of tools designed to identify, prioritize, and quantify ecosystem services (Newcomer-Johnson et al. 2020). For MassBays, we have further demonstrated how a literature-based matrix can be modified with expert knowledge, to account for local differences in quality or community use of ecosystem services. Furthermore, the capacity matrix, though based on mean values, includes an assessment of variability represented as the standard deviation or range of values obtained from the literature and from the local experts. This variability could be used to characterize the uncertainty in ecosystem services capacity and is important to managers seeking to make informed decisions within the context of the often subjective or ambiguous value systems of ecosystem services.

7.1. General trends

This assessment of ecosystem services capacity for MassBays revealed that while seagrass loss across the landscape has contributed to an overall loss in all combined ecosystem services from 1996 to 2016, each region tells a different story that reflects its unique service distribution and history of land cover change. Seagrass cover in MassBays fell by 50% between 1996 and 2016, resulting also in a 50% loss in ecosystem services provided by this habitat. This loss was consistent across all regions except the Upper North Shore, which gained 9% (0.04 km^2) more seagrass in 2016 than in 1996. Most of the lost seagrass (83%) in the other regions was replaced by open water, an ecosystem with relatively low capacity for the focal services in this analysis. However, some (17%) was replaced by tidal flats, which as a relatively productive ecosystem, provided some buffer against the lost seagrass services. Still, the combined 23 km^2 of lost seagrass in Cape Cod and the South Shore contributed to the greatest losses in ecosystem services across the five regions. Overall, seagrass conversions were responsible for 67% of the change in services from 1996 to 2016, with another 30% occurring within tidal flats. Thus, these two habitats accounted for 97% of the service capacity change, despite representing only about 20% each of the total focal habitat area. This indicates that they are the most influential in driving changes in service capacity and might therefore be the focus of planned management strategies aimed at restoring and/or protecting ecosystem services.

This clear trend in the dominance of seagrass and tidal flats in service capacity changes across the NEP was apparent within individual regions as well, despite differences among them in inherent physical and biological characteristics, as well as the social values reflected in the matrices. The Upper North Shore, for example, had virtually no services from tidal flats or seagrass in 1996, primarily a result of the relative absence of these habitats at the time. However, although there were minor changes in the dominant habitat of saltmarsh from 1996 to 2016, their influence in service production paled in light of the nearly 4 km^2 of tidal flats gained in the same time period. This gain in tidal flats, which came primarily from open water, increased the focal ecosystem's services production by 4%, with relatively minor changes resulting from other land cover conversions. Also, the Lower North Shore, Boston, and the South Shore experienced more saltmarsh losses to development than any other regions. Still, aside from small losses in Boston, these conversions were relatively inconsequential to service capacity when compared to the seagrass losses to open water and the tidal flat gains from a mix of other habitats.

7.2. Influence of connectivity

As habitats converted across the landscape, the subsequent changes in service provisioning depended on the differences in the relative capacity of the two habitats to provide a service, as well as the relative change in connectivity in the two habitats. While most of the observed changes in services provisioning stemmed from differences in the capacity scores, there was some influence from connectivity that at times determined a loss or gain in net services. In Namskaket Creek of Cape Cod for example, 0.4 km^2 of seagrass was converted to tidal flats from 1996 to 2016. Because both of these habitats are scored the same for edible or commercially important fauna, this conversion would theoretically not change the net capacity of the landscape if connectivity was not considered. In this case, however, the conversion diminished the connectivity of both seagrass and tidal flats, resulting in a 27% loss of this service when only 10% of the combined habitat area was converted. Although it may seem counterintuitive that additional tidal flats would diminish overall connectivity, in this case the additional patch was more than 0.5 km from the other patches, resulting in an overall decrease in the collective connectivity.

Overall, changes in connectivity contributed a median of 40% to the observed changes in service production, with the median value for changes in areal coverage making up the remaining 60 %. As with much of this analysis, these ratios varied considerably with region. In Boston, the change in service production was due almost entirely to changes in habitat cover (98%), with connectivity contributing only 1%. In the South Shore and Upper North Shore, the median contribution from connectivity was 22 % and 27%, respectively. On the other hand, in the Lower North Shore and Cape Cod, the contribution from connectivity was 56% and 59%, respectively. By habitat, these ratios were also variable, with seagrass and tidal flats being more sensitive to connectivity (65% and 69%, respectively) than high and low elevation saltmarsh (40% and 20%, respectively).

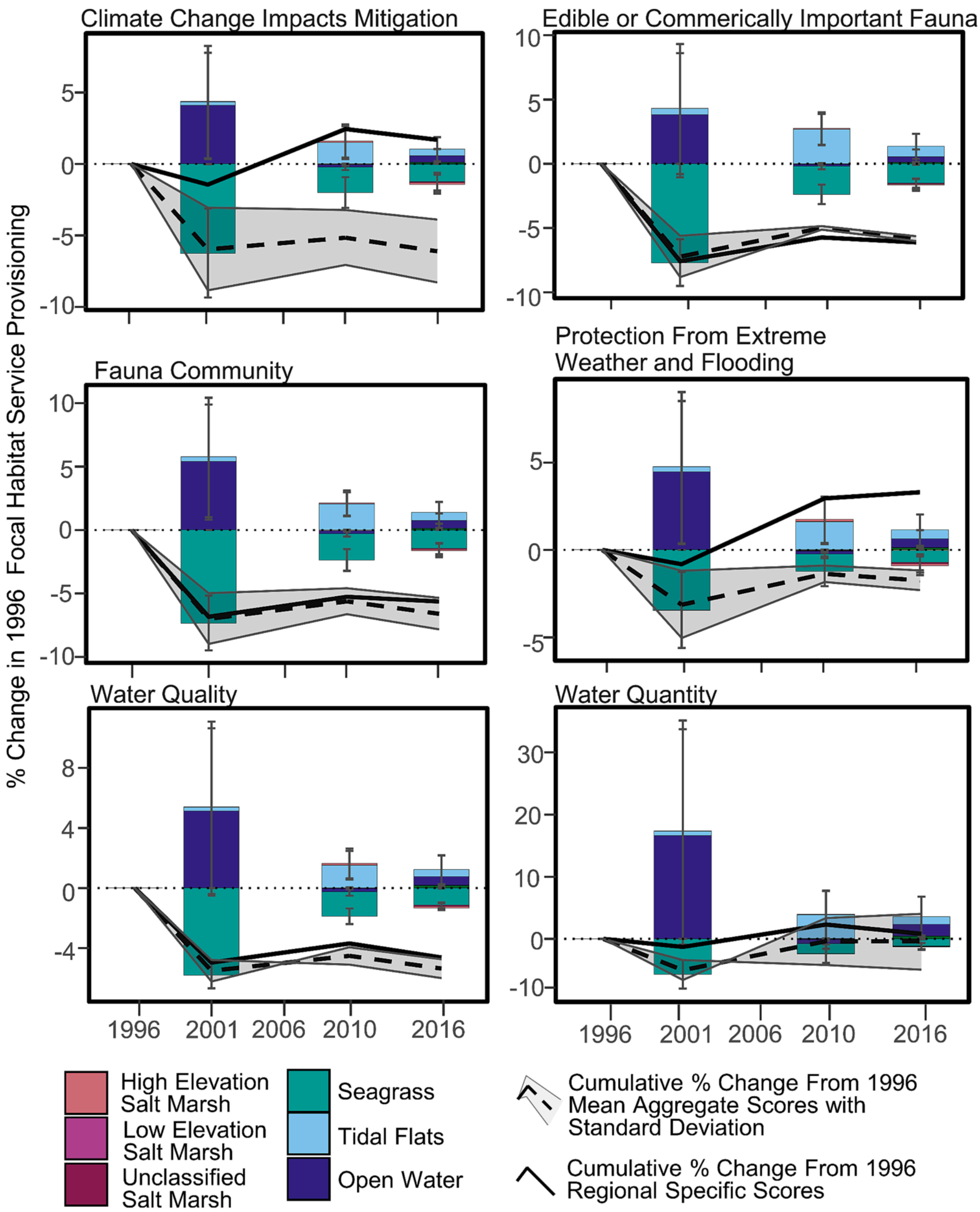

Fig. 5. The percent change in focal ecosystem service capacity NEP wide. Positive bars signify the percentage of focal service gains attributed to the corresponding habitat for that time-step. Negative bars signify the percent of the loss in services attributed to the corresponding loss of habitat. Lines represent the cumulative change in 1996 focal ecosystem services using both the NEP wide mean scores and standard deviations, as well as the mean of the regional specific scores. Error bars and shaded areas show the standard deviation from the 100 boot strap iterations. A more detailed version with regional specific analysis is available in the supplementary materials S2.

7.3. Practical applications

One major benefit of using spatial data to examine service capacity, aside from a more complete accounting of services across the landscape, is that it can more easily identify specific locations for management action (Burkhard et al., 2012). To demonstrate this, we mapped the change in service production from 1996 to 2016 in each of the MassBays Embayments, with each pixel reflecting the percent change in service provisioning within the pixel's coverage area (Supplementary S2). These maps show specifically where service provisioning has most changed, and as a result, where proposed management plans (e.g. preservation, restoration, rehabilitation, etc.) might be most effective in returning to historical service levels. As an added benefit, they also demonstrate counterintuitive scenarios of provisioning change that may be useful in avoiding costly management errors based on over simplistic assumptions. For example, in the embayment of First Encounter Beach in Cape Cod, there was a 0.2 km^2 patch of seagrass that was replaced by tidal flats between 1996 and 2016. In other regions, this conversion would have lowered overall ecosystem services production of the landscape, but in Cape Cod, seagrass is not ranked as high as tidal flats for most services and as a result, this patch of landscape's capacity was reduced. All individual embayment maps are presented in the supplementary materials S2.

Discussions about historic losses of coastal habitats and their benefits with MassBays NEP partners are helping to support restoration target setting to restore seagrass, prevent further losses of saltmarsh and tidal flats, and maintain and restore valuable ecosystem services by improving the quality of coastal habitats (MassBays National Estuary Program, 2015). MassBays is also working to identify metrics for monitoring restoration progress beyond just acres of habitat. To achieve targets, local-scale restoration projects are currently in planning or implementation phases across the partnership area. Scaling ecosystem services assessments to local-scales (Fig. 7) can help support an understanding of local loss or gain in habitats, prioritize and compare alternative restoration projects, and to communicate and track the potential benefits of restoration over space and time.

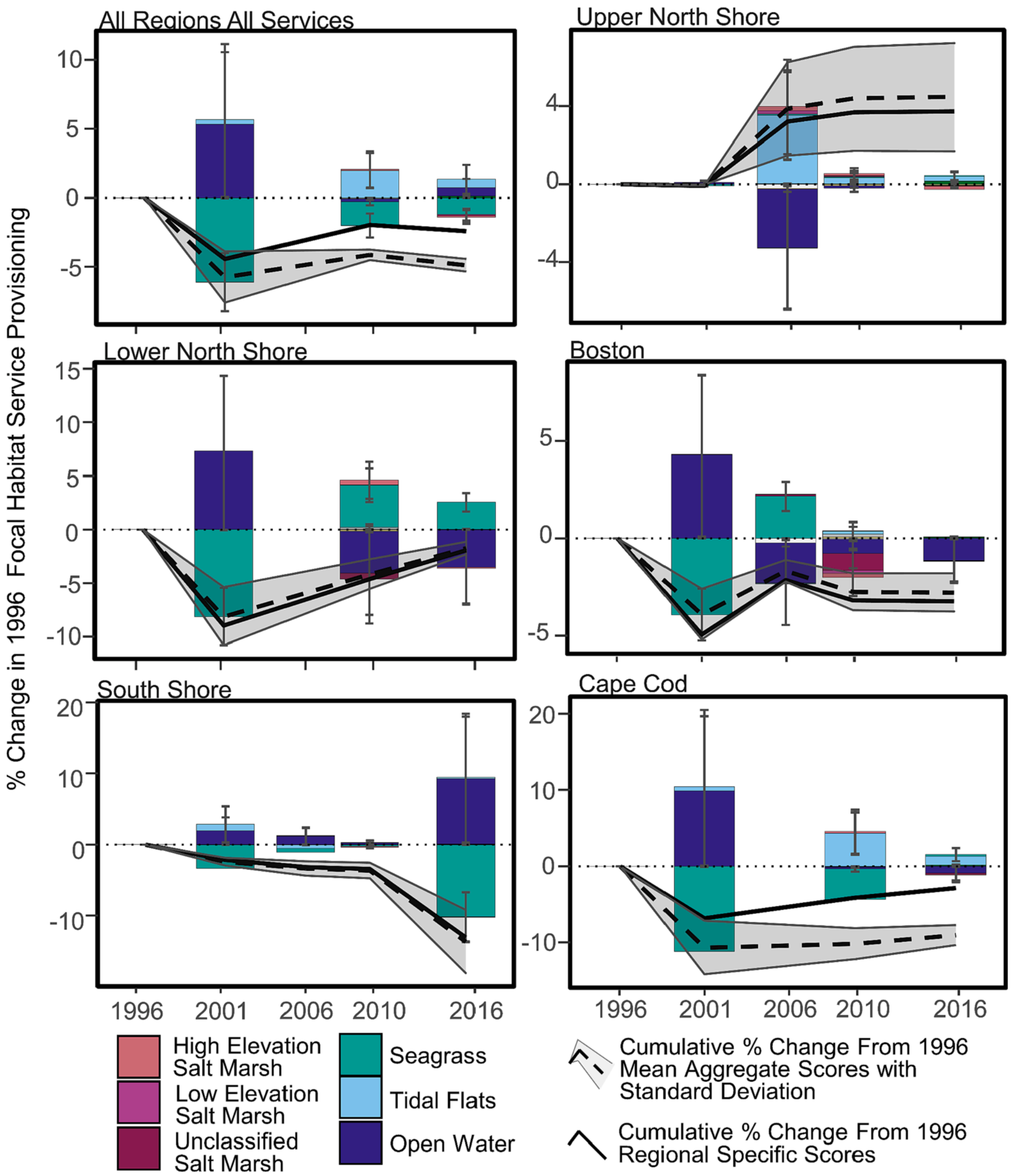

Fig. 6. The percent change in combined focal ecosystem service capacity in the five regions of MassBays. Positive bars signify the percentage of focal service gains attributed to the corresponding habitat for that time-step. Negative bars signify the percent of the loss in services attributed to the corresponding loss of habitat. Lines represent the cumulative change in 1996 focal ecosystem services using both the NEP wide mean scores and standard deviations, as well as the regional specific scores. Error bars and shaded areas show the standard deviation from the 100 boot strap iterations. A more detailed version with service specific analysis is available in the supplementary materials S2.

7.4. Caveats and perks to our approach

This spatial approach does involve potential drawbacks, however. Remotely sensed LULC data, for example, is not always accurate and classification errors will permeate throughout a spatial analysis such as this one. The National Land Cover Dataset, for example, from which the NOAA CCAP data is derived, has been found to be particularly inaccurate for wetland classes (Hollister et al., 2004). Further, seagrass has also been singled out as particularly challenging for remotely sensed data (Rowan and Kalacska, 2021). For these reasons, supplementary data and in particular on-site surveys should be used to fortify remotely sensed data, when available. Although the present analysis is based largely on the satellite derived CCAP classifications, both seagrass and saltmarsh classes were at least partially verified with alternative data, both from on-site surveys. Finally, although we have attempted to account for habitat patch connectivity in our analysis, very little empirical evidence is available by which to base such calculations. We have simplified our approach to assume that the influence of connectivity is linear, that it is independent of the specific service, and that it is a simple added percentage of the baseline contribution from area alone. Although we believe these assumptions to be well justified given the existing research, they have yet to be verified. Testing these assumptions would go a long way in improving future spatial analyses and models of ecosystem service provisioning.

Another limitation of our approach lies in the numerical scores given to each habitat-service combination, which imply a level of exactness that may not always reflect reality, especially in the case of the literature derived scores. In many cases, matrices from the literature were originally scored from 1 to 3, representing qualitative service capacity scores of “low” to “excellent”, respectively. For consistency, our analysis standardizes all matrix values to a range of 0 to 10. Thus, the original integer values of 1, 2, & 3 become 0.0, 5.0, and 10.0. Other matrix values may standardize from 3 and 13 for example, to 0.73 and 3.33, respectively. These new values may imply a level of capacity scoring accuracy that is misleading and/or was intentionally averted in the original publication. Although we have intended to reduce the effect of this on our analysis by primarily using local expert scoring and by providing all of the original scores in the supplementary materials, it is important to consider this limitation in the scoring interpretation. This is especially true if the aggregated matrix is to be used for other analyses.

Although our presentation of service capacity in terms of relative change from a given baseline shies away from quantifying actual service

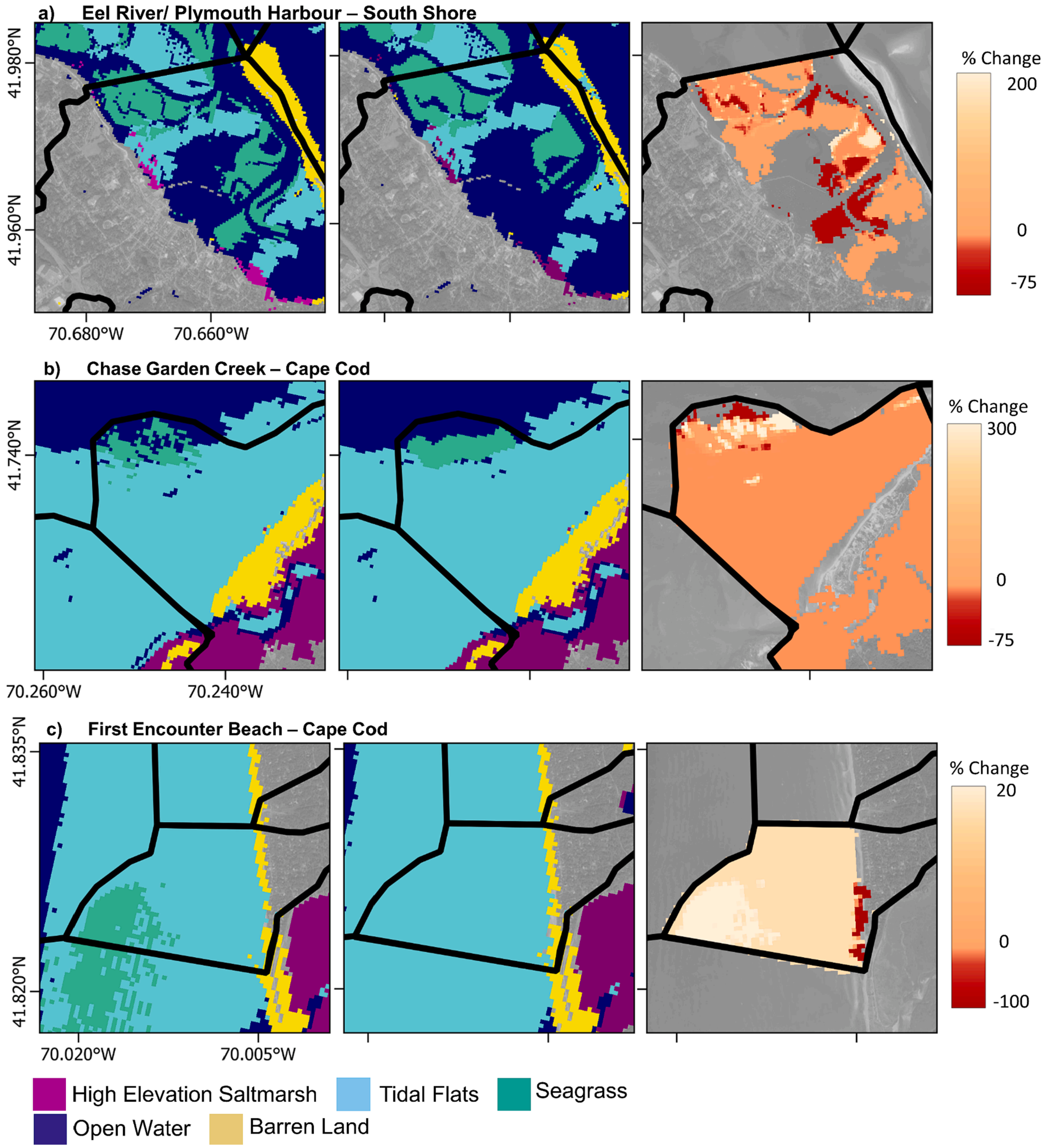

Fig. 7. Example maps for individual embayments showing the 1996 (left) and 2016 (center) habitat cover as well as the percent change in overall services provisioning for each pixel (right). Such maps simplify management planning on the basis of ecosystem services. The top panel (a) shows large areas of service loss due to conversion of seagrass to open water. The middle panel (b) shows large areas of gain due to seagrass expansion. The bottom panel (c) shows how assumptions (e.g. seagrass restoration will improve service provisioning) may not be in line with local expert concepts regarding habitat connectivity and overall service provisioning (note how a loss of seagrass in Cape Cod translated to a gain in services, due to a higher score for the replacement habitat of tidal flats).

value changes, it also avoids the uncertainties and pitfalls associated with some valuation techniques (Farber et al., 2002; Pascual et al., 2010). By simplifying the value to a percent change from a baseline, results can be easily interpreted by stakeholders who may not be familiar with monetary (or other valuation) metrics. Further, if an actual service value can be confidently determined, the relative change approach can still be used to calculate and extrapolate new values based on the known baseline. For example, Costanza et al. (2014) estimated that seagrass value for food production in 2011 was \$2384 ha^{-1} yr^{-1}. For MassBays under the present analysis, this would have meant a total food production worth of \$6.5 MM in 2010, and a loss of \$722,000 from 2010 to 2016. It would also mean a total gain of \$5.8 MM yr^{-1} if the

1996 seagrass cover were to be restored. Thus, if an accurate assessment of the service value for these habitats can be obtained, the relative changes presented here can provide valuable information to managers and policy makers seeking to reach stakeholders on the benefits of planned management actions. Further, socio-economic influences may also be incorporated according to local expert knowledge based on the fine-scale spatial analysis, such as where planned preservation/rehabilitation/restoration efforts may be most beneficial according to the local managers. In this way, analyses such as the one presented here can be most strongly leveraged as tools that combine with local knowledge to facilitate complex managerial decision making.

8. Conclusions

As the influence of ecosystem services in policy and ecosystem management continues to rise, so too does the importance of accurate assessments of service value, capacity potential, and provisioning. While our understanding of these metrics for individual services and ecosystems has improved greatly in recent years, the challenge of synthesizing ecosystem service dynamics across expansive landscapes and through time remains considerable. Matrices that provide relative values for arrays of service-habitat combinations are powerful tools due to their simplicity and their compatibility with land cover datasets that have become more ubiquitous and readily available. When paired with existing methods in spatial data processing, such as connectivity analyses, researcher have the potential to provide more comprehensive landscape scale assessments of ecosystem service dynamics to local managers and policy makers. The MassBays NEP is seeking such assessments for their management plans and have collaborated with us on in order to provide local expert knowledge. Here, we demonstrate an approach that combines this local collaboration with existing methodologies in ecosystem services assessments and spatial analyses to provide an evaluation of coastal ecosystem services across multiple scales. By incorporating variability, we also retain the uncertainty that is inherent but also critical to the decision-making process. In this analysis, seagrass appears to be the dominant player in MassBays ecosystem services across the partnership area, but local level assessments often deviate from this general trend. As our understanding of the spatial influences of ecosystem services continues to expand, this approach can be further optimized to provide still more realistic estimates, which will in turn result in more informed decisions.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data are included in supplementary files and an R script is available in an online repository linked to in the mansucript.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoser.2023.101530.

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