# Ecosystem Services Conceptual Model Application

NOAA and NERRS Salt Marsh Habitat Restoration

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SUMMARY

Interest in using ecosystem services to integrate considerations of people and the environment continues to grow in federal agencies. One method that can help agencies incorporate ecosystem services into decision making is the use of ecosystem services conceptual models, which link changes in biophysical systems caused by an intervention to socio-economic and human well-being outcomes. Evidence-based ecosystem services conceptual models can provide efficiency and consistency in application, transitioning ecosystem services from an interesting concept to an actionable approach for natural resource management.

Despite the potential usefulness of these models, there are few examples available to build from and little published detail on how to implement them. This report provides an illustrative ecosystem services conceptual model for salt marsh restoration at National Estuarine Research Reserve System sites. The National Estuarine Research Reserve System, which is closely associated with the National Oceanographic and Atmospheric Administration, wants to protect and restore coastal ecosystems while reinforcing local social and cultural systems. Developed by Nicholas Institute for Environmental Policy Solutions staff with staff at the National Oceanographic and Atmospheric Administration and National Estuarine Research Reserve System, the ecosystem services conceptual model captures the potential outcomes of a salt marsh habitat restoration. An accompanying evidence library provides a summary of the currently available evidence for each relationship in the model and an assessment of the strength of that evidence.

INTRODUCTION

Estuarine systems are areas of immense ecological importance and provide social, economic, and environmental benefits. The National Estuarine Research Reserve System (NERRS) is responsible for estuarine stewardship, research, training, and education across a network of 29 sites throughout the United States (NOAA OCM n.d.). NERRS represents a partnership between the National Oceanographic and Atmospheric Administration (NOAA) and U.S. coastal states.

NERRS has acknowledged that using an ecosystem services lens for research, management, and decision making would be helpful to reflect the socio-ecological linkages of estuarine systems. Ecosystem services have therefore been incorporated into the new 2017–2022 NERRS Strategic Plan. This plan promotes strategies for NERRS sites to "provide information about ecosystem services and apply knowledge […] to support protection and restoration of coastal habitats," to "leverage and apply NOAA partnerships, funding, and expertise to integrate biophysical and socioeconomic data, thereby providing the foundation for interdisciplinary and ecosystem services research," and to "develop a better understanding of how natural and nature-based features can increase economic value to coastal communities through enhanced ecosystem services" (NOAA OCS 2017). Although ecosystem services are at the forefront of many NERRS activities, there exists no standardized approach for integrating these services into new and existing projects and programs.

Ecosystem services conceptual models (ESCMs) represent a possible entry point for incorporating ecosystem services into NERRS management. These models illustrate the way that a management intervention cascades through an ecological system and results in ecosystem service and other human welfare impacts. ESCMs can provide a foundation for understanding and communicating ecosystem services to audiences not familiar with the ecosystem services concept. These models can also help managers more easily identify and anticipate ecosystem service-related outcomes of management interventions, provide transparency around expected project outcomes, help avoid unintended consequences and identify additional project co-benefits, and even form the basis for quantitative models (Mason and Olander 2018; Olander et al. 2018).

With NERRS representatives, Nicholas Institute for Environmental Policy Solutions staff developed an ESCM for salt marsh restoration and a corresponding evidence library to test the utility of ESCMs for standardizing the implementation of ecosystem services-related projects across the 29 NERR sites. The hypothesis was that general ESCMs developed for common habitats at NERR sites (e.g. mangrove, sea grass, oyster reef, coral reef) could be adapted and applied at local sites across the network. Users could access generalized reference models and adapt them to their specific needs by specifying them to their local context (Figure 1). A set of general models for the important NERRS habitats could help provide consistency and common metrics and could reduce duplication of effort in application. If this salt marsh ESCM and associated evidence library is valuable to NOAA, NERRS, or both, additional models could be created for other habitats or management interventions and be applied in future NERRS projects.

Figure 1. Illustration of how a general model might be adapted and applied at different NERR sites, with each site only using the parts of the model relevant to its individual context.

Map source: https://coast.noaa.gov/nerrs/.

GENERAL SALT MARSH MODEL

Some of the information presented in this section is adapted from Olander et al. (2018).

Conceptual Model

The conceptual model was initially drafted with input from NOAA staff to represent the hypothesized effects of salt marsh restoration or preservation on ecosystem service and social outcomes. Evidence for each of the hypothesized relationships represented in the model was examined. The conceptual model was continually revised on the basis of information gathered in the evidence collection process and feedback from expert reviews (Figure 2). The final components in each chain in the conceptual model are outcomes that are directly relevant to people, including ecosystem services (green ovals) and associated values (teal boxes).

Model Notes and Considerations

Salt Marsh Restoration

The starting node of the model is "salt marsh restoration." This node was left in a generalized form, though many specific restoration actions could be chosen to start a conceptual model such as this one (e.g., planting native species, removing invasive species, reconnecting waterways, increasing marsh elevation, removing dykes and levees). Restoration or preservation actions could affect various features of the marsh, each of which could be indicators for the following node, "change in salt marsh quantity or quality." Example marsh features that could be represented by a "change in salt marsh quantity or quality" include marsh habitat extent, vegetation density, vegetation height, and vegetation species composition.

Ecosystem Service and Social Outcome Endpoints

This model is meant to be not only general, but also inclusive. The ecosystem service and social outcome endpoints (green circles in Figure 2) represent a wide array of possible outcomes resulting from an intervention that affects salt marsh quantity or quality. These endpoints are meant to align with the concept of benefit-relevant indicators (BRIs), which reflect changes in ecological condition that are relevant to people (NESP 2016; Olander et al. 2018b). Although endpoints presented here are not BRIs, they represent categories of services or benefits that could be specified to BRIs specific to a particular site. See Mason and Olander (2018) for example indicators for each endpoint of the diagram.

Endpoint Connections

There are multiple ways that ecosystem service and social endpoints may influence each other, though these interactions are not displayed in the model. In certain cases, these connections/interactions may be important; determination of whether they should be included in a site-specific model must be decided on a case-by-case basis. Examples of these endpoint connections include the relationship between commercial fishing and culture; in some locations, commercial fishing is a large part of a population's cultural identity. Aesthetics and recreation can often be linked, because most people will want to recreate at an aesthetically pleasing site. Research and education can be linked to existence values, because environmental education can result in a person's appreciation for nature and wildlife.

Economic Effects

If economic effects of ecosystem and social services is important, the model can be extended to include these values (teal boxes in Figure 2). The values shown in Figure 2 represent options for monetary valuation that are accepted and have been used by NOAA; however, other options for monetary valuation of ecosystem services exist. The figure does not show monetary value options for services related to "culture and heritage" and "research and education." Conversations with NOAA economists revealed that NOAA does not usually promote the monetary valuation of services falling in these categories.

Spatial and Temporal Considerations

ESCMs are a conceptual schematic to help think through the logic of a change in a system, but they do not depict all important aspects of these changes. They can sometimes include a simplified indication of the temporal dimensions of change, such as short-term temporary changes versus long-term persistent changes, but often the temporal dimension is missing. Spatial considerations are also important—certain services, such as coastal protection or water filtration, will

vary on the basis of the spatial orientation of the marsh and its surrounding environment. ESCMs do not show the very important spatial dimensions of system dynamics, and they are not designed to capture system feedbacks, but they can provide a starting place for considering such feedbacks (see below). However, both spatial and temporal considerations are addressed in the evidence library and are often included in the "other factors" section.

Feedback Loops

Many biophysical and social feedback loops are present in a salt marsh system, though they are not represented visually in this model diagram. Instead, feedback loops are often addressed in the text of the evidence library to simplify the model image and keep the focus on the predominant flows or cascades between the ecological and social aspects of a salt marsh system. Example feedback loops include the relationships among algae blooms, turbidity, and primary production; the negative feedbacks of some recreational activities on wildlife populations; and the impacts of overfishing or overharvesting on fish, shellfish, or crustacean populations. Feedback loops could be incorporated into an ESCM if the user requires their inclusion.

External Drivers

This model includes only aspects of the socio-ecological system that are affected directly or indirectly by an intervention meant to protect or restore a salt marsh. Also influencing estuarine systems are many external drivers, which are not represented graphically in the model. Many of these drivers, such as sea-level rise, climate change, land use change, invasive species, and storms, are addressed where applicable in the "other factors" sections of the evidence library. If a conceptual model is specified to a site and turned into a quantitative model, external drivers will have to be incorporated to accurately model the system. See Mason and Olander (2018) for a depiction of a quantitative model incorporating external drivers.

Figure 2. Generalized salt marsh ecosystem service conceptual model
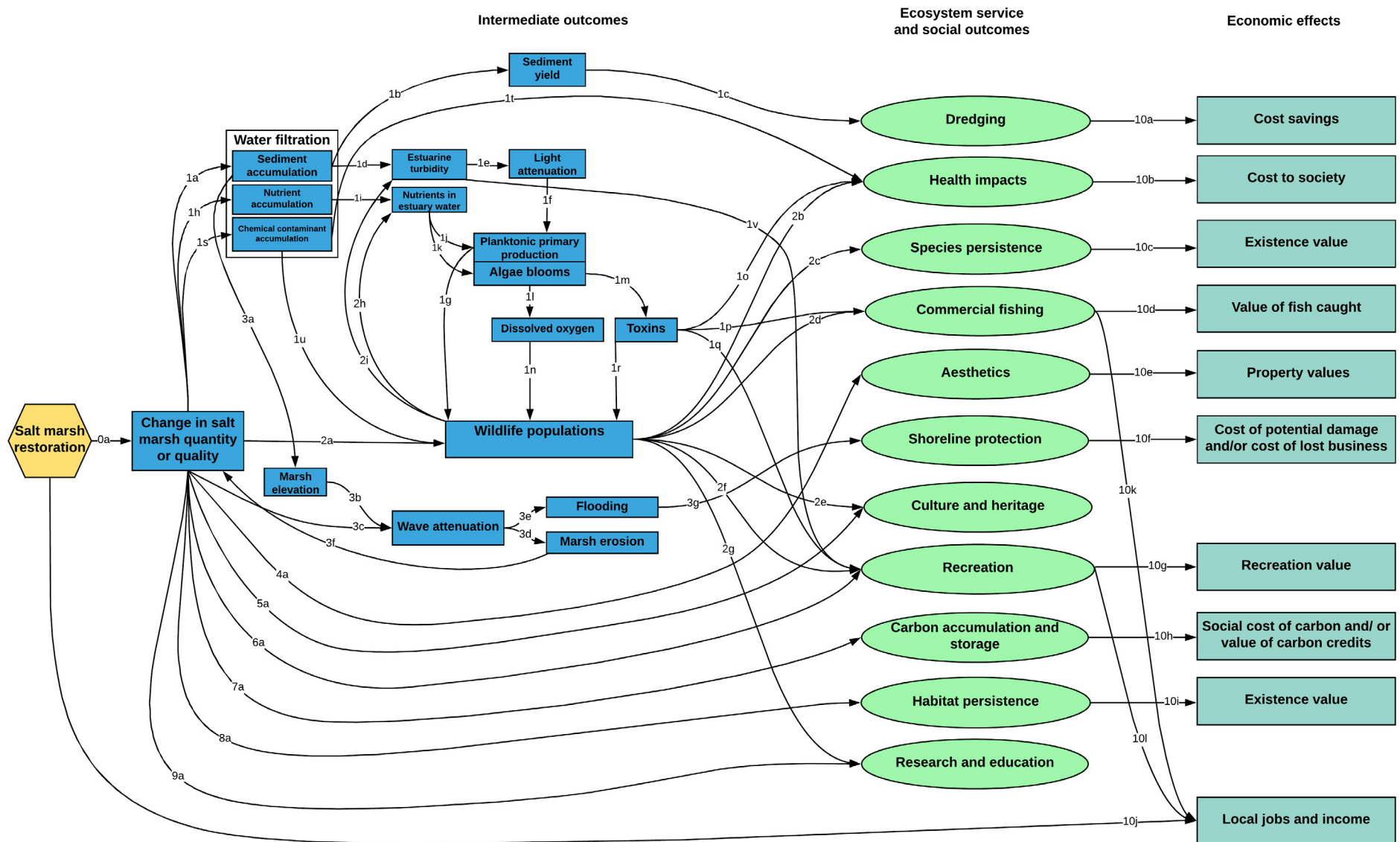
Note: The economic effects listed in the far-right column do not all measure the same thing and cannot be added together or directly compared to each other.

Evidence Collection

Evidence was collected to support each link (arrow) in the model. This evidence was found through a search of online academic databases and Google Scholar using key words from each link. To assess the current level of understanding, generalizability of evidence, and consistency of effects for each relationship, the search emphasized meta-analyses, research syntheses, and review articles. Other types of evidence, including individual research studies, technical reports, computer models, and interviews, were also considered. The literature search for each link was not exhaustive, but it was reasonably extensive and should be sufficient for a general sense of the available evidence. Ideally, further work would be done to refine and update the evidence over time.

Evidence Library

Evidence collected for each link was entered into an evidence library. The evidence library is organized by link number (see Figure 2), with each link entry representing the relationship between two nodes in the conceptual model. All link entries contain the information described in Table 1.

Table 1. Description of the contents of the evidence library

Evidence library contents for each link	Description
Description of the relationship	Description of the relationship between the starting and ending node; when possible, a specific statement of change is included, along with an indication of the direction and magnitude of change
Summary of evidence	Discussion of the relationship between the two nodes works and the supporting evidence for the relationship
Other factors	A list of other factors that may influence the relationship between the two nodes
Strength of evidence	An evidence grade for the strength of the relationship, determined using an evidence matrix (see Table 2)

Although most evidence library entries contain information pertaining to a single link, some entries in the library contain evidence that combines multiple links—but only when an applicable model or tool encompasses more than one link. For example, there is an InVEST fisheries model that uses salt marsh habitat inputs to predict fish landing outputs. This InVEST model (as a form of evidence) combines information from links 2a and 2d, and it is listed with the link number 2a,d to represent this combination. These combination links do not have an evidence grade because they represent one piece of evidence (a single model) rather than a body of evidence.

Kinds of Information

Two kinds of information are included in evidence libraries: evidence and examples. Evidence describes general or site-specific relationships between nodes and can include individual research studies, models, calculators, and meta-analysis results. Individual research studies can provide evidence for the existence of a relationship, but they are usually low-quality evidence for contexts other than those in which the studies were conducted (see below).

For links with missing or weak evidence, examples of site-specific studies that could be done at a site or for a particular intervention to fill an evidence gap for this library are provided. In many cases, the example studies are individual research studies conducted in other contexts that are considered part of the body of evidence for the relationship but that also illustrate how the relationship could be assessed in the focal context. In a few cases, the example studies are more general methods papers that describe an approach but that do not contribute to the evidence for the relationship.

Strength of Evidence Assessment

The strength of evidence available for each link was assessed on the basis of the following criteria (Table 2). Of the two kinds of information described above, this method was used to assess only evidence. Evidence must score "high" for each of the four criteria to receive a "high" strength of evidence rating. If the evidence for a link does not receive the same rating for all criteria, the overall strength of evidence for the link is determined by the evaluator, taking into account each of the individual ratings.

Table 2. Strength of evidence assessment rubric

Confidence level	Criteria			
	Types of evidence	Consistency of results	Methods	Applicability
High	Multiple	Direction and magnitude of effects are consistent across sources, types of evidence, and contexts	Well documented and accepted	High
Moderate	Several	Some consistency	Some documentation, not fully accepted	Some
Fair	A few	Limited consistency	Limited documentation, emerging methods	Limited
Low	Limited, extrapolations	Inconsistent	Poor documentation or untested	Limited to none
None	None	N/A	N/A	N/A

Note: N/A = not applicable. This evidence assessment rubric was adapted from a product created by the Bridge Collaborative: http:// bridgecollaborative global.org/wp-content/uploads/2018/02/Practioners_Guide_Final_2.pdf.

Types of evidence can include individual research studies (experimental or observational), meta-analysis or synthesis studies, tools, models, expert opinion, and local knowledge. Consistency of results takes into account the direction and magnitude of effects shown in the evidence. Methods includes the level of documentation provided, whether methods are supported by other literature and appropriate for the study objective, and whether limitations of the methods are discussed. Applicability refers to the relevance of the evidence to the relationship, including the geographic, social, and biophysical contexts of the evidence relative to the relationship in question.

Evidence Considerations
Some of the information presented in this section is adapted from Olander et al. (2018).

In assessment of the available evidence for a particular link, two distinct aspects of the evidence require consideration: *existence* and *predictability.* The first consideration is *existence* of a relationship between the two nodes involved—does a change in one node lead to some change in the other? The second aspect, which is dependent on the first, is *predictability* of that change. Do we have evidence to show how one node will change with the other? Is this information generalizable to all scenarios, or is it context specific? During collection of evidence for a general model, consideration of the generalizability of predictive capability becomes especially important. Our evidence libraries focus on the evidence for *existence* of a relationship, and where possible, we highlight the *predictability* of the relationship.

Strength of evidence also needs to take "other factors" into consideration. Take the hypothetical relationship between nodes A and B. There may be a large body of evidence describing the existence of the relationship between A and B, but there may also be other factors that also influence B. Those other factors might not appear in the conceptual model diagram (because they are not affected by the intervention), but they may be important in the estimation of an outcome in node B. The existence of these other factors will likely lower the evidence grade because they reduce the applicability and consistency of the evidence that links A and B. Alternatively, those other factors can be added to the ESCM, and the strength of evidence for their influence on intervention effects can be directly considered.

Evidence collection for generalized conceptual models has limitations. Certain nodes in a general model are purposefully left vague or general, and they will need to be specified once a local site is chosen. For example, the "wildlife populations" node is general, and specific wildlife species will have to be selected when applying the general model at a local site. These generalizations limit ESCM users' ability to gather applicable evidence in some cases. Although it may be possible to gather evidence for linkages between various nodes and the general "wildlife population" node, it is impossible to gather relevant evidence for all species. Due to the general nature of that node and ESCM users' inability to make definitive statements about the connection between other nodes and general "wildlife populations," the evidence grade for those links in the general model will often suffer. In many cases, these nodes will list example studies (rather than evidence) to illustrate how the links might be assessed once a more specific node is selected for a local site.

Strength of Evidence Map

Once evidence has been evaluated using some confidence rubric, the confidence in each link can be expressed visually in the ESCM. A number of researchers use what they call evidence gap map tables to provide a visual summary of the number of studies done to test a broad suite of interventions and a broad suite of targeted outcomes. Multiple examples have been developed by The International Initiative for Impact Evaluation (3ie) (Snilstveit et al. 2017) and others. However, when details on intermediate as well as final outcomes are to be included in an evidence review focused on a single or a few interventions, it is suggested that the expression of confidence within the ESCM be displayed using the conceptual model framework (e.g., Figure 3). The conceptual model can be used as the template, and arrows can be colored to represent the grade received by the evidence. We call these strength of evidence maps. See Figure 3 for the generalized salt marsh strength of evidence map.

A strength of evidence map for salt marsh restoration allows for a quick visual assessment of how well connections between restoration and outcomes that matter to people (ecosystem services, social outcomes, and economic effects) are supported by currently available evidence (Figure 3). These maps can also be used to inform research priorities because they identify research gaps and provide context for which gaps might be most important for addressing significant uncertainties or risks.

Figure 3. Strength of evidence map for the general salt marsh model
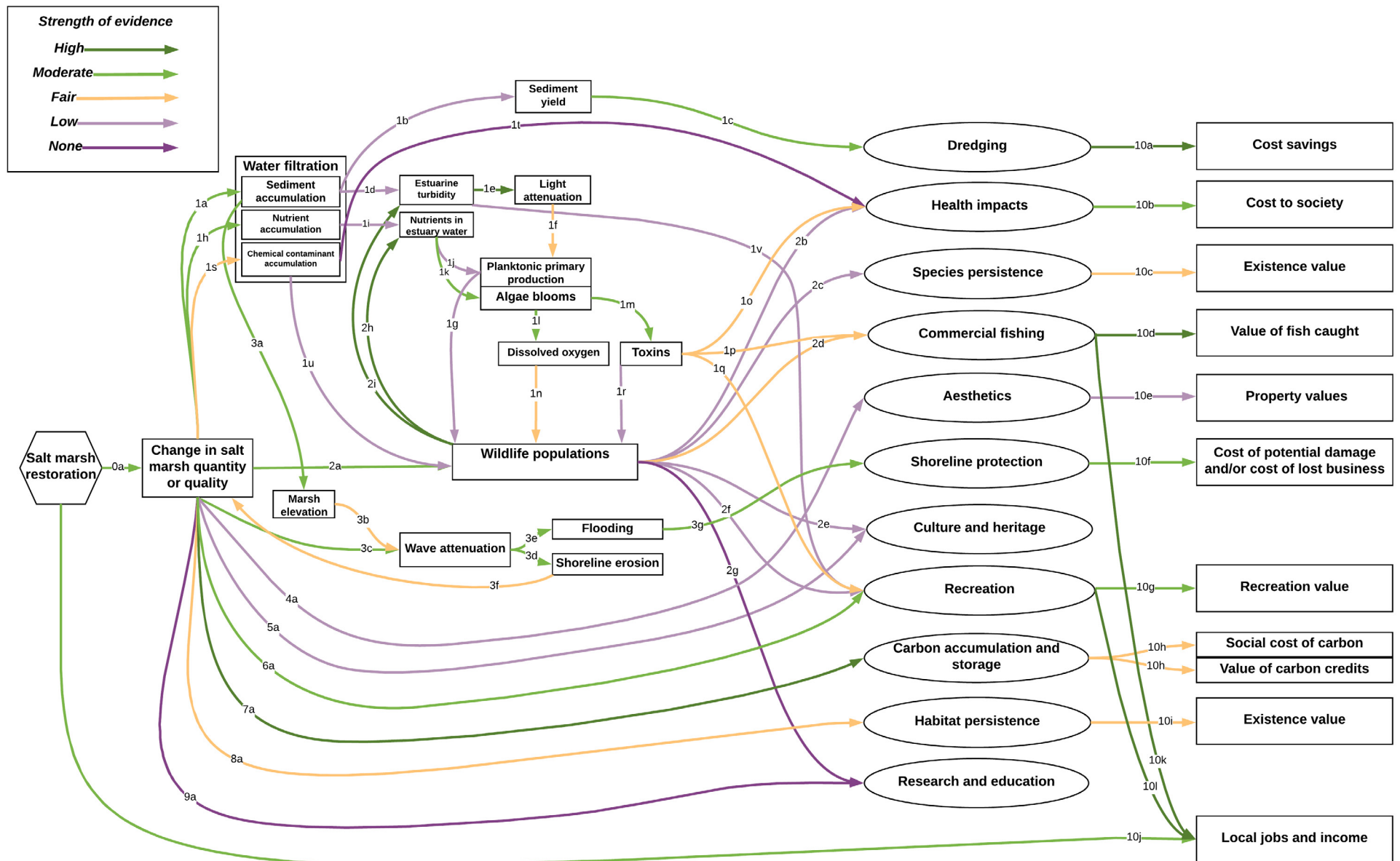
EVIDENCE LIBRARY FOR SALT MARSH MODEL

Figure 4. Evidence library table of contents
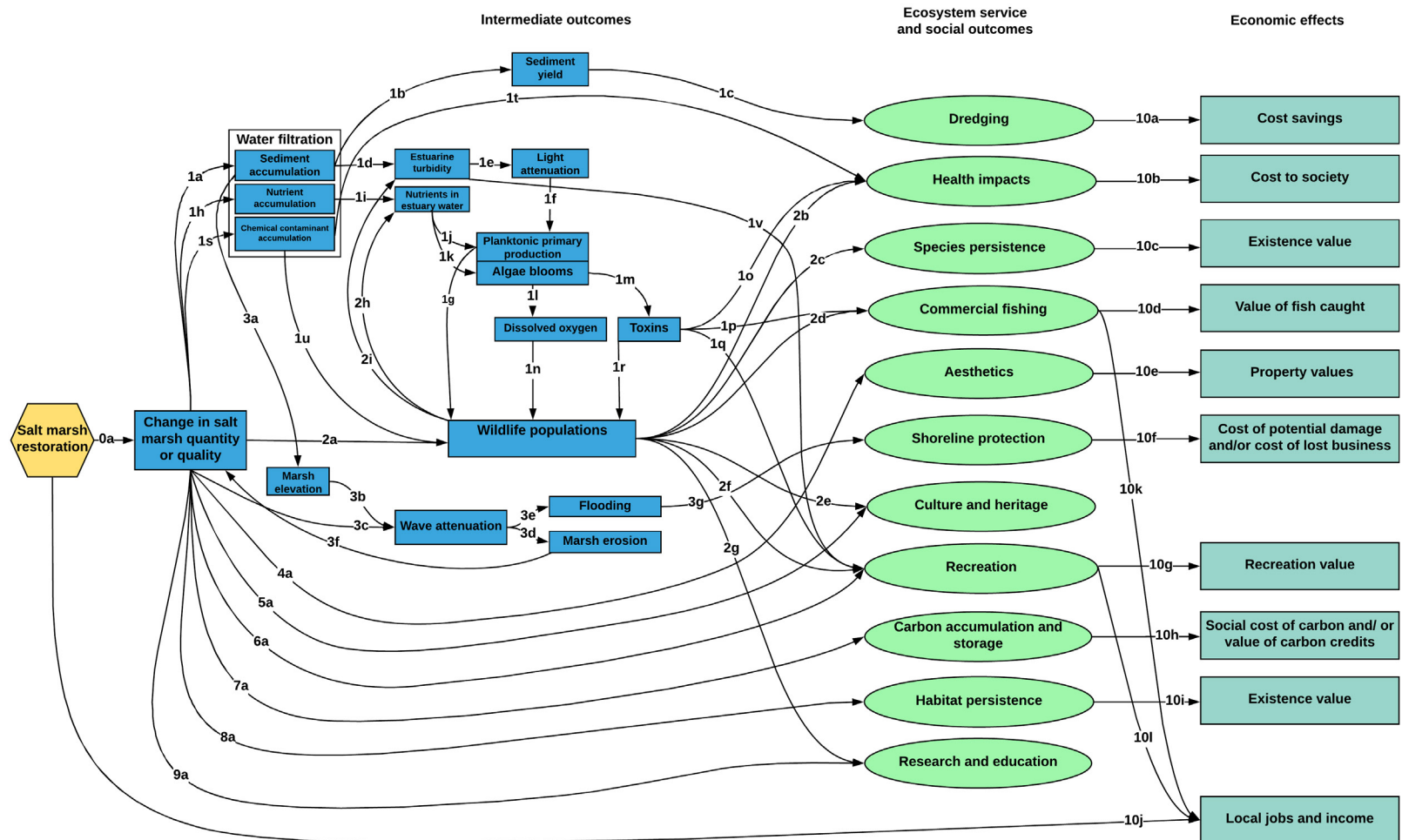
0a: Salt Marsh Restoration → Change in Salt Marsh Quantity or Quality

Description of Relationship

There are many actions and techniques used to restore salt marsh habitat. Site attributes and restoration actions will affect the resulting change in salt marsh habitat area or habitat quality.

Summary of Evidence

Salt marsh restoration occurs all over the world and across many different regions of the United States. It is important to acknowledge that a restored marsh site does not necessarily imply that the site will function equally to a similarly placed natural salt marsh. Research comparing restored salt marshes and natural salt marshes is ongoing, and many indicators have been used to specify the success of restored sites' ability to replicate natural marsh habitat. Metrics used to compare restored marshes and natural marshes include vegetated habitat area, vegetation diversity, diversity of wildlife using the site, specific wildlife species' use of the site, and various sediment characteristics.

Other Factors

Site Suitability: Suitability of a site will factor into how effective a restored marsh becomes. Research is ongoing, but studies have found that microtopography/elevation (Rezek et al. 2017), geomorphology (including sediment type and characteristics) (Calloway 2005; Capooci et al. 2016; Rezek et al. 2017), site hydrology (Calloway 2005; Capooci et al. 2016), site use and site history (Chang et al. 2015; Capooci et al. 2016) all affect the ability of a site to support and sustain a functional marsh. All these factors must be taken into consideration when establishing and enhancing restoration site suitability.

Time Lags: It may take time for a restored marsh to become fully established, and certain marsh functions may take longer than others to come into effect. Many studies examine restored marshes over time to document how features of a restored marsh change (e.g., Warren et al. 2002; Chang et al. 2015; Capooci et al. 2016).

Outside Risks: Once a marsh site has been restored, there are always risk factors beyond the control of the restoration team that can affect the establishment or sustainability of the marsh site. Some risks include large storms or other disturbances (e.g., an oil spill), sea-level rise, or the introduction of invasive species. Some of these risks can be addressed; sea-level rise can be addressed by increasing marsh elevation with added sediments, and invasive speces can be actively removed. Depending on the severity of the threat, addressing these risks through management may or may not be effective.

Strength of Evidence

Moderate. A large community of specialists (scientists and practitioners) focus on salt marsh restoration in the United States. Research to date suggests that restoration will establish a new salt marsh habitat if it is done properly and takes multiple site factors into account. Functionality of the habitat is less certain.

Predictability: This link is highly context dependent. The link between salt marsh restoration actions and the establishment of new salt marsh habitat or a marked increase in marsh quality will depend on the specific restoration actions taken and the suitability of the site for restoration.

Sources

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Capooci, Margaret, A.C. Spivak, and K. Gosselin. 2016. "Salt Marsh Ecosystem Responses to Restored Tidal Connectivity across a 14y Chronosequence." In *American Geophysical Union, Ocean Sciences Meeting 2016,* Abstract# EC14B-0972.

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Minello, Thomas J. 2017. "Fishery Habitat in Estuaries of the Gulf of Mexico: Reflections on Geographical Variability in Salt Marsh Value and Function." *Gulf and Caribbean Research* 28 (1): ii–xi

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Warren, R. Scott, Paul E. Fell, Ron Rozsa, A. Hunter Brawley, Amanda C. Orsted, Eric T. Olson, Varun Swamy, and William A. Niering. 2002. "Salt Marsh Restoration in Connecticut: 20 Years of Science and Management." *Restoration Ecology* 10 (3): 497–513.

Links 1a-v

1a: Change in Salt Marsh Quantity or Quality → Sediment Accumulation

Description of Relationship

Salt marsh habitats can trap sediments that flow across the marsh. There are various methods to measure and model sediment accumulation across the marsh surface.

Summary of Evidence

Salt marshes trap sediments through two primary pathways: (1) by sediments adhering to salt marsh vegetation (Pethick 1984; Li and Yang 2009), and (2) by sediments settling out of suspension from the water column because of salt marsh vegetation slowing the current (Pethick 1984; Shi et al. 2000). The source and accumulation of marsh sediments is unique to each marsh site. There are four sources of sediment inputs to an estuary environment: (1) marine sources derived from the sea bed, (2) coastal sources derived from cliff erosion, (3) fluvial sources brought in by rivers, and (4) in situ sources derived from within the estuary. The primary source of sediment at a given marsh will vary depending on the system (Pethick 1984), but in marshes in estuaries, river mouths, and deltas, sediments are often primarily derived from fluvial (river) inputs (Davidson-Arnott 2010). It is understood that salt marsh environments are generally areas of net sediment deposition (i.e., they are sediment sinks), rather than a source of sediment (Davidson-Arnott 2010).

There are various ways to measure sediment accretion in a marsh (the rate of sediment deposition), and many of these methods are reviewed in a paper by Thomas and Ridd (2004). These methods could be used to track marsh sediment accretion over time. No reviews or summaries of generalized marsh accretion rates were found. Examining salt marsh sediment accretion over the long term (longer than a decade) can be done using methods of stratigraphic dating, which measure the accumulation of ^{210}Pb, ^{137}Cs, or both, which provide marker horizons (Davidson-Arnott 2010).

A variety of tidal marsh evolution models (that incorporate marsh accretion outputs) have been developed, and they can account for various topographical and ecological aspects of the marsh environment when estimating sedimentation rates. A review of these models can be found in Fagherazzi et al. (2012), and most relevant here are the empirical and physical models that estimate sediment fluxes across a marsh platform (see sections 2.0, 2.1, 2.2, and 2.3 of Fagherazzi et al. 2012). If data on sediment accretion and environmental variables are available for a marsh site, statistical empirical models (multiple regression models) are relatively easy to develop (see Fagherazzi et al. 2012 section 2.1 and Temmerman et al. 2003).

When erosion rates are greater than sediment accumulation rates, net sediment accumulation on a marsh may be negative.

Other Factors

Vegetation Density: The density of marsh vegetation and properties of the marsh plant species may alter sedimentation rates; denser vegetation is associated with higher sedimentation (Li and Yang 2009).

Physical and Chemical Characteristics: Characteristics of the marsh and estuarine environment may play a role in sedimentation. These characteristics include sediment particle size, marsh elevation, suspended sediment concentration, and marsh vegetation proximity to sediment supply (Shi 2000; Li and Yang 2009).

Strength of Evidence

Moderate. It is widely accepted that marshes accumulate sediments, and there are numerous studies and models to support this link. However, there are no available reviews or studies that generalize the relationship between marshes and sediment accumulation, so consistency for the amount of sedimentation is limited.

Predictability: Sedimentation rate can be predicted, but additional site-specific information is required (see example section below).

Example. To measure or model the sediment accumulation rates for a particular marsh, a local study will have to be performed. There are many studies that document the accumulation of sediments in salt marshes and by using one of the methods outlined in Thomas and Ridd (2004) it would be possible to track sediment accumulation at an individual marsh

site. Using accretion data with other environmental datasets for a particular marsh site would enable creation of multiple regression models that could estimate sedimentation across the entire marsh. The utility of the model and its capability to truly predict sedimentation will depend on the environmental predictor variables used and the r-squared value of the regression model.

Sources

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Pethick, John S. 1984. "An Introduction to Coastal Geomorphology." Dept. of Geography, Univ. of Hull. https://www.osti.gov/scitech/biblio/6715017.

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Temmerman, Stijn, Gerard Govers, S. Wartel, and P. Meire. 2003. "Spatial and Temporal Factors Controlling Short-Term Sedimentation in a Salt and Freshwater Tidal Marsh, Scheldt Estuary, Belgium, SW Netherlands." *Earth Surface Processes and Landforms* 28 (7): 739–755.

Thomas, Séverine, and Peter V. Ridd. 2004. "Review of Methods to Measure Short Time Scale Sediment Accumulation." *Marine Geology* 207 (1): 95–114.

1b: Sediment Accumulation → Sediment Yield

Description of Relationship

As salt marshes capture sediments, the amount of sediment accumulating in nearby channels and estuaries will change.

Summary of Evidence

There are many different approaches to model sediment yield; however, no studies could be found that explicitly studied the link between salt marsh presence and sediment yield.

A review of sediment yield prediction and modeling from the *Encyclopedia of Hydrological Sciences* identifies four primary approaches for estimating sediment yield (White 2006):

- Empirical models based on broad basin and climate descriptors

- Soil erosion and sediment delivery approaches, whereby measured or estimated soil erosion rates are factored by a sediment delivery ration, which is often based on basin characteristics

- Physically based and distributed basin modeling approaches, whereby movement of water and soil is estimated in a distributed way throughout a basin

- Models relating sediment concentration or load to river flow, whereby measured sediment concentration data are related to river flow characteristics.

The review details benefits and drawbacks to all four modeling approaches, and it continually emphasizes that modeling sediment yield is highly uncertain and that many model predictions have been incorrect (White 2006). White (2006) notes that sediment yield modeling is difficult due to variability in sediment yields over time and space, which increases uncertainty. Making a single numerical prediction of sediment yield is inadequate, and uncertainty predictions should always be included when reporting sediment yield estimates (White 2006).

One of the more widely used tools for sediment yield modeling is the Soil and Water Assessment Tool (SWAT). This model fits into the third category of models described in the White (2006) review (physically based models). Though relatively data intensive, this model is especially helpful because it can predict how sediment yields may change given various land use change scenarios. SWAT has been used to model sediment yields in relation to land cover in multiple studies (Parajuli et al. 2008; Betrie et al. 2011; Tyagi et al. 2014); however, no studies were found that examined the effect of salt marsh on sediment yields.

Other Factors

Climate and Flow Variability: Sediment yield changes with precipitation and flow of water. Because these are dynamic ecological factors, incorporating them into model predictions is often difficult due to uncertainty of future conditions (White 2006).

Extreme Events: Unusually high sediment yields often result during extreme events such as extreme weather or large-scale human interventions. Not all models incorporate such stochastic events (White 2006).

Strength of Evidence

Low. There are multiple approaches to predict how sediment yields will change depending on land cover. However, no studies were identified that used these approaches to examine how salt marshes might alter sediment yields in surrounding water bodies, so it is not clear whether sediment yield will change due to salt marsh presence. Additionally, this assessment of evidence assumes that all else is held constant and that the only change in the sediment budget is due to the salt marsh. Because estuarine systems are dynamic and constantly under development, it is unclear whether sediment yield will actually be affected by sediment accumulation in a salt marsh, therefore applicability of this evidence is limited.

Predictability: Using models such as SWAT, it is possible to calculate sediment budgets on the basis of local data. Importantly, no studies using this tool in a salt marsh context were found, and therefore the applicability of the results is unclear.

Sources

Betrie, Getnet D., Yasir A. Mohamed, Ann van Griensven, and Raghavan Srinivasan. 2011. "Sediment Management Modelling in the Blue Nile Basin Using SWAT Model." *Hydrology and Earth System Sciences* 15 (3): 807.

Parajuli, P.B., K.R. Mankin, and P.L. Barnes. 2008. "Applicability of Targeting Vegetative Filter Strips to Abate Fecal Bacteria and Sediment Yield Using SWAT." *Agricultural Water Management* 95 (10): 1189–1200.

Tyagi, J.V., S.P. Rai, Nuzhat Qazi, and M.P. Singh. 2014. "Assessment of Discharge and Sediment Transport from Different Forest Cover Types in Lower Himalaya Using Soil and Water Assessment Tool (SWAT)." *International Journal of Water Resources and Environmental Engineering* 6 (1): 49–66.

White, Sue. 2006. "Sediment Yield Prediction and Modeling." In *Encyclopedia of Hydrological Sciences*. New York: John Wiley & Sons, Ltd. doi:10.1002/0470848944.hsa089.

1c: Sediment Yield → Dredging Frequency

Description of Relationship

If sediment yields change, the frequency of dredging needed in channels and estuaries may change.

Summary of Evidence

If a yearly sediment yield estimate can be made (see link 1b), then an updated frequency of dredging can be estimated based on past dredging in the area. Past dredging amounts and frequencies can be used for comparison with new estimated sediment yields. If the amount of sediment removed during past dredging operations is known, then it will be possible to calculate how long it will take for sediment to build up to that level, based on the new yearly sediment yield estimate.

Other Factors

None.

Strength of Evidence

Moderate. A change in sediment yield can result in altered dredging needs; however, local data on past dredging are needed to estimate this change.

Predictability: If past dredging amounts and frequencies are known and an updated sediment yield estimate is available, updated dredging frequencies could be calculated.

Sources

None—local data are required.

1d: Sediment Accumulation → Turbidity

Description of Relationship

Accretion of sediments by the salt marsh reduces suspended sediment concentrations in the water column, and therefore decreases the turbidity of the water.

Summary of Evidence

It is known that sediment deposition in upstream freshwater wetlands benefits downstream water quality by reducing turbidity and suspended solid concentration (Johnston 1991). The cumulative effect of sediment retention by individual wetlands can have important water quality effects at the watershed scale (Johnston 1991). Data specific to salt marsh environments are scarce. One study estimated that 5%–11% of the total annual sediment input to the Chesapeake Bay was trapped by surrounding estuarine marsh areas (Stevenson et al. 1988), whereas a previous study estimated that value to be 15% (Nixon 1980). Although these studies do not make the direct link between marsh sediment accretion and turbidity levels, they do provide evidence that estuarine marshes are able to reduce the amount of sediment entering an estuary by a measurable amount.

Other Factors

None.

Strength of Evidence

Low. No evidence was found to directly link salt marsh sediment accumulation to any change in turbidity level of the surrounding estuary. Extrapolations about this link can be made from freshwater wetland environments, but there are ecological differences between freshwater wetlands and salt marshes that make extrapolating difficult. A notable difference is the source of input sediments; sediment sources for freshwater wetlands are limited to overland flow/runoff/erosion, whereas salt marsh sediment sources include fluvial sediments as well as sediments originating from marine waters. No methods to predict a change in turbidity resulting from salt marsh sediment accumulation were found.

Predictability: Predicting turbidity of an estuary on the basis of sediment accumulation by a marsh is not easy due to the low level of available evidence linking these two nodes.

Sources

Johnston, Carol A. 1991. "Sediment and Nutrient Retention by Freshwater Wetlands: Effects on Surface Water Quality." *Critical Reviews in Environmental Science and Technology* 21 (5–6): 491–565.

Nixon, Scott W. 1980. "Between Coastal Marshes and Coastal Waters—a Review of Twenty Years of Speculation and Research on the Role of Salt Marshes in Estuarine Productivity and Water Chemistry." In *Estuarine and Wetland Processes,* edited by P. Hamilton and K.B. Macdonald, 437–525. Boston, MA: Springer. http://link.springer.com/10.1007/978-1-4757-5177-2_20.

Stevenson, J. Court, Larry G. Ward, and Michael S. Kearney. 1988. "Sediment Transport and Trapping in Marsh Systems: Implications of Tidal Flux Studies." *Marine Geology* 80 (1–2): 37–59.

1e: Turbidity → Light Attenuation

Description of Relationship

A change in turbidity will change light attenuation into the water column (which changes the amount of light reaching through the water). Lower turbidity reduces the scatter of light entering estuary water, which will increase the amount of light able to penetrate the water column (corresponding to a decrease in light attenuation).

Summary of Evidence

Turbidity is an optical determination of water clarity and is a measure of the amount of light scattered by suspended particles in water. The more suspended particles in the water, the more those particles will scatter incoming light. Light attenuation represents the reduction of intensity of a beam of light traveling through a medium, such as water. Therefore, with less turbidity (scatter of light from suspended particles), light attenuation will correspondingly decrease, resulting in more available light to reach the water column.

There is an understood connection between measurements of turbidity (using a Secci disk) and light attenuation of water (measured by the light attenuation coefficient, K_d). Predictions of K_d from Secci depth measurements can be made using an index, represented by the equation below.

$$K_d = a/Z_{secchi}$$

Where a is a constant derived from reflectance properties, and Z_{secchi} is the secchi depth, measured in meters

The constant a is often considered to be 1.7 (Padial and Thomaz 2008); however, it has been shown to vary on the basis of site characteristics. Smith et al. (2006) used a values of 1.0, 1.4, and 1.7 for naturally turbid, moderately turbid, and clear water estuaries, respectively (see Smith et al. 2006 formulas 2 and 3 for further detail). Liu et al. (2005) did a literature review of a values and found them to range between 1.1 and 2.02.

If both Secchi depths and light meter readings with resulting K_d values are available for a site, statistical models relating Secchi depth and K_d can be developed. These models enable the user to predict K_d values from known Secchi depths. Such models can be seen in Padial and Thomaz (2008), Devlin et al. (2008), and Liu et al. (2005).

Other Factors
Water flow and weather are other factors. Water with a high flow rate will prevent suspended particles from settling on the bottom and maintain higher levels of turbidity. Weather events that result in higher stream flows will often be associated with temporary higher turbidity levels due to increased flow as well as particle runoff.

Strength of Evidence
High. The literature shows that there is general consensus about the relationship between turbidity and light attenuation as well as about the relationship between Secchi depth and the light attenuation coefficient, K_d. Using common values of the constant a, it is possible to calculate K_d from known Secchi depths; however, local data that can provide specific values of a or generate a local model are preferred.

Predictability: Using local data on Secchi depth and light meter readings to generate a statistical model that links these two variables has been shown to be successful, enabling the user to predict K_d values on the basis of Secchi depths.

Sources

Devlin, M.J., J. Barry, D.K. Mills, R.J. Gowen, J. Foden, D. Sivyer, and P. Tett. 2008. "Relationships between Suspended Particulate Material, Light Attenuation and Secchi Depth in UK Marine Waters." *Estuarine, Coastal and Shelf Science* 79 (3): 429–439.

Fondriest Environmental Inc. 2014. "Turbidity, Total Suspended Solids & Water Clarity." Fundamentals of Environmental Measurements. June. http://www.fondriest.com/environmental-measurements/parameters/water-quality/turbidity-total-suspended-solids-water-clarity/.

NOAA (National Oceanic and Atmospheric Administration). 2017. "Turbidity." NOAA Ocean Service Education. https://oceanservice.noaa.gov/education/kits/estuaries/media/supp_estuar10e_turbinity.html.

Padial, André Andrian, and Sidinei Magela Thomaz. 2008. "Prediction of the Light Attenuation Coefficient through the Secchi Disk Depth: Empirical Modeling in Two Large Neotropical Ecosystems." *Limnology* 9 (2): 143–151.

Smith, Lisa M., Virginia D. Engle, and J. Kevin Summers. 2006. "Assessing Water Clarity as a Component of Water Quality in Gulf of Mexico Estuaries." *Environmental Monitoring and Assessment* 115 (1): 291–305.

1f: Light Attenuation → Primary Production
Description of Relationship
In light limited systems, primary production by phytoplankton is directly linked to the amount of light available in the water column.

Summary of Evidence
When nutrient availability in estuaries is adequate to support primary production, the limiting factor of phytoplankton photosynthesis is then considered to be light (Cole and Cloern 1987). Cole and Cloern (1987) performed a meta-analysis on data from six estuaries around the United States and found that daily phytoplankton primary production is directly related to a composite parameter consisting of phytoplankton biomass, photic depth, and surface irradiance (i.e.,

planktonic primary production is linked to plankton biomass and light availability). (Note: photic depth is calculated using the light attenuation coefficient, connecting this relationship directly to link 1e). This study found that 82% of variance in primary production was explained by this composite parameter, indicating the high importance of planktonic biomass and light in governing variations in primary production in estuarine environments. Using this relationship, it may be possible to predict primary production levels on the basis of known plankton biomass (measured by chlorophyll a) and expected light attenuation measures. Cole and Cloern (1987) found that:

$$P = 150 + 0.73(BZ_pI_o) \text{ [see figure 3 in the paper]}$$

Where, P = production (mg C/ m^2d), B = biomass of phytoplankton (mg chlorophyll a/m^3), Z_p = photic depth (m), and I_0= surface irradiance (solar power/area/time, variable units)

Other Factors
Salinity: Salinity can influence the primary production rates of phytoplankton; however, the exact relationship between salinity and primary production depends on the phytoplankton community (Qasim et al. 1972; Lionard et al. 2005).

Nutrients: It is possible for nitrogen or phosphorus limitation to dictate phytoplankton production (Howarth 1988).

Strength of Evidence
Fair. The meta-analysis described here studied only six estuaries, but the explanatory power of the statistical model was high. Using the equation developed by Cole and Cloern (1987) and site-specific data on phytoplankton biomass, photic depth, and surface irradiance, it is possible to calculate primary production for an estuary. Knowledge of whether the estuary is more light or nutrient limited will determine if the relationship between light and planktonic production is important to consider; importantly, factors other than light can influence the productive capability of plankton. The variety of factors that influence primary production also limit the applicability and consistency of this link—light availability is only one piece of the equation.

Predictability: Existing equations make it possible to predict planktonic production on the basis of light availability, but additional site-specific data are required to complete the calculation.

Example. A review of phytoplankton primary production of estuarine systems provides a comprehensive collection of global studies that have measured planktonic primary production (Cloern et al. 2014, see supplementary materials for list of individual studies). It is possible to monitor primary production by measuring ^{14}C assimilation, ^{13}C assimilation, or oxygen production rates. This review does note there is an interaction of factors that influence planktonic production, most importantly nutrients, light, temperature, and plankton biomass (Cloern et al. 2014). Measuring plankton production therefore does not necessarily prove the link between light availability and production, but if light data are also collected, it would be possible to develop a statistical model using light availability and production data to evaluate that relationship.

Sources
Cole, Brian E., and James E. Cloern. 1987. "An Empirical Model for Estimating Phytoplankton Productivity in Estuaries." *Marine Ecology Progress Series* 36 (1): 299–305.
Cloern, James E., S. Q. Foster, and A. E. Kleckner. 2014. "Phytoplankton Primary Production in the World's Estuarine-Coastal Ecosystems." *Biogeosciences* 11 (9): 2477–2501.
Howarth, Robert W. 1988. "Nutrient Limitation of Net Primary Production in Marine Ecosystems." *Annual Review of Ecology and Systematics* 19 (1): 89–110.
Lionard, Marie, Koenraad Muylaert, Dirk Van Gansbeke, and Wim Vyverman. 2005. "Influence of Changes in Salinity and Light Intensity on Growth of Phytoplankton Communities from the Schelde River and Estuary (Belgium/The Netherlands)." *Hydrobiologia* 540 (1–3): 105–115.
Qasim, S.Z., P.M.A. Bhattathiri, and V.P. Devassy. 1972. "The Influence of Salinity on the Rate of Photosynthesis and Abundance of Some Tropical Phytoplankton." *Marine Biology* 12 (3): 200–206.
Walsh, P., and L. Legendre. 1983. "Photosynthesis of Natural Phytoplankton under High Frequency Light Fluctuations Simulating Those Induced by Sea Surface Waves." *Limnology and Oceanography* 28 (4): 688–697.

1g: Primary Production → Wildlife Populations (fish)

Description of Relationship

Primary production by phytoplankton forms the basis of the marine food chain, and changes in primary production will affect species in higher trophic levels. There is a generally acknowledged positive correlation between primary production by phytoplankton in ocean water and fish biomass; however, the exact numerical relationship is not agreed on.

Summary of Evidence

Multiple modeling exercises have found complimentary results, indicating that phytoplanktonic primary production is positively correlated to fish biomass/production. A dynamical size-based food web model used for large marine ecosystems showed that primary production by phytoplankton is linked to fish biomass and production, corroborating empirical work that found similar patterns (Blanchard et al. 2012). Applications of a marine food web model applied to Australian marine systems similarly found that primary production is linked to fisheries and that, in general, model outputs showed increases in fish landings corresponded with increases in primary production, proportional to the size of primary production increase (Brown et al. 2009). However, the authors acknowledge that there are many other complex interactions that factor into this link.

Using fisheries catch data and satellite-derived levels of planktonic primary production, a global analysis showed that primary production limits fisheries' catch at the scale of large marine ecosystems or LMEs (Chassot et al. 2010). In a study of the North American West Coast (Alaska to southern California), it was found that variation in alongshore primary production by phytoplankton was highly correlated with alongshore variation in fish yield, explaining up to 87% of the variation in fish production (Ware and Thomson 2005).

Other Factors

Other Primary Production Sources: Many of the modeling exercises that display the relationship between planktonic production and fish production do not take into account other primary production sources. In coastal systems, primary production can also come from salt marshes, mangroves, or sea grass. The models ignore these sources because primary production from these coastal sources is such a relatively small percentage of total marine primary production (~5.5%) (Blanchard et al. 2012). However, in the case of this conceptual model, which focuses on salt marsh ecosystems, the relative importance of coastal primary production from a marsh might be more important.

Scale: The issue of scale is important in this case. Many of the studies reported here examined fish populations on very large scales, often at a global, continental, or large marine ecosystem scale. It is therefore unclear whether the effects of altered primary production in one estuary or part of an estuary, related to a salt marsh restoration or improvement, would have these same kinds of impacts. Because no direct, numerical relationship can be applied here, this issue of scale is not an easy one to resolve and should be considered carefully.

Top-Down Influences: The magnitude of the effect of changing primary production levels on fish biomass can be different for different functional groups or trophic levels in the ecosystem. This magnitude can be due to top-down (predator-mediated) influences on the ecosystem as well as to other interactions among species (Brown et al. 2009). In fact, one literature review found contradictory results and reported that there was actually weak coupling amoung phytoplankton, herbivores, and higher trophic levels, and it hypothesized the reason was resource and consumer interactions, species interactions, and other top-down controls (Micheli 1999).

Strength of Evidence

Low. Data and modeling show that the link between planktonic primary production and fish stocks does exist. However, numerical predictions are site/region-specific and will require local data. Other factors in this relationship (e.g., top-down influences) must be taken into consideration. Modeling of this relationship has been done in coastal areas, but mostly at a very large (continental, LME) scale that is less applicable to site-based protection or restoration of salt marshes. Additionally, the question remains whether primary production by salt marsh plants would alter the relationship between planktonic primary production and wildlife and at what scale this phenomenon might be relevant.

Predictability: Although there are models to predict fish stocks on the basis of planktonic production, additional site-specific data are required. Numerous other factors influence fish stock levels, making predictions based on planktonic production difficult.

Sources

Blanchard, Julia L., Simon Jennings, Robert Holmes, James Harle, Gorka Merino, J. Icarus Allen, Jason Holt, Nicholas K. Dulvy, and Manuel Barange. 2012. "Potential Consequences of Climate Change for Primary Production and Fish Production in Large Marine Ecosystems." *Phil. Trans. R. Soc.* B 367 (1605): 2979–2989.

Brown, C.J., E.A. Fulton, A.J. Hobday, R.J. Matear, H.P. Possingham, C. Bulman, V. Christensen, et al. 2010. "Effects of Climate-Driven Primary Production Change on Marine Food Webs: Implications for Fisheries and Conservation." *Global Change Biology* 16 (4): 1194–1212.

Chassot, Emmanuel, Sylvain Bonhommeau, Nicholas K. Dulvy, Frédéric Mélin, Reg Watson, Didier Gascuel, and Olivier Le Pape. 2010. "Global Marine Primary Production Constrains Fisheries Catches." *Ecology Letters* 13 (4): 495–505.

Micheli, Fiorenza. 1999. "Eutrophication, Fisheries, and Consumer-Resource Dynamics in Marine Pelagic Ecosystems." *Science* 285 (5432): 1396–1398.

Ware, Daniel M., and Richard E. Thomson. 2005. "Bottom-up Ecosystem Trophic Dynamics Determine Fish Production in the Northeast Pacific." *Science* 308 (5726): 1280–1284.

1h: Change in Salt Marsh Quantity or Quality → Nutrient Retention

Description of Relationship

Salt marsh retains and absorbs nutrients from water draining into an estuary. Salt marsh habitat can remove and absorb nutrients like nitrogen and phosphorous. Table 1 summarizes the relevant literature on values for nutrient retention/accumulation by salt marshes.

Table 3. Nitrogen and phosphorous accumulation rates in salt marsh habitats

Marsh location (salinity)	N accumulation (g/m²/yr)	P accumulation (g/m²/yr)	Study
Georgia (fresh tidal marsh, salinity <0.5 psu)	8.2	0.69	Loomis and Craft (2010)
Georgia (brackish marsh, salinity = 0.5-15 psu)	6.5	1.02	Loomis and Craft (2010)
Georgia (salt marsh, salinity >15 psu)	2.4	0.29	Loomis and Craft (2010)
Louisiana (salinity not available	13-21	0.8-1.7	DeLaune et al. (1981)
Maine (salinity = 28 psu)	2.8-6.7	n/a	Drake et al. (2015)
Massachusetts (salinity = 20 psu)	5.7-11.3	n/a	Drake et al. (2015)
New York (salinity =12 psu)	5.3-7.6	n/a	Drake et al. (2015)
New Jersey (salinity = 18 psu)	3.8-8.8	n/a	Drake et al. (2015)
Georgia, restored marsh (salinity=25-35 ppt)	6.3	0.6	Craft (2001)
Georgia, natural marsh (salinity= 25-35 ppt)	5.5	0.35	Craft (2001)
North Carolina, restored marsh			Craft et al. (1999)
(salinity = 25-35 ppt)	7.1	0	
(salinity = 7-10 ppt)	11.5	0.9	
North Carolina, natural marsh			Craft et al. (1999)
(salinity = 25-35 ppt)	4.5	0	
(salinity = 7-10 ppt)	2.2-3.9	0-0.3	
Rhode Island (salinity = 20-33 ppt)	0.46	0.12	Nixon et al. (1986)

Summary of Evidence

A review of the relevant literature provided the nitrogen and phosphorous accumulation rates seen in the Table 1. One study in Georgia also estimated that the salt marshes studied removed 13%–32% of the nitrogen entering estuaries from terrestrial sources (Loomis and Craft 2010). There are relatively few studies reporting this type of data, as experts have

noted. The Nature Conservancy's website, "Mapping Ocean Wealth," states that "[salt marshes] are also important for soaking up nitrogen, fueling their growth and, in so doing, reducing […] the amount of nitrogen that passes by into open waters. To date, there has been no global synthesis of these services from coastal wetlands" (The Nature Conservancy 2016).

Other Factors
Soil Characteristics: Various soil characteristics such as soil type, soil density, and redox potential can influence the rate of nutrient retention in a salt marsh (DeLaune 1981; Loomis and Craft 2010; Drake et al. 2015).

Salinity: A study along a salinity gradient in Georgia salt marshes found that salinity affects the rate of nutrient accumulation in marsh soils. Nitrogen accumulation is negatively correlated with salinity (i.e., increasing salinity means that there will be less nitrogen accumulation), whereas phosphorous accumulation is highest at mid-range salinities (Loomis and Craft 2010).

Strength of Evidence
Moderate. Though there are published nutrient accumulation rates, it is clear that individual marsh sites have varying rates. Although it is possible to say with high certainty that a marsh will retain nutrients, knowing how much accumulation occurs depends on the site. Therefore, consistency of nutrient retention is high, but consistency of the magnitude of this relationship is low.

Predictability: It would be possible to extrapolate nutrient accumulation rates on the basis of published data, but accurate nutrient accumulation would likely be best calculated using site-specific measurements and data.

Example. Calculation of nutrient accumulation rates using standard methods at a specific site is well documented and accepted in the literature. Field studies such as the ones performed in Loomis and Craft (2010), DeLaune et al. (1981), and Drake et al. (2015) provide details about nutrient accumulation measurement methods at marsh sites.

Sources
Craft, Christopher B. 2001. "Soil Organic Carbon, Nitrogen, and Phosphorus as Indicators of Recovery in Restored 'Spartina' Marshes." *Ecological Restoration* 19 (2):87–91.

Craft, Christopher, Judy Reader, John N. Sacco, and Stephen W. Broome. 1999. "Twenty-Five Years of Ecosystem Development of Constructed Spartina Alterniflora (Loisel) Marshes." *Ecological Applications* 9 (4):1405–1419.

DeLaune, R.D., C.N. Reddy, and W.H. Patrick. 1981. "Accumulation of Plant Nutrients and Heavy Metals through Sedimentation Processes and Accretion in a Louisiana Salt Marsh." *Estuaries and Coasts* 4 (4): 328–334.

Drake, Katherine, Holly Halifax, Susan C. Adamowicz, and Christopher Craft. 2015. "Carbon Sequestration in Tidal Salt Marshes of the Northeast United States." *Environmental Management* 56 (4): 998–1008.

Loomis, Mark J., and Christopher B. Craft. 2010. "Carbon Sequestration and Nutrient (Nitrogen, Phosphorus) Accumulation in River-Dominated Tidal Marshes, Georgia, USA." *Soil Science Society of America Journal* 74 (3): 1028–1036.

Nixon, S.W., C.D. Hunt, and B. L. Nowicki. 1986. "The Retention of Nutrients (C, N, P), Heavy Metals (Mn, Cd, Pb, Cu), and Petroleum Hydrocarbons in Narragansett Bay." *Elsevier Oceanography Series* 43:99–122.

The Nature Conservancy. 2016. "Filtration." Mapping Ocean Wealth. http://oceanwealth.org/ecosystem-services/filtration/.

1i: Nutrient Accumulation → Nutrients in Estuary Water
Description of Relationship
As the marsh environment accumulates nitrogen (N) and phosphorus (P), the amount and rate of N and P delivery to estuarine waters could be altered.

Summary of Evidence
Nutrient delivery to an estuary could be altered by nutrient accumulation in the salt marsh environment. In a systematic review of nutrient removal by created and restored freshwater wetlands, it was found that average removal efficiency (measured in percent of total load) for total nitrogen was 39% and average removal efficiency for total phosphorous was 41% (Land et al. 2016). The review extracted data on 203 freshwater wetlands from 93 studies, with most of the data coming from Europe and North America. The study focused on nutrient removal by wetlands from wastewater and

urban and agricultural runoff. Removal rates for both nitrogen and phosphorous were found to be highly dependent on the loading rate of those nutrients into a wetland. No similar review could be found for salt marshes; however, one study estimated that tidal marshes collectively removed 13%–32% of total nitrogen entering estuaries in Georgia (Loomis and Craft 2010).

Other Factors

Loading rate of nutrients is another factor. Both N and P removal efficiencies were found to vary with the hydrologic loading rate of the nutrients into a wetland area. Wetlands with precipitation-driven loading, hydrologic pulses, or both showed lower total nutrient removal efficiency.

Temperature

Nitrogen removal efficiency was found to be correlated with air temperature. In general, nitrogen removal efficiency was positively correlated with average air temperature.

Strength of Evidence

Low. Though the systematic review examining nutrient removal by wetlands is extensive, the extrapolations that must be made to apply this information to a salt marsh environment make this evidence less useful. Careful considerations must be taken into account when extrapolating the results found in the Land et al. (2016) study to salt marsh environments. First, the Land et al. (2016) study was conducted on freshwater wetlands, not salt marshes. The most frequent wetland type used in the study was emergent wetlands (n=117), followed by mixed (n=28) and submerged wetlands (n=24). Second, the Land et al. (2016) study examined only constructed or restored wetlands, not existing wetlands. Third, to make extrapolations from the Land et al. (2016) study, the total load of nutrients entering a wetland must be known.

Predictability: Predicting levels of nutrients in estuary water on the basis of salt marsh accumulation of those nutrients is not easy, because the applicability of the evidence collected is low (due to differences between freshwater wetlands and salt marshes).

Sources

Land, Magnus, Wilhelm Granéli, Anders Grimvall, Carl Christian Hoffmann, William J. Mitsch, Karin S. Tonderski, and Jos TA Verhoeven. 2016. "How Effective Are Created or Restored Freshwater Wetlands for Nitrogen and Phosphorus Removal? A Systematic Review." *Environmental Evidence* 5 (1): 9.

Loomis, Mark J., and Christopher B. Craft. 2010. "Carbon Sequestration and Nutrient (Nitrogen, Phosphorus) Accumulation in River-Dominated Tidal Marshes, Georgia, USA." *Soil Science Society of America Journal* 74 (3): 1028–1036.

1j: Nutrients in Estuary Water → Primary Production

Description of Relationship

The ratio of nutrients in estuarine waters can determine which nutrient is most limiting for primary production. Phytoplankton require nitrogen and phosphorus at a ratio of 16:1 (moles) for maximum productivity. Lower ratios indicate nitrogen limitation; higher ratios indicate phosphorous limitation (Howarth 1988).

Summary of Evidence

A review of nutrient limitation in marine ecosystems by Howarth (1988) as well as a report on understanding and reducing the effects of nutrient pollution by the National Research Council (2000) provide a good basis for this link. Nitrogen is most often considered the limiting nutrient in seawater, but others have argued that phosphorus is limiting or that limitation switches from nitrogen to phosphorus seasonally (Howarth 1988; NRC 2000), and Howarth (1988) notes that in systems where nutrients are limiting, the N to P ratio of nutrient inputs is a major factor that determines which nutrient is most limiting.

Phytoplankton maximum productivity can be seen when the molar ratio of N to P is 16:1. This is called the Redfield ratio, while accepted, has some subtle variations in certain circumstances (Howarth 1988; NRC 2000; Gruber and Deutsch 2014). Generally, ratios lower than 16:1 indicate nitrogen limitation, whereas higher ratios indicate phosphorous limitation.

There are multiple ways to evaluate whether a system is nutrient limited and if so, which nutrients are limiting for primary production. A common method is through bioassay experiments, which involves taking water samples from an estuary,

adding different nutrient amounts, and measuring production rates of plankton (see Howarth 1988, 92). However, it has been noted that laboratory conditions of bioassays may not capture the extent of biogeochemical processes occurring in the marine environment and therefore could produce misleading results in some cases. Other methods for determining nutrient limitation include examining the N:P ratio of nutrient inputs to an estuary, or manipulating nutrient levels for mesocosm experiments *in situ* and measuring planktonic production (see Howarth 1988, 94, and NRC 2000, chapter 3).

Other Factors
Biogeochemical processes in the marine environment can alter the N:P ratio. Removal of N, P, or both can occur due to processes such as denitrification, sedimentation of N in zooplankton fecal pellets, or microbial decomposition (see Howarth 1988, 97–99, and NRC 2000, chapter 3, for more details).

Strength of Evidence
Low. The Redfield ratio is widely accepted in the scientific community; however variations in the ratio have been noted. Using the ratio, it is possible to predict whether a system is N or P limited; however, there are confounding factors, primarily biogeochemical conditions. Using the ratio simply tells you whether the system is N or P limited, but not how primary production rates will actually change on the basis of the ratio. Using the ratio, it is therefore possible to estimate whether primary production is occurring at its maximum capacity, but not what the actual rate of production is. Our inability to estimate the production rate severely limits the applicability of this evidence.

Predictability: The predictability of primary production based on nutrients is limited by the multitude of additional variables that influence planktonic production. Site-specific measurements will likely be needed to make predictions about primary production levels (see example below).

Example. A review of phytoplankton primary production of estuarine systems provides a comprehensive collection of global studies that have measured planktonic primary production (Cloern et al. 2014, see supplementary materials for list of individual studies). It is possible to monitor primary production by measuring ^{14}C assimilation, ^{13}C assimilation, or oxygen production rates. This review does note that there is an interaction of factors that influence planktonic production, most importantly nutrients, light, temperature, and plankton biomass (Cloern et al. 2014). Measuring plankton production therefore does not necessarily prove the link between nutrients and production, but if data on nutrients are also collected, it would be possible to develop a statistical model using nutrient and production data to evaluate that relationship.

Sources
Cloern, James E., S.Q. Foster, and A.E. Kleckner. 2014. "Phytoplankton Primary Production in the World's Estuarine-Coastal Ecosystems." *Biogeosciences* 11 (9): 2477–2501.

Gruber, Nicolas, and Curtis A. Deutsch. 2014. "Redfield's Evolving Legacy." *Nature Geoscience* 7 (12): 853–55. doi:10.1038/ngeo2308.

Howarth, Robert W. 1988. "Nutrient Limitation of Net Primary Production in Marine Ecosystems." *Annual Review of Ecology and Systematics* 19 (1): 89–110.

NRC (National Research Council) Board of Ocean Studies. 2000. Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution. Washington D.C.: National Academies Press.

1k: Nutrients in Estuary Water → Algae Blooms
Description of Relationship
Nutrient levels in estuary water can influence the likelihood that an algae bloom will occur. Algae blooms can occur because of "overfeeding" of algae by nutrient runoff (NOAA NOS 2017). This phenomenon occurs if phosphorus, nitrogen, and carbon levels are discharged to a waterbody at a rate that causes increased algae growth.

Summary of Evidence
Algae blooms are a form of excessive primary production. Under the proper conditions, high levels of nutrients can stimulate explosive growths of algae, resulting in algae blooms (Howarth et al. 2000; NRC 2000; NOAA OSE 2017).

Some algal blooms can be categorized as harmful algal blooms (HABs) because of the damaging biotoxins they produce. Numerous studies from estuarine systems around the world find a positive correlation between nutrient runoff from anthropogenic activities to an increase in HAB frequency (Hallegaeff 1993; Sellner et al. 2003; Gilbert et al. 2007; Heisler et al. 2008). Harmful algae blooms have received a lot of attention in the United States, resulting in national-level assessments

of trends in HAB occurrences (Bricker et al. 2008; Anderson et al. 2008) as well as government-funded working groups, committees, and preparedness plans for assessing, predicting, and handling HAB events (Ramsdell et al. 2005; Jewett et al. 2007; Jewett et al. 2008). In 2003, the U.S. Environmental Protection Agency held an expert roundtable discussion on the relationship between HABs and nutrients, and developed the following relevant consensus statements:

- Degraded water quality from increased nutrient pollution promotes the development and persistence of many HABs and is one of the reasons for their expansion in the United States and other nations.

- The composition—not just the total quantity—of the nutrient pool impacts HABs.

- High-biomass blooms must have exogenous nutrients to be sustained.

- Both chronic and episodic nutrient delivery promote HAB development.

- Management of nutrient inputs to the watershed can lead to significant reduction in HABs (Heisler et al. 2008).

Nuances in these statements should be considered when thinking about HABs at a site-specific level. More detail and supporting evidence for these statements can be found in Heisler et al. (2008).

Algae blooms are difficult to predict because a complex network of biotic and abiotic factors create conditions appropriate for a bloom. A neural network modeling approach has been used to attempt to predict coastal algal blooms, but this technique has primarily been used in freshwater systems (Lee et al. 2003). Research on specific bloom types, such as red tide in Florida, is progressing to the point that factors supporting bloom development are becoming well enough understood to support reasonable predictions. NOAA has developed models that form a HAB Monitoring System to forecast red tide occurrences in the Gulf of Maine, Gulf of Mexico, and Lake Erie. The monitoring system is meant to minimize HAB impacts on public health and coastal economies (Stumpf et al. 2003; NOAA 2013). By using HAB observational data from monitoring networks and linking them to optically based (remotely sensed) models, it is possible to predict where HABs may occur (Sellner et al. 2003; Stumpf et al. 2003; Stumpf et al. 2009). Linking general circulation models to algal biological models can also predict HAB distributions; however, developing these models is quite difficult (Sellner et al. 2003; Heisler et al. 2008). There are two general HAB model types: (1) models that predict general likelihood of occurrence, and (2) models that create explicit HAB predictions in time or space (Heisler et al. 2008). Models that have had relative success in predicting harmful algae blooms have been developed for the Gulf of Maine, Florida, the Gulf of Mexico, and several sites in Europe (Sellner et al. 2003; Heisler et al. 2008). These complicated simulation models have many inputs in addition to nutrient levels.

Other Factors

Abiotic Factors: Algae blooms are not caused by nutrients alone. They can occur any time conditions are right for either micro or macroalgae to grow out of control (NOAA NOS 2017). Some estuarine systems are more susceptible to algae blooms than others due to hydrodynamic and other physical factors. Chief among these are light availability to support photosynthesis and frequency with which the estuary is flushed due to runoff, tidal flushing, or wind mixing (Howarth et al. 2000; Ferreira et al. 2005; Bricker et al. 2007). If flushing rates are high, algae growth cannot keep up with dilution, and cells are lost from the system limiting bloom development. Other hydrodynamic conditions such as development of salinity, temperature-driven frontal zones, or wind-driven accumulation of surface scums can also concentrate cells facilitating bloom development and maintenance. High temperatures also tend to favor blooms of cyanobacteria and certain toxic dinoflagellate species by promoting high growth rates (NOAA NOS 2017).

Nutrient Ratios and Speciation: A relationship between changing nutrient compositions and harmful algae blooms has been determined, though neither the quantity nor ratio of inorganic nutrients can explain fully when and where a harmful algae bloom will occur (see section 2.2 of Heisler et al. 2008 for more detail). The speciation of nutrients is also a factor as shown in the relationship of dissolved organic nitrogen (DON) and blooms of the brown tide organism, *Aureococcus anophagefferens.* This harmful algal bloom species preferentially uses dissolved organic nitrogen for its nutrient rather than inorganic nitrogen forms and thus is a symptom of organic- rather than inorganic-driven eutrophication (Glibert et al. 2007).

Algae Grazers: Algal grazing by zooplankton and benthic suspension feeders can help control explosive algae growth (Cloern 1982; Howarth et al. 2000).

Strength of Evidence
Moderate. There is strong evidence to suggest that links between nutrient levels and algal blooms exist; however, that evidence does not extend easily to predicting when or where those blooms will occur. Additionally, the literature suggests that these blooms are dependent on many factors, some of which are difficult to measure accurately, to model accurately, or both, and it is hard to predict when nutrients will reach a threshold level that would result in a bloom.

Predictability: Models described in Lee et al. (2003), Sellner et al. (2003) and Heisler et al. (2008) have shown success in modeling and, in some cases, predicting different types of algae blooms. However, these are highly complex models that are likely inaccessible except to expert users. These models have numerous inputs, including nutrients, which are by no means the only predictive factor incorporated.

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1l: Algae Blooms → Dissolved Oxygen

Description of Relationship

Algae blooms can deplete dissolved oxygen because of high respiration rates by algae, but most often oxygen depletion is due to bacterial respiration during decay of the bloom (Hallegraeff 1993; Howarth et al. 2000; Sellner et al. 2003).

Summary of Evidence

As algae from a bloom dies, bacterial decomposition of the algae takes place and this decomposition process uses dissolved oxygen in the water. When the rate of oxygen consumption by bacteria exceeds the supply of oxygen provided by the environment, dissolved oxygen levels can drop and result in hypoxic conditions (Howarth et al. 2000; National Science and Technology Council 2016). Additionally, algal blooms can reduce water clarity to a level that prevents sunlight from reaching submerged aquatic plants, reducing their ability to photosynthesize and produce oxygen (Howarth et al. 2000; Bricker et al. 2008; National Science and Technology Council 2016). Jewett et al. (2010) provide a good overview of hypoxia in U.S. coastal waters.

Some models predict dissolved oxygen levels in relation to events such as algae blooms. Most models have been developed for specific locations, such as the ChesROMS model for the Chesapeake Bay (Wiggert et al. 2017). Dissolved oxygen is just one of many outputs of this model, and inputs include data on sediments, atmospheric deposition, nutrient and dissolved organic matter inputs, and benthic interactions. Algal blooms are not a singular predictor of dissolved oxygen, and the model's complexity incorporates the various interactions between biotic and abiotic factors that result in dissolved oxygen levels (Wiggert et al. 2017). The model has been validated using fine-scale dissolved oxygen data, and it has been found to accurately represent dissolved oxygen fluctuations at various sites in the Chesapeake Bay.

Other Factors

Stratified layers of water will often have different levels of dissolved oxygen. Usually, deeper layers of water will contain less oxygen because of reduced oxygen exchange with the air.

Strength of Evidence

Moderate. The literature strongly suggests that high biomass blooms will cause a reduction in dissolved oxygen. But it does not necessarily follow that the reduction will reach a critical threshold level with cascading ecological effects.

Predictability: There is no good way to predict exactly how much high biomass algae blooms will reduce dissolved oxygen levels. Therefore, the magnitude of that effect cannot be predicted.

Example. Models can predict dissolved oxygen levels for a specific estuary, but these models are often highly complex and are developed to work at a specific site. One such model is ChesROMS, applicable for the Chesapeake Bay (Wiggert et al. 2017). Transferability of such a model would be difficult, limiting the model's more generalized use.

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1m: Algae Blooms → Toxins
Description of Relationship
Many harmful algae blooms (HABs) produce biotoxins. The severity and type of the bloom will determine the amount and type of toxin produced.

Summary of Evidence
Harmful algae blooms produce a wide variety of toxins, and the toxin produced depends on the species of algae that blooms. The major classes of HAB toxins include saxitoxins (PSP), brevetoxins (NSP), domoic acid (ASP), okadaic acid/dinophysistoxins (DSP), azaspiracids (AZP), ciguatoxins (CFP), and microcystins/ anatoxins/ cylindrospermopsin (CTP) (Sellner et al. 2003). The toxins are produced by algae, but they persist in marine waters, and in other organisms in the marine environment, through trophic transfer (Sellner et al. 2003). In some cases of toxin accumulation in marine organisms, toxins can be metabolized or biotransformed into slightly different compounds that can either be more or less toxic than the original compound (Sellner et al. 2003).

Direct detection of algal toxins can be done in three primary ways, (1) chemical analysis, (2) in vitro assays, and (3) in vivo assays. See Sellner et al. (2003) for more detail on detection methods and equipment.

Other Factors
The amount of toxins produced during a harmful algae bloom have been shown to differ with different nutrient combinations and concentrations (Heisler et al. 2008).

Strength of Evidence
Moderate. The evidence strength of this link is somewhat unique because the outcome we care about is binary: either toxins are present, or they are not. It was given a moderate ranking because harmful algae blooms are often defined by the production of harmful toxins, so it follows that when a bloom occurs that toxins will often be present. However, we are not able to say with certainty what the concentration or extent of toxins will be because the threshold for toxicity varies with algal species. However, this moderate evidence ranking is conditional; the probability of toxin presence would be high *if* the algal bloom is a species that produces toxins. The type, potency, and concentration of toxins will depend on the algal species, spatial extent, cell count, and bloom length.

Predictability: If the algal species is known, the presence of certain toxins can be predicted with reasonably high certainty.

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1n: Dissolved Oxygen → Wildlife Populations
Description of Relationship
Below certain thresholds of dissolved oxygen (DO), wildlife species will be negatively affected. Marine wildlife requires dissolved oxygen in the water to survive, and when DO levels get too low, marine species can experience physiological stress or even death. Low oxygen conditions have also been shown to alter ecological communities.

Summary of Evidence

A review by Vaquer-Sunyer and Duarte (2008) summarizes the literature of DO thresholds for marine biodiversity. Though most sources will cite a critical threshold of 2.0 mg O_2/L as the hypoxia threshold, this review found that the threshold varies among different groups of marine organisms. The review of 872 studies on 206 marine species found that the median lethal DO threshold ranged from 8.6 mg O_2/L to 0 mg O_2/L (the 0 mg O_2/L threshold represented persistent resistance of total anoxia by an oyster species *Crassostrea viirginica*). The median of all lethal thresholds examined was 1.6 mg O_2/L. Median lethal DO threshold was shown to differ among the taxa examined (crustaceans, fish, bivalves, and gastropods); crustaceans have the highest threshold (i.e., lowest tolerance to low dissolved oxygen). The review's supplemental Table 3 contains the threshold for all individual species examined.

Howarth et al. (2000) discusses the effects of low oxygen on ecological communities. Frequent periods of low oxygen have been shown to change the seafloor community, shifting it from long-lived species like clams to more opportunistic and short-lived species like polychaete worms that thrive in between periods of hypoxia (Howarth et al. 2000). Food chains can be affected by low oxygen as well. Some zooplankton graze in shallow surface water at night and retreat to deeper water to avoid being eaten by fish, but if hypoxia persists in deeper waters, these zooplankton stay in shallow water during the day and are more susceptible to being preyed upon (Howarth et al. 2000).

Other Factors

Time: The length of exposure to low oxygen conditions can have varying impacts on different marine species, depending on the limits of their physiology (Vaquer-Sunyer and Duarte 2008). Vaquer-Sunyer and Duarte (2008) also report "lethal times" for various marine species. Lethal time refers to the length of time it takes for low DO levels to kill an organism. Some organisms die very quickly when exposed to low oxygen levels (minimum lethal time reported was less than 1 hour), whereas others can persist longer (maximum lethal time reported >1,000 hours). In some cases, relatively short occurrences of moderate hypoxia can temporarily change fish abundance and community structure, but not cause die-offs (Stevens et al. 2006).

Organism Mobility: The Vaquer-Sunyer and Duarte (2008) review found that organism mobility accounted for some of the variability in DO thresholds. Mobile organisms can attempt to escape hypoxic waters; sessile organisms do not have that ability. It has been suggested that sub-lethal effects of hypoxia on fish can be more influential than lethal effects, because fish are able to detect and avoid hypoxic areas. Therefore, hypoxic zones can greatly influence fish distributions as well as fish survival (Zhang et al. 2009).

Other Stressors: Hypoxia often occurs in combination with additional physiological stressors such as increased temperatures or other water contaminants. Most experiments included in the Vaquer-Sunyer and Duarte (2008) review isolate the impacts of low oxygen only, removing the additive effects of other stressful conditions. It is possible that combinations of stressors in addition to low dissolved oxygen would alter the lethal DO threshold for some organisms.

Strength of Evidence

Fair. There is little doubt that low DO levels in water will negatively affect wildlife species, causing physiological stress or death. The extent to which low dissolved oxygen will negatively affect certain species depends on their taxa, physiology, and mobility.

Predictability: Predicting outcomes for wildlife species will depend on site- and species-specific information. The Vaquer-Sunyer and Duarte (2008) review allows for estimations of critical DO thresholds for certain species. The Vaquer-Sunyer and Duarte (2008) review allows for estimations of critical DO thresholds for certain species, but assumptions must be made about how many in a wildlife population or what percentage of the population will die at that threshold level.

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1o: Toxins → Health

Description of Relationship

Toxins produced during a harmful algae bloom (HAB) can negatively affect human health. HAB toxins can harm humans in a variety of ways, but common symptoms include respiratory distress. Toxins can be spread by inhalation of marine aerosols, swimming in or accidentally ingesting contaminated water, and eating contaminated seafood.

Summary of Evidence

The National Science and Technology Council Subcommittee on Ocean Science and Technology released a report on harmful algae blooms. Appendix 1 and 2 of the report provide a summary of the toxins produced during different kinds of blooms, and those that impact human health are provided in Table 2. It should be noted that there are different vectors for HAB toxins; some directly impact human health by physical exposure or inhalation, and some are transmitted by consumption of contaminated seafood.

Table 4. Harmful algal taxa and effects on human health

HAB taxa	Toxin/bioactive compound	Human health effects	Vector	Impacted areas in United States
Karenia	Brevetoxins	Respiratory effects (acute eye irritation, respiratory distress, asthma exacerbation)	Marine aerosols	Gulf of Mexico
Karenia	Brevetoxins	Neurotoxic shellfish poisoning (nausea, vomiting, diarrhea, numbness, muscle aches, fever, chills, reduced heart rate)	Ingestion of contaminated shellfish	Gulf of Mexico, Atlantic coast up to North Carolina
Akashiwo sanguineium	Surfactants	Suspected respiratory irritant	Marine aerosols	Pacific coast
Macroalgae	H2S, dopamine	Respiratory effects	Marine aerosols	All coasts
Pseudo-nitzschia	Domoic acid	Amnesic shellfish poisoning (vomiting, diarrhea, abdominal pain, confusion, disorientation, memory loss)	Ingestion of contaminated shellfish	West Coast, Florida, Maine
Dinophysis; Prorocentrum	Okadaic acid, dinophysotoxins	Diarrhetic shellfish poisoning (nausea, vomiting, diarrhea, abdominal pain, chills, headache, fever)	Ingestion of contaminated shellfish	Oregon, Texas, Washington
Gambierdiscus; Fukuyoa	Ciguatoxins	Ciguatera fish poisoning (abdominal pain, nausea, vomiting, diarrhea, paresthesia, temperature dysesthesia, pain, weakness, bradycardia, hypotension)	Ingestion of contaminated fish	Florida, Gulf Coast, Hawaii, Pacific, Caribbean
Alexandrium; Gymnodinium; Pyrodinium bahamense	Saxitoxins	Paralytic shellfish poisoning (tingling, burning, numbness, drowsiness, incoherent speech, respiratory paralysis leading to death)	Ingestion of contaminated seafood	Pacific coast (including Alaska), Northeast Atlantic coast, Florida

Source: National Science and Technology Council (2016).

More detail on the types of health effects of four of the more common HAB toxins can be found in Hallegraeff (1993), summarized in Table 5.

Table 5. Clinical symptoms of various types of fish and shellfish poisoning

	Paralytic shellfish poisoning (PSP)	Diarrhetic shellfish poisoning (DSP)	Amnesic shellfish poisoning (ASP)	Ciguatera
Cuasative organism	*Alexandrium catenella; Alexandrium minimum; Alexandrium tamarense; Gymnodinium catenatum; Pyrodinium bahamense*	*Dinophysis acuminate; Dinopysis fortii*	*Nitzschia pungens* f. *multiseries; Nitzschia pseudodelicatissima; Nitzschia pseudoseriata*	*Gambierdiscus toxicus;* ? *Ostreopsis siamensis;* ? *Prorocentrum lima*
Symptoms: mild case	Within 30 min: tingling or numbness around lips, gradually spreading to face and neck; prickly sensation in fingertips and toes; headache, dizziness, nausea, vomiting, diarrhea	After 30 min to a few h (seldom more than 12 h): diarrhea, nausea, vomiting, abdominal pain	After 3-5 h: nausea, vomiting, diarrhea, abdominal cramps	Symptoms develop w/in 12-24 h of ingestion: gastrointestinal symptoms; diarrhea, abdominal pain, nausea, vomiting
Symptoms: extreme case	Muscular paralysis; pronounced respiratory difficulty; choking sensation; death through respiratory paralysis may occur w/in 2-24 h of ingestion	Chronic exposure may promote tumor formation in the digestive system	Decreased reaction to deep pain; dizziness, hallucinations, confusion; short-term memory loss; seizures	Neurological symptoms: numbness and tingling of hands and feet; cold objects feel hot to touch; difficulty balancing; low heart rate and blood pressure; rashes. In extreme cases death through respiratory failure
Treatment	Patient has stomach pumped and is given artificial respiration. No lasting effects.	Recovery after 3 d, irrespective of medical treatment	n/a	No antitoxin or specific treatment is available. Neu-rological symptoms may last for months and years. Calcium and mannitol may help relieve symptoms

Source: Hallegraeff (1993).

It should be noted that this link can be broken down to incorporate exposure: Toxins → Exposure to Toxins → Health. The types of health effects resulting from HAB toxins, as described by Hallegraeff (1993), occur only when a human is exposed to the toxin. Therefore, the connection between toxin exposure and health is clear, but the connection between toxin production and any change in human exposure is not, and the latter connection will be highly site specific. However, because some HAB toxins can be transmitted through the air and because estuaries are often locations of high population density, there is high likelihood of exposure to these toxins when they are produced.

Other Factors
Acute Toxicity vs. Long-term Health Impacts: All the HAB toxins listed in Table 3 have acute toxicity symptoms; however, some toxins have long-term health impacts as well that can be exacerbated by multiple exposures. For more detail, see Appendix 2 of the National Science and Technology Council Report (2016).

Pre-existing Conditions: People with respiratory issues such as asthma may have more severe reactions to airborne toxins (NOAA NOS 2017). For example, it has been found that healthy people exposed to aerosolized brevetoxins from Florida red tides find relief of symptoms once they leave the beach. However, people with asthma show more acute and long-term pulmonary symptoms, and they report respiratory symptoms days after exposure at a beach (Moore et al. 2008).

Toxin Concentrations: HAB-causing species can occur at different densities, resulting in varying toxin concentrations (Sellner et al. 2003). Additionally, symptoms of HAB toxins appear at varying toxin concentrations, depending on the

potency of the toxin. More detailed information on the toxin of interest and the concentration at which it will produce health impacts in humans will be helpful.

Mitigation Actions: There are many mitigation actions in place to prevent human exposure to HAB toxins when harmful algae blooms occur. Beach closures prevent visitors from developing respiratory symptoms, and fishery closures prevent the sale and consumption of contaminated seafood. It is when these mitigation actions fail or when harmful algae blooms are not properly detected that human health is negatively affected by HAB toxins.

Strength of Evidence
Fair. It is very well documented that toxins produced by harmful algae blooms will affect human health when people are exposed to the toxins (see symptoms in Table 3). However, exposure to toxins must be taken into account, and the likelihood of exposure will be site specific.

Predictability: Site-specific information on the toxin type and the local population will be essential for predicting the health outcomes of algal toxins.

Example. Predictions of how many people will be affected will depend on the type of toxin, toxin exposure route, and exposure level of people to the toxin. Site-specific information on the proximity of people to the toxin, vulnerable populations, and other factors will be important. Site-specific studies and data such as those collected in a study on *K. Brevis* toxins (brevetoxins) and emergency room visits related to respiratory issues can be performed to establish a relationship between toxins and health outcomes (Hoagland et al. 2009).

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1p: Toxins → Commercial Fishing
Description of Relationship
HAB toxins can contaminate fish and shellfish. These toxins can cause harmful symptoms and even death for humans when ingested, so when these toxins are detected in seafood or water, commercial fisheries are often shut down.

Summary of Evidence
The National Science and Technology Council Subcommittee on Ocean Science and Technology released a report on harmful algae blooms. Appendixes 1 and 2 of the report provide a summary of the toxins produced during different kinds of blooms, and those that impact commercial fishing are provided in Table 6.

Table 6. Harmful algal taxa and effects on fisheries

Hab taxa	Toxin/bioactive compound	Fishery closure reason	Impacted areas in the United States
Pseudo-nitzschia	Domoic acid	Amnesic shellfish poisoning → Shellfish harvesting closure	West Coast, Florida, Maine
Dinophysis; Prorocentrum	Okadaic acid, dinophysotoxins	Diarrhetic shellfish poisoning → Shellfish fishery closure	Oregon, Texas, Washington
Gambierdiscus; Fukuyoa	Ciguatoxins	Ciguatera fish poisoning → Bans on fish sales from affected areas	Florida, Gulf Coast, Hawaii, Pacific, Caribbean
Karenia	Brevetoxins	Neurotoxic shellfish poisoning → Shellfish fishery closure	Gulf of Mexico, Atlantic coast up to North Carolina
Alexandrium; Gymnodinium; Pyrodinium bahamense	Saxitoxins	Paralytic shellfish poisoning → Shellfish fishery closure	Pacific coast (including Alaska), northeast Atlantic coast, Florida
Prorocentrum minimum—Mahogany Tides	Not characterized	Mortality of spat in shellfish hatcheries → Lost shellfish	Chesapeake Bay

Source: National Science and Technology Council (2016).

A review of studies that examined commercial fishery effects resulting from harmful algae blooms from 1987 to 1992 showed annual costs across the United States ranging from $7 million to $19 million (measured in 2000 USD) (Hoagland et al. 2002, see Table 3). The effects measure impacts such as harvest losses, reduced sales, and farmed fish kills.

Other Factors
Depending on the HAB type and severity, fishery closure lengths may differ. The length of a closure will in part determine the severity of the impact on a fishery.

Strength of Evidence
Fair. Toxin detection will almost always result in temporary closure of relevant commercial fisheries. The specific impacts of closures will depend on HAB type, length, and extent; however, the impact on fisheries is reasonably certain. Estimating specific outcomes will be determined by the site and the species that are commercially harvested.

Predictability: Site-specific information on the toxin type and the local fish species will be essential for predicting the commercial fishing impacts of algal toxins.

Example. Site-specific studies and local data are needed to make accurate estimates of how HAB toxins will affect a local fishery. A review of such studies can be found in Hoagland et al. (2002); these studies examine outcomes such as temporary or permanent fishery closures, harvest losses, reduced sales, fish kills, and seafood recalls.

Sources
Hoagland, P., D.M. Anderson, Y. Kaoru, and A.W. White. 2002. "The Economic Effects of Harmful Algal Blooms in the United States: Estimates, Assessment Issues, and Information Needs." *Estuaries and Coasts* 25 (4): 819–837.
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1q: Toxins → Recreation (Beach Closures, Fish and Shellfish Harvest)
Description of Relationship
HAB toxins can negatively affect a variety of recreational opportunities, mainly because certain activities will be restricted during a harmful algae bloom.

Summary of Evidence

Karenia brevis, the organism that produces a red tide, is a common source of beach closures in the Gulf of Mexico (NOAA NOS 2017a). The toxin produced by *K. brevis* can become airborne and cause eye irritation and respiratory issues. This toxin also causes water discoloration, making water-based recreation unappealing (NOAA NOS 2017b).

Any HAB toxin that results in fishery closure will also negatively affect recreational fishing and shellfish harvesting. The National Science and Technology Council Subcommittee on Ocean Science and Technology released a report on harmful algae blooms. Appendixes 1 and 2 of the report provide a summary of the toxins produced during different kinds of blooms, and those that affect fisheries are provided in Table 7.

Table 7. Harmful algal taxa and effects on fisheries

HAB Taxa	Toxin/ Bioactive Compound	Fishery Closure reason	Impacted areas in the United States
Pseudo-nitzschia	Domoic Acid	Amnesic shellfish poisoning → Shellfish harvesting closure	West Coast, Florida, Maine
Dinophysis; Prorocentrum	Okadaic acid, dinophysotoxins	Diarrhetic shellfish poisoning→ Shellfish fishery closure	Oregon, Texas, Washington
Gambierdiscus; Prorocentrum; Ostreopsis	Ciguatoxins	Ciguatera fish poisoning → Bans on fish sales from affected areas	Florida, Gulf Coast, Hawaii, Pacific, Caribbean
Karenia	Brevetoxins	Neurotoxic shellfish poisoning → Shellfish fishery closure	Gulf of Mexico, Atlantic coast up to North Carolina
Alexandrium; Gymnodinium; Pyrodinium bahamense	Saxitoxins	Paralytic shellfish poisoning → Shellfish fishery closure	Pacific coast (incl. Alaska), northeast Atlantic coast, Florida
Prorocentrum minimum—Mahogany Tides	Not characterized	Mortality of spat in shellfish hatcheries → Lost shellfish	Chesapeake Bay

Source: National Science and Technology Council (2016).

Other Factors

Depending on the type of monitoring and local regulations, certain harmful algae blooms may or may not result in recreational restrictions. It will depend on whether blooms are detected, whether the public knows and understand the restrictions, and whether there are ways to enforce those restrictions. If restrictions are not made or are ignored, human health issues may arise due to HAB toxin exposure (see link 1o for more detail on health impacts).

Strength of Evidence

Fair. When HAB toxins are detected, relevant recreational opportunities exist, and local regulation and enforcement are in place, it is almost certain that recreational activities will be affected by the presence of HAB toxins.

Predictability: Site-specific information on the toxin type, the local population, and recreational opportunities will be essential for predicting the recreation impacts of algal toxins.

Example. Site-specific studies and local data are needed to make accurate estimates of how HAB toxins will affect recreation. There are few studies examining this type of data, but a review of such studies can be found in Hoagland et al. (2002); these studies examine outcomes such as recreational fishery closures (and the number of people participating in those fisheries) and reduced tourism numbers.

Sources

Hoagland, P., D.M. Anderson, Y. Kaoru, and A.W. White. 2002. "The Economic Effects of Harmful Algal Blooms in the United States: Estimates, Assessment Issues, and Information Needs." *Estuaries and Coasts* 25 (4): 819–837.

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NOAA National Ocean Service. 2017a. "What Are Beach Advisories and Beach Closures?" National Ocean Service. https://oceanservice.noaa.gov/facts/beach-closures.html.

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1r: Toxins → Wildlife (Marine and Terrestrial)

Description of Relationship

HAB toxins can be damaging to wildlife, and a wide variety of symptoms in different species have been seen, including mortality.

Summary of Evidence

The National Science and Technology Council Subcommittee on Ocean Science and Technology released a report on harmful algae blooms. Appendixes 1 and 2 of the report provide a summary of the toxins produced during different kinds of blooms, and those that affect wildlife species are provided in Table 8.

Table 8. Harmful algal taxa and effects on animal species

HAB Taxa	Toxin/bioactive compound	Animal effects	Impacted areas in the United States
Pseudo-nitzschia	Domoic acid	Sea bird and marine mammal mortality	West Coast, Florida, Maine
Gambierdiscus; Prorocentrum; Ostreopsis	Ciguatoxins	Possible marine mammal illness	Florida, Gulf Coast, Hawaii, Pacific, Caribbean
Karenia	Brevetoxins	Fish kills, manatee, dolphin, marine turtle, and bird deaths	Gulf of Mexico, Atlantic coast up to North Carolina
Alexandrium; Gymnodinium; Pyrodinium bahamense	Saxitoxins	Marine mammal deaths	Pacific coast (incl. Alaska), northeast Atlantic coast, Florida
Karlodinium	Karlotoxins	Fish kills	Atlantic and Gulf coasts
Aureococcus anophagefferens—Long Island Brown Tide	Not characterized	Shellfish die-offs	Mid-Atlantic coast
Akashiwo sanguineum	Surfactants	Migratory bird deaths	Pacific coast
Heterosigma akashiwo	Ichthyotoxins	Fish kills	Washington, Mid-Atlantic coast
Other Raphidophytes: Chattonella, Fibrocapsa	Brevetoxins; Ichthyotoxins	Fish kills	Mid-Atlantic coast
Alexandrium monilaturm	Goniodomin	Fish and shellfish mortality	Gulf of Mexico and Atlantic coast up to New Jersey
Cochlodinium	Not characterized	Fish kills	West Coast, Mid-Atlantic
Macroalgae	H2S, dopamine	Impair nesting protected species	All coasts

Source: National Science and Technology Council (2016).

It should be noted that this link can be broken down to incorporate exposure: Toxins → Exposure to Toxins → Wildlife. The National Science and Technology council report clearly describes the types of effects that wildlife can experience resulting from HAB toxins; for those effects to occur, wildlife must be exposed to the toxin. Therefore, the connection

between toxin exposure and wildlife is clear, but the connection between toxin production and any change in wildlife exposure is not, and that latter connection will be highly site specific.

Other Factors
HAB coverage, length of time, and concentration will determine the extent and severity of impacts on wildlife species. It is best to have studies specific to the toxin and wildlife species of interest.

Strength of Evidence
Low. The impacts of HAB toxins on wildlife have been studied, and multiple reports of toxin impacts on wildlife leave little doubt that there is a link between toxins and wildlife. The extent to which toxins will affect wildlife in a specific area will depend on the type, extent, and concentration of toxins resulting from the HAB bloom as well as on which susceptible wildlife species are present at the time of the bloom.

Predictability: Site-specific information on the toxin type and the local wildlife will be essential for predicting impacts of algal toxins on specific wildlife species. It is often difficult to predict HAB outcomes on a wildlife population, but local data on wildlife effects of previous blooms could provide a basis for estimating future impacts.

Example. One study details the connection between a *Pseudo-nitzschia australis* bloom in 1998 and the deaths of more than 400 California sea lions; however, the study notes that "establishing an unambiguous connection between HABs and marine mammal mortality is difficult" (Scholin et al. 2000).

Sources
National Science and Technology Council. 2016. "Harmful Algal Blooms and Hypoxia: Comprehensive Research Plan and Action Strategy." https://coastalscience.noaa.gov/research/docs/FINAL_HABs%20Hypoxia%20Research%20Plan%20and%20Action.pdf.

Scholin, Christopher A., Frances Gulland, Gregory J. Doucette, Scott Benson, et al. 2000. "Mortality of Sea Lions along the Central California Coast Linked to a Toxic Diatom Bloom." *Nature* 403 (6765): 80.

1s: Change in Salt Marsh Quantity or Quality → Chemical Contaminant Accumulation
Description of Relationship
Salt marsh habitat retains chemical contaminants (such as heavy metals, pesticides, and petroleum hydrocarbons) from water draining into an estuary.

Summary of Evidence
Salt marshes are known to act as sinks for contaminants such as heavy metals; however, the rates of accumulation and concentration in salt marsh soils and plants varies widely. Gedan et al. (2009) report that "in anoxic marine soils, free metal ions are precipitated as metal sulfides of low solubility, making deeper sediments of salt marshes stable repositories for pollutants in the absence of bioturbation or oxidation of soils." A review of heavy metal accumulation in salt marshes reveals that concentrations of metal in salt marsh sediments depend on the contaminant source, the distance to that source, and the biophysical composition of the marsh (Williams et al. 1994). See Table 9 for a summary of heavy metal concentrations in salt marsh sediments and salt marsh plants contained in Williams et al. (1994).

Table 9. Range of heavy metal concentrations in salt marsh sediments and plants, from a 1994 review article

	Concentration range	
	In sediments (µg/g)	In plants (µg/g)
Cd	0.13-8.5	0.1-5
Co	8-13	No data
Cr	27-1070	1.5-11.9
Cu	6.2-190	2.2-42
Fe	No data	37-2830
Hg	0.05-3.5	0.01-0.1
Ni	7.9-542	No data
Pb	10.4-282	0.3-40
Zn	13.6-715	11-300
Mn	79-1604	No data

Source: Tables 1 and 2 in Williams et al. (1994).

Because the figures in Table 7 are concentrations, not accumulation rates, they do not provide information about how much of the contaminants are being removed over time or what percentage of input heavy metals are trapped by the marsh. One study on a *Spartina altiflora* salt marsh in China found that although salt marsh plants accumulate and remove heavy metals (Cr, Pb, Cu, Zn, and Mn) from the soil, annual plant death and litter decomposition re-release some of those metals back into the sediment. However, net accumulation of metals was greater than net loss of metals from the plants (Lian et al. 2017). Leendertse et al. (1996) had similar findings—they found that sediments and vegetation in experimental salt marshes absorbed and transformed 30–65% of deposited metals and concluded that salt marshes represent metal sinks, except during events causing high erosion. Other chemical contaminants such as polychlorinated biphyenyl (PCB) (a now-banned substance that was widely used in coolant fluids), petroleum hydrocarbons (from oil spills), and organochlorine compounds (used in pesticides) have also been found to accumulate in salt marsh sediments (Scrimshaw et al. 1996; Reddy et al. 2002; Barra et al. 2004). Though multiple studies provide measurements of concentrations of these contaminants in salt marsh sediments, these measurements do not indicate how accumulation in the marsh relates to the total amount of the contaminants that are imported to or exported from the marsh.

Tool: The OpenNSPECT tool (Nonpoint Source Pollution and Erosion Comparison Tool), which is available through NOAA's digital coast website (https://coast.noaa.gov/digitalcoast/), can aid examinations of water quality impacts from development and land use changes. The model could be used to indicate how additions of salt marsh habitat affect water quality. Data requirements include C-CAP coastal land cover, elevation, soils, precipitation, R-factor (rainfall factor), and pollutant coefficients. The outputs of the OpenNSPECT tool include runoff volume, accumulated pollutants, pollutant concentrations, and pollutant comparisons to water quality standards. (Find the tool here: https://coast.noaa.gov/digitalcoast/tools/opennspect.)

Online, three-hour trainings are available for the OpenNSPECT tool. Use this site to register: https://coast.noaa.gov/digitalcoast/training/opennspect.html.

Tool: Multiple reviewers mentioned the possible use of the SWAT or SPARROW models for application here; however, these models are usually applied at a watershed scale, and they are often used for modeling nutrients, sediments, or hydrological flows rather than pollutants. No studies could be found that apply these models in a scenario that would predict pollutant accumulation in a particular habitat.

Other Factors

Marsh Biochemical and Physical Structure: A review of heavy metal accumulation in salt marsh soils reveals that concentrations of metal in salt marsh sediments depend on the marshes' biochemical features, including soil composition, particle size, marsh morphology, water circulation, flooding frequency, vegetation cover and type, and chemical conditions of estuarine waters and the marsh soil (Williams et al. 1994). Accumulation of heavy metals in salt marsh plants depends on salinity, soil temperature, growth characteristics of plants during different life stages, soil particle size, organic matter content of the soil, soil pH, cation exchange capacity, and soil moisture (Lian et al. 2017).

Sediment and Pollutant Sources: A marsh will accumulate pollutants only if those pollutants are being deposited in the marsh. Sediment sources may be important to consider and will vary depending on hydrogeomorphic setting. Whether a marsh is erosional or depositional may affect the likelihood that pollutants will accumulate. Legacy issues associated with land use are also important—if heavy industry was present along a river that can transport the metals to the coast, pollutants might be more important to consider.

Strength of Evidence

Fair. Though there are multiple documented cases of salt marshes accumulating chemical contaminants, there are also many factors that can alter how much of or whether these contaminants accumulate, most notably chemical and hydrological conditions.

Predictability: Site-specific data are required to accurately predict accumulation of chemicals by a salt marsh. Models that aid in these predictions do exist, though they incorporate many variables. See an example modeling tool below.

Example. The OpenNSPECT tool is used in the NOAA community and incorporates many of the considerations important for modeling erosion, runoff, sediments, and chemical pollutants. Although documented and applied in multiple watersheds and at multiple sites around the United States, the tool's use could benefit from further testing (see the NOAA digital coast OpenNSPECT page for case studies). Model inputs do require site-specific data, including land cover, elevation, soils, precipitation, R-factor, and local pollution coefficients (if possible). Importantly, the model uses general pollution export coefficients (PECs) as a default to model the relationship between land cover/land use and pollutant measures. Though generic, PECs are built into the model, and PECs derived specifically for a given area of interest will provide model outputs better calibrated to the site (Schenk et al. n.d.).

Sources

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Williams, T.P., J.M. Bubb, and J.N. Lester. 1994. "Metal Accumulation within Salt Marsh Environments: A Review." *Marine Pollution Bulletin* 28 (5): 277–290.

1t: Chemical Contaminant Accumulation → Health

Description of Relationship

Chemical contaminants accumulating in the marsh could affect human health by changing human exposure to those contaminants. Human exposures to chemical contaminants that accumulate in a marsh environment could change in multiple ways. Chemical accumulation in a salt marsh removes certain contaminants from entering estuarine waters, possibly reducing human exposure in the estuary. However, accumulation of chemical contaminants in marsh soils could also potentially increase human exposure within the marsh.

Summary of Evidence

No evidence was found linking chemical accumulation in a salt marsh directly with human health outcomes. However, as the salt marsh accumulates various contaminants, it is possible that human exposure to those contaminants could change.

Toxicity related to chemical contaminants that are known to accumulate in salt marsh habitats will depend on dose, exposure route, chemical species (which chemical form the contaminant is in), and demographics of exposed populations (age, gender, genetics, nutritional status, and so on) (Tchounwou et al. 2012). Chemical contaminants that accumulate in marshes include heavy metals (Williams et al. 1994); a book chapter titled "Heavy Metal Toxicity and the Environment" by Tchounwou et al. (2012) provides a good foundation for information related to environmental exposures and health outcomes related to heavy metal toxicity. The heavy metals considered most toxic (those considered to pose the highest risk to human health) are arsenic, cadmium, chromium, lead, and mercury. All five of these metals are considered systemic toxicants, linked to multiple organ damage, and classified as carcinogens (Tchounwou et al. 2012). Table 10 summarizes information from Tchounwou et al. (2012).

Table 10. Heavy metal sources, exposure routes, and human health effects

Heavy metal	Natural levels	Common anthropogenic sources	Exposure route	Human health effects
Arsenic (As)	Air: 1-3 ng/m³ (rural), 20-100 ng/m³ (cities) Water: <10µg/L Food: 20-140 ng/kg Soil: 1-40 mg/kg	Industrial production, agricultural products (pesticides, herbicides, fungicides, algicides, sheep dips), pharmaceuticals	Ingestion, inhalation, dermal contact	Cardiovascular disease, developmental anomalies, neurologic disorder, diabetes, hearing loss, hematologic disorders, cancers
Cadmium (Cd)	Soil: 0.1 mg/kg	Industrial activities, batteries	Inhalation (often cigarette smoke), ingestion	Pulmonary function, olfactory function, osteoporosis, acute gastrointestinal symptoms, cancers
Chromium (Cr)	Air: 1-100 ng/cm³ Seawater: 5-800 µg/L Freshwater: 26 µg/L – 5.2mg/L Soil: 1-3000 mg/ kg Food: <10-1300 µg/kg	Industrial production, tanneries	Ingestion, inhalation, dermal contact	Renal damage, asthma, respiratory tract cancers, stomach ulcers, other cancers, stomach tumors, death (with high dose ingestion)
Lead (Pb)		Fossil fuel burning, industrial production, batteries, ammunition, metal pipes, paint	Inhalation, ingestion	Nervous system damage, kidney damage, liver damage, endocrine damage, reproductive system damage, diminished intelligence
Mercury (Hg)		Electrical industry, industrial production, nuclear reactors, antifungals, pharmaceutical preservative, dental practices	Ingestion (especially from fish and shellfish)	Gastrointestinal toxicity, neurotoxicity, nephrotoxicity

Source: Tchounwou et al. (2012).

Other pollutants accumulating in marshes include petroleum hydrocarbons and pesticides. Humans can be exposed to petroleum hydrocarbons through inhalation, ingestion, or dermal contact (ATSDR 1999). Effects of petroleum hydrocarbons on human health vary with the chemical species and exposure type and length, but they can include nausea, irritation, immune system impairment, or temporary or permanent impacts on the central nervous system (ATSDR 1999). Pesticide exposure can result in effects on the nervous system or endocrine system, skin or eye irritation, or development of certain cancers (EPA 2017).

This link can be broken down to incorporate exposure: Chemical Contaminant Accumulation → Exposure to Chemicals → Health. Sources cited here summarize the known health effects related to certain pollutant toxicities, but for those health effects to occur, a human must be exposed to the chemical pollutant. Therefore, the connection between exposure and health is clear, but the connection between pollutant accumulation in a marsh and any change in human exposure is not. Site-specific data may provide more information about exposure, but no example studies could be found to document data of this kind.

Other Factors
Exposure Type: Exposure dosage, route, and frequency can drastically change the type and severity of chemical contaminant toxicity.

Disturbance: A salt marsh may, over the long term, act as a sink for chemical contaminants, but when the plants or sediments in it are disturbed by storm events or anthropogenic activities (such as dredging), the marsh could act as a contaminant source area.

Strength of Evidence
None. Though there is clear proof that exposure to chemical contaminants such as heavy metals has a variety of human health effects, it is unclear whether accumulation of those contaminants in a salt marsh environment changes the likelihood and level of human exposure to those contaminants. No evidence was found to show that accumulation of chemical contaminants in a salt marsh changes the level of those contaminants in the surrounding estuary. No evidence was found to show that accumulation of chemical contaminants in a salt marsh changes the exposure rate of people to those contaminants while they are in the marsh.

Sources
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1u: Chemical Contaminants → Wildlife Populations
Description of Relationship
Chemical contaminants accumulated by the marsh could change the exposure risk of wildlife populations to those contaminants. Wildlife exposures to chemical contaminants that accumulate in a marsh environment could change in multiple ways. Chemical accumulation in a salt marsh removes certain contaminants from entering estuarine waters, possibly reducing wildlife exposure in the adjacent estuary. However, accumulation of chemical contaminants in marsh soils could also potentially increase wildlife exposure within the marsh.

Summary of Evidence

Little evidence was found directly linking chemical accumulation in a salt marsh directly with wildlife population outcomes. However, as the salt marsh accumulates various contaminants, it is possible that wildlife exposure to those contaminants may change. There has been concern that re-suspended metals and plant translocation of sediment-bound metals may introduce metals into marine food webs (Gedan et al. 2009).

Applicability of this link will depend on the toxins and wildlife present at a specific site. Heavy metals and other chemical pollutants have been shown to affect the health, survival, or reproduction of a variety of wildlife taxa including mammals (Das et al. 2003), crustaceans (Connor 1972), fish (Jezierska et al. 2009), and birds (Fry 1995). One study found that mercury and PCBs (both toxins that have been found to accumulate in salt marshes) changed the benthic invertebrate community structure in a salt marsh (Horne et al. 1999), and though not discussed in the paper, community structural changes could potentially have cascading impacts within the ecosystem.

It has been shown that salt marsh/estuarine sediment contaminant concentrations are often positively correlated with tissue concentrations of benthic invertebrates that live in/on the sediment (Table 5 of Bryan and Langston 1992; Horne et al. 1999), indicating introduction of these accumulated contaminants from the sediment into the marine food chain at the level of these benthic species. It could be assumed that these contaminants may pass to other organisms through the food chain.

This link can be broken down to incorporate exposure: Chemical Contaminant Accumulation → Exposure to Contaminants → Wildlife Populations. Multiple studies described above show different effects on wildlife populations resulting from chemical contaminants, but for those health effects to occur, wildlife must be exposed to the contaminants. Therefore, the connection between toxin exposure and wildlife is described here, but the connection between chemical contaminant accumulation in a marsh and any change in wildlife exposure is not as clear, and that latter connection will be highly site specific. The studies showing that benthic invertebrates often contain chemical contaminants present in salt marsh sediment (Bryan and Langston 1992; Horne et al. 1999) is the best evidence we have to show that wildlife exposure to contaminants may be altered by salt marsh accumulation of those contaminants.

Other Factors

Exposure dosage, route, and frequency can drastically change the type and severity of chemical contaminant toxicity.

Strength of Evidence

Low. It is clear that certain wildlife populations can be affected by chemical contaminants found in marine water, sediments, or both. However, there is little evidence to show the direct link between changes in accumulation of those contaminants by salt marshes and resulting changes in the exposure of wildlife to those contaminants. The high correlation between sediment contaminant concentrations and tissue contaminant concentrations of benthic organisms represents the most direct evidence found for this link. Although this evidence does show that contaminants found in sediments do appear in wildlife species, the impact of those contaminants in benthic organisms on wildlife populations is not clear.

Predictability: Predictability of this link is low because the outcomes will be site specific, depending on the chemicals and wildlife species present. Additionally, little evidence is available to connect these two nodes, making predictability difficult due to a lack of information.

Sources

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1v: Turbidity (Water Clarity) → Recreation (Swimming, Fishing)

Description of Relationship

Turbidity (water clarity) affects the likelihood that people will recreate in a body of water. With increased turbidity (decreased water clarity), people are less likely to consider a water body appropriate for recreation. Perceived swimming suitability in fresh water can be related to black disc visibility. Studies on U.S. lakes found that water clarity increases the number of visits to a lake, but that lake users were also willing to travel up farther for an increase in water clarity. In freshwater environments, it has also been found that turbidity may be linked to the number of trips that recreational anglers take.

Summary of Evidence

In multiple studies of freshwater lakes in New Zealand, the threshold for perceived swimming suitability was found to be 1.1m–2.2m black disc visibility. In a survey of freshwater experts utilizing the Delphi method, water was considered marginally suitable for swimming at 1.1m black disc visibility and suitable at 1.6m black disc visibility (Smith and Davies Colley 1992). In studies that used public surveys at various lake sites, it was found that for 75%–80% of survey participants to consider a lake suitable for swimming, the visibility had to be 1.2m black disc visibility (corresponding to a Secchi depth of 1.5m), and for 90% of survey participants to consider a lake suitable for swimming, the visibility had to be 2.2m black disc visibility (corresponding to 2.75m Secchi depth) (Smith et al. 1991; Smith et al. 1995). These studies note that the critical region to examine at a finer scale should be a black disc depth of 0.9m–1.6m (Smith et al. 1995). In a study in Finland, it was found that with increased close-to-home water clarity, both frequency of swimming and fishing would increase, as would the number of fishers; however, water clarity had no apparent impact on boating frequency (Vesterinen et al. 2009). A study of Minnesota and Iowa lakes found that water clarity not only increases the number of visits to a lake, but also that lake users were also willing to travel 56 minutes farther for a 1m increase in water depth clarity (Keeler et al. 2015). However, this study did not distinguish types of recreational use, only visits to a lake. Notably, no studies were found that exclusively examine the relationship between water clarity and recreation in salt water bodies, and it is unclear how transferrable results from freshwater are to estuarine environments.

Little evidence could be found to link turbidity to other forms of estuarine recreational activities. Activities such as boating and fishing may be more relevant than swimming in the estuarine environment, but little information could be found to describe those connections. One study on recreational angling found a connection between turbidity and fishing demand; however, the data used for the study was for freshwater sites (Englin et al. 1996). The study found that increases in turbidity led to slight decreases in total seasonal consumer surplus of anglers. Consumer surplus was not very responsive to changes in turbidity; turbidity had a small influence on the total number of trips that anglers took. According to the models, a 50% increase in turbidity (a *very* large change) will only reduce total consumer surplus of individual anglers by $8 per season (Englin et al. 1996).

If snorkeling, scuba diving, or both are relevant recreational activities for a site of interest, they will most likely be affected by changes in turbidity because they are heavily dependent on visibility and water clarity. However, no studies were found documenting the relationship between turbidity and these recreational activities.

Other Factors

Surrounding Area: The area surrounding a water body can influence how suitable a person believes it to be for recreational purposes (i.e., is the surrounding area natural, developed, clear, forested, and so on) (Smith et al. 1991).

Survey Respondent Bias: In survey work, there are always factors that can influence a person's response. For recreational surveys linking water clarity to suitability for recreation, studies mention that ethnicity, age, and socio-economic factors may affect a respondent's degree of awareness about water pollution and other water quality issues. Additionally, people may base their opinions on hearsay or media reports (Smith et al. 1991).

Other Features: Perhaps water clarity is not the major factor in a swimmer's decision to use a certain body of water. For example, one study found that clarity in fact did not play an important role; instead, it hypothesized that other factors such as bottom substrate, swimming facility availability, or clean beaches might be more important determinants (Scribner et al. 2004).

Strength of Evidence

Low. Though there is evidence linking turbidity/water clarity to recreational use, no studies were found that examined this relationship in salt water environments (thus this link is based on extrapolations). It is very possible that people have different considerations and perceptions when recreating in fresh versus salt water.

Predictability: Predictability of this link is low because the available evidence does not rank highly for applicability.

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Links 2a-i

2a: Change in Salt Marsh Quantity or Quality → Wildlife Populations

Description of Relationship

Salt marsh habitat provides resources to wildlife species whose populations are promoted or protected by the presence of the marsh. Salt marsh habitat provides nursery grounds for fish species, increasing fish stocks and supporting multiple levels of the ocean food chain (NOAA OCM, NH Department of Environmental Services Coastal Program, and Eastern Research Group 2016). Salt marsh habitat can be linked to increases in nekton production. Salt marsh-dependent bird species' presence or abundance can be related to marsh size, habitat quality, or both.

Summary of Evidence

There is little generalizable support for this link, other than to say that salt marsh provides habitat and resources to many different wildlife species. The specific relationship between salt marsh and a wildlife species will depend completely on the marsh site and the species of interest. At the most basic level, if the density of a species is known for similar marshes (number of individuals per unit area), a very basic estimate of the number of individuals supported by a new marsh site could be estimated. However, more data and information are required for detailed estimates of this linkage. Two example links and evidence are provided below for reference: (1) Salt Marsh → Nekton and (2) Salt Marsh → Birds.

Example: Salt Marsh → Nekton: Using a trophic transfer approach, it is possible to estimate the biomass of nekton resulting from the addition of a certain number of acres of salt marsh habitat (McCay et al. 2003; Kneib 2003). Nekton represents free-swimming organisms whose movements are independent from the current, the tide, or both. These organisms include both fish and crustaceans (NOAA NMFS 2006). Calculation of trophic transfer requires estimation of the primary productivity of a specific ecosystem type and of the productivity lost at each trophic level. To perform this calculation, data on dry weight of primary production are needed (primary productivity in salt marshes comes from

two sources: primary productivity of marsh grasses, or grams of dry weight m^{-2} yr^{-1}, and, benthic microalgal production, or grams of dry weight m^{-2} yr^{-1}). A conversion between dry weight and wet weight is also needed (dry weight is often considered 22% of wet weight) (Kneib 2003). Kneib (2003) calculates the percentage of primary production remaining at the trophic level of estuarine resident nekton to be 0.22% of original primary production dry weight. Additional trophic transfer calculations would need to be performed to make calculations for higher trophic levels.

Therefore, a simple formula for calculating a rough estimate of wet weight of nekton produced by an additional m^2 of salt marsh habitat is

$$\text{WW nekton (g m}^{-2}\text{ yr}^{-1}\text{)} = \text{Primary productivity dry weight (g m}^{-2}\text{ yr}^{-1}\text{)}*0.0022*4.55$$
[Because wet weight of nekton equals 0.22% of the primary production of dry weight, and dry weight is 22% of wet weight, requiring a multiplier of 4.55]
Where primary productivity dry weight = dry weight productivity of marsh grass + dry weight productivity of benthic microalgae

Conversions can be made to determine the wet weight of nekton per acre of marsh per year (kg ac^{-1} yr^{-1}) by using the following formula:

$$\text{WW (kg ac}^{-1}\text{ yr}^{-1}\text{)} = \text{(WW (g m}^{-2}\text{ yr}^{-1}\text{)}/0.000247)/1000$$
[Because 1m^2=0.000247 acres, and 1000g = 1kg]

Because energy transfers between trophic levels in a salt marsh are complex, some might prefer to do a site-specific study similar to Minello et al. (2008) to develop wildlife production rates specific to a site of interest. This study used distribution patterns, size frequencies, size-weight relationships, and growth rates to estimate production of multiple nekton species on salt marsh habitat in Galveston Bay, Texas. The study estimated that for each hectare of salt marsh, 128 kg of brown shrimp, 109 kg of white shrimp, and 170 kg of blue crabs were produced (Minello et al. 2008).

Example: Salt Marsh → Birds: No reviews or generalizable information about salt marsh habitat impact on bird populations could be found, but individual studies do indicate that marsh habitat is important for certain bird species. Local studies could be performed for species of interest to generate data similar to those provided in the following studies. A study of estuarine marsh birds in the Connecticut River Estuary found that breeding bird species richness increased with marsh area (Craig and Beal 1992). A study of the endangered tidal marsh obligate species the California Clapper Rail in the San Francisco Bay area found that Clapper Rail density increases with marsh size, up to 100 hectares, after which bird density does not increase (Liu et al. 2012). The same study found that marsh quality was as important, or more important, than the quantity of marsh habitat (i.e., restoring degraded marshes will have a large impact on Clapper Rail density) (Liu et al. 2012). Another study on tidal marsh-dependent bird species in San Francisco Bay found that Tidal Marsh Song Sparrow abundance was positively correlated with marsh patch size, and the Marsh Wren presence was positively correlated with the percent of marsh habitat within a 50m radius (Spautz et al. 2006). In all of these studies, there were multiple other environmental and landscape variables included in the models that predicted abundance, habitat suitability, or both (i.e., marsh size and habitat quality were not the only variables included in the final models). For those looking to perform studies that relate bird populations to marsh habitat features, Liu et al. (2012) provide many useful details about methods and model types.

Other Factors Related to Nekton Example
Region: Primary production by salt marshes will vary among different parts of the country. In the southeastern United States, the range of primary productivity of marsh grasses is 130–3700 g dw m^2 and the range of benthic microalgae productivity is 140–470 g dw m^{-2} (Kneib 2003). One study documenting primary production in Rhode Island found that salt marsh grass primary production was 450 g dw m^{-2} and microalgae primary production was 106 g dw m^{-2}.

Salinity: Trophic linkages can vary with salinity. One study of the Delaware Bay found significant variation in salt marsh trophic linkages along a salinity gradient (Litvin and Weinstein 2003).

Life History Strategy: Life history traits of individual nekton species will also determine how they utilize the primary production provided by the salt marsh and therefore the rate of transfer between different trophic levels (Litvin and Weinstein 2003).

Other Factors Related to Bird Example

This link will be strongest for bird species that are salt marsh dependent for either breeding or feeding. Birds that are habitat generalists likely will not show strong associations with salt marsh habitats.

Strength of Evidence

Moderate. It is clear that salt marsh provides important habitat for many wildlife species. Little general evidence is available to support this link, but numerous sources (including experts who reviewed this document) cite the importance of salt marsh habitat for maintenance of biodiversity and wildlife species. This relationship will be entirely dependent on the marsh site and the wildlife species of interest.

Predictability: Predictability is low for this link because the link will entirely depend on the site and local wildlife species. Relationships between salt marsh habitat and specific species or groups of species can be predicted, but predicting these relationships will still require site-specific data (see below).

Evidence Strength for Nekton Example

Moderate. It is widely acknowledged that salt marsh habitat provides sustenance and protection for many marine species; however, the mathematical relationship between the salt marsh and nekton (or higher trophic level) production is site specific and requires data unique to the site of interest. Measurements of primary production at a marsh site are needed, along with estimates of the rate of trophic transfer. Unknown is whether the one trophic transfer rate presented here (Kneib 2003) is accurate for all marshes.

Example. The Minello et al. (2008) study provides an example of the type of research that could be done to use models to predict the amount of nekton production provided by salt marsh area. Doing a study such as this one would be most useful when a marsh site produces a few key nekton species of interest.

Evidence Strength for Bird Example

Example. This link is completely dependent on species of interest at the site. Local studies on bird presence, densities, or breeding behavior will be needed to make connections between salt marsh habitat quantity or quality and bird density. There are many existing studies on birds and bird habitats, so it may be possible that a study linking any given species of interest to salt marsh habitat exists. Because habitat models tend to include many environmental variables, it is unlikely that salt marsh size or quality will completely explain or predict the presence of a certain bird species.

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2b: Wildlife Populations (Fish, Crustaceans, Shellfish) → Health (Nutrition)
Description of Relationship
Certain human communities depend on marine wildlife resources for subsistence and nutrition. When fish, shellfish, or other marine wildlife populations decline, human communities that depend on those resources for subsistence and nutrition will be less healthy.

Summary of Evidence
Some communities in the United States depend heavily on seafood for subsistence, nutrition, or both, and when these resources are depleted, community members' health can deteriorate. These fish-dependent communities are overwhelmingly communities of color, low-income communities, and communities of indigenous peoples (NEJAC 2001). This link will not be relevant everywhere, but it is important to acknowledge that it does exist in certain circumstances, and communities' health and nutrition cannot be discounted when making management decisions. Some of the sources listed below are good resources for learning more about the specific nutritional benefits that fish provide and about the dependence that certain communities have on these foods. Although some of these resources pertain to developing countries, they would be good resources to consult if you believe this link may be important for your conceptual model.

Other Factors
The strength and resilience of other food systems is a major factor in this link. In locations where crops regularly fail or where there is no alternative source of protein (or affordable protein), the impacts of reduced fishing on human health will be more pronounced.

Strength of Evidence
Low. There is a lack of evidence directly linking wildlife populations to individual and community nutritional health in the United States. Because U.S. communities often have access to food from a wide variety of sources, it is unclear whether nutrition or health would suffer due to decreased marine wildlife populations, and no studies were found that directly studied this link. This link will likely not exist in many locations, but it may be essential in specific communities where fishing is highly important. This link is more studied in developing countries where it has been found that in certain communities, fish products play a critical role in food security and nutrition (FAO 2014; Golden et al. 2016).

Predictability: Predictability is low for this link due to a lack of evidence, possible mediating factors (such as other food sources), and the importance of site-specific information.

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2c: Wildlife Populations→ Species Persistence
Description of Relationship
Reduced wildlife population size decreases a population's viability. Decreasing the population size by one individual reduces the population's long-term viability (probability of persistence) by X%.

Summary of Evidence
A reduction in the size of a wildlife population can influence the population's long-term viability in several ways. Population size thresholds represent a minimum viable size for a population of a given species to persist; if a population

falls below that threshold, it will go extinct (Traill et al. 2007). There are multiple reasons for the existence of population size thresholds. Demographic stochasticity (the probabilistic nature of reproduction and death) causes population size fluctuations that average out in large populations but that can cause extinction in small populations. Allee effects refer to the positive effects of higher population density on processes that lead to individual fitness (e.g., finding mates, social dynamics, predator-prey interactions) (Kramer et al. 2009). A decline in the population size brings the population closer to its minimum viable size and lowers the probability of long-term persistence (Traill et al. 2007).

Smaller populations also have reduced genetic diversity and inbreeding depression, which can decrease their probability of persistence (Frankham 2005). Loss of genetic diversity and inbreeding depression both depend on the effective population size (the number of adults that are actually breeding in the population) (Frankham 2005).

In laboratory studies, inbreeding depression has been shown to affect many aspects of reproduction and survival, decreasing overall fitness rates; subsequent research in captive and wild populations of wildlife species has shown that wildlife in natural habitats experience inbreeding depression (Frankham 2005). Few field studies have examined the effect of inbreeding depression on extinction risk for wild populations, but those that do exist have found a significant effect of inbreeding depression on extinction risk, and computer simulations of populations showed that the median time for extinction was reduced 25%–31% for populations with 50, 250, and 1,000 individuals, relative to populations with no inbreeding depression (Brook et al. 2002). A later study that estimated the levels of inbreeding depression in wild populations using a meta-analysis found much higher inbreeding depression levels than was assumed in the Brooks study. When population persistence was simulated using these results, it was found that the mean overall inbreeding effect seen in wild populations decreased the median time to extinction by 37% on average (O'Grady et al. 2006).

Lower genetic diversity limits the ability of the population to adapt to environmental change in the future through evolution. This effect takes place over a much longer time period than effects from inbreeding depression, and some studies have shown that inbreeding depression is likely a much stronger determinant of extinction risk than reduced genetic diversity (Frankham 2005).

Note: For the purposes of creating a conceptual model, you will likely want to focus efforts on studying the persistence of wildlife populations that people care about or that are especially threatened. When extending this link (2d) to a value (10c), it is key to examine a species that people care about so that it has an existence value. Marine mammals, birds, or iconic fish or shellfish are likely good candidates because people are highly invested in their persistence for current and future generations.

Other Factors
Each individual species and population is unique, and it will therefore react differently to reduced population size. Life-history strategy and mobility also affect the likelihood that a species will experience inbreeding depression. Studies specific to each species of interest are required to determine the impacts of reduced population sizes on that species' persistence.

Strength of Evidence
Low. A meta-analysis of minimum viable population studies estimated a mean minimum viable population for various taxa, but it found that minimum viable population is very specific to each individual population. There is also a meta-analysis that confirmed the effects of life-history strategy and population size relative to carrying capacity on a population's ability to compensate for increased anthropogenic mortality.

Predictability: With the requisite data, population viability analyses for marine species can be performed. (Note: these analyses are highly data intensive). Population viability analysis software can predict trends for local populations; these models have been shown to be relatively accurate (Brook et al. 2000), but they are dependent on many parameters that may be difficult to assess for some populations. A comparison of six population viability analysis models for the whooping crane found that the projected mean population size and extinction risk (after 50 years) varied among population viability analysis packages, mostly due to differences in package features (Brook et al. 1999). When the models were standardized to remove these differences (essentially, the more complex features were simplified to match features available in the simplest models), results across packages were much more similar. Because researchers generally want to be conservative in modeling rare and threatened species, they generally will want to use the full models that include more potential threats. It is not usually known which of the models will provide the most accurate prediction for the species in question, so there is a moderate degree of uncertainty associated with population viability analysis.

Example. Population viability studies have been completed for marine animals, including multiple species of salmon (Ratner et al. 1997; Legault 2005), marine mammals (Burkhart and Slooten 2003; Heinsohn et al. 2004; Winship and Trites 2006), and commercially fished species (Curtis and Vincent 2008).

Sources

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2d: Wildlife Populations → Commercial Fishing

Description of Relationship

Wildlife populations support the commercial fishing industry. An increase in wildlife populations will result in an increase in fish landings.

Summary of Evidence

This link will depend on the species or taxa being harvested commercially. Two example linkages and evidence are provided below for (1) Fish → Commercial Fishing and (2) Shellfish Aquaculture → Commercial Fishing.

Example: Fish → Commercial Fishing: It can be assumed that with an increase in fish populations, fish landings will also increase. The National Marine Fisheries Service and state jurisdictions set fishing policy, and with healthy fish stocks, increased commercial fishing will be allowed to occur, up to a point. Fish landings can be monitored using the NOAA National Marine Fisheries Service database (http://www.st.nmfs.noaa.gov/commercial-fisheries/commercial-landings/annual-landings/index).

The striped bass fishery in the Chesapeake Bay provides an example of fish population size leading to commercial fishing impacts. The striped bass population declined in the Chesapeake Bay during the 1980s and resulted in a moratorium on fishing in many adjacent states and large cutbacks in harvest in other states. Only when populations had rebounded in the mid 1990s were the fisheries reopened (Pendleton 2010, 66). Science-based management informs current fishing quotas, which are related to current bass population levels (NOAA 2017).

Example: Shellfish Aquaculture → *Commercial Fishing:* Shellfish aquaculture farms provide yearly yields that create revenue for aquaculturists. Shellfish harvest amounts depend on area available for aquaculture, number of aquaculture permits sold in an area, shellfish food supply, shellfish density, and other environmental parameters (water quality measures).

Tool: The Farm Aquaculture Resource Management Model (FARM) allows users to determine optimal carrying capacity for aquaculture shellfish farms. The model outputs the number of adult (harvestable) shellfish and adult shellfish biomass. Input variables include environmental parameters related to the food supply to the farm (current velocity, concentration of food sources, and so on), farm dimensions (area of sea bed available for farming), and shellfish stock (number of small animals at the beginning of cultivation). The model combines physical, biogeochemical, and bivalve growth models for determining shellfish production outputs (http://www.farmscale.org/). Although shellfish grown in aquaculture may not be considered "wildlife," they do represent a wildlife species that is commercially important.

Note: The FARM model is applicable only for aquaculture shellfish. Therefore, the tool should only be used if aquaculture sites are established in the salt marsh site of interest, or if the intervention to change the quantity or quality of salt marsh creates conditions conducive to new or enhanced aquaculture sites outside of or adjacent to the marsh. In short, the FARM model is applicable only if the intervention to restore/protect salt marsh somehow establishes new or enhanced aquaculture sites.

Other Factors
Relevant to the Fish Link: State, local, regional, and national fishing policies can determine how much fish (and which fish species) the commercial fishing industry can harvest.

Relevant to the Shellfish Link:
Shellfish Species: Species of shellfish being grown will be a factor in the yield that results from a particular farm. The FARM model includes various species of mussels, oysters, and clams. The model also allows for species combinations (polyculture) to be modeled.

Seasonal Variation: Shellfish will grow variably depending on how much food they are receiving from the water, which can vary seasonally depending on currents and nutrient regimes. The FARM model is being updated and will include functionalities to address these seasonal variations in food inputs.

Seed Density: The density and size of small animals at the beginning of cultivation (seed) will also be a factor in the timeline and output of a farm.

Strength of Evidence
Fair. General evidence for this link is fair because it will vary depending on the location, commercially harvested species, presence of commercial fishing operations, and local or national fishing regulations. Creating a link between wildlife and commercial fishing requires site-specific information that makes it difficult to generalize this link beyond saying that commercial fishing can only exist when wildlife populations are available to support it.

Predictability: Predictability is low due to the site-specific data requirements needed to understand this link. Predictability is also low due to the numerous external factors that influence the relationship. Predictability is increased when a specific species or type of fishery is identified.

Evidence Strength for the Fish Example:
Moderate. If fish stocks increase (and fishery policy allows it), commercial fishery catches will increase as well.

Evidence Strength for the Shellfish Aquaculture Example:

Moderate. The FARM model is widely used to estimate shellfish aquaculture outputs; however, data specific to the aquaculture farm is needed. But more generally, if aquaculture sites exist and shellfish are farmed, there will be a link to commercial fishery harvest levels.

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2a,d: Change in Salt Marsh Quantity or Quality → Commercial Fishing

(This link represents a combination of two links in the model: Change in salt marsh quantity or quality → wildlife populations → commercial fishing. The reason is that there is a model that combines these two links.)

Description of Relationship

Fish landings for fish that use salt marsh as nursery habitat will increase with increased area of salt marsh habitat.

Summary of Evidence

Tool: The InVEST Fisheries model can estimate harvest volume of single species fisheries for different scenarios, including changing salt marsh habitat area. The marine ecosystem's ability to support fisheries depends in part on availability of habitat for fish, and the impact of this habitat on fish landings can be modeled. The model user guide states that "it is best to compare outputs from multiple runs of the model, where each run represents different scenarios of habitat extent, environmental conditions, or fishing pressure." By keeping environmental conditions and fishing pressure constant, the user can isolate the impact of salt marsh habitat extent on fish landings. Model inputs include life history characteristics of the species of interest, information on fishing pressure, and habitat dependencies such as importance and availability of nursery habitat. Model outputs include volume and economic value of fish harvest. As a single-species model, this tool is best used for locations where a single species of fish is of high importance or interest and is known to rely on salt marsh as nursery habitat. (Download the model here: https://www.naturalcapitalproject.org/invest/).

Other Factors

Fisheries Markets and Technology: The model assumes that market operations are fixed. Changes in markets can alter pressures on fish populations and should be considered if such changes are of interest. The model also assumes that harvest rate stays constant over time and that no changes in fishing technology would alter fishing practices.

Substitutability of Nursery Habitat: The model assumes that "habitat dependencies are obligatory" and that no nursery habitat substitutes are available.

Habitat Quality: The model assumes that fish will respond to a change in habitat area and does not take into account the quality of that habitat. One way to adjust or account for habitat is to include in the model only functional salt marsh habitat—that is, salt marsh habitat that falls above a certain threshold for habitat quality.

Population Density: The model assumes that fish survival depends on habitat availability and does not factor in population density.

Predictability: InVEST models are a convenient way to estimate natural capital and ecosystem services; however, they inherently simplify certain ecological processes and make assumptions. These assumptions are well described, and the user can run this model fully aware of its limitations. However, these model limitations mean that there are also output limitations. The fishery model has been tested in multiple cases, and the natural capital project website links to practical applications of the model for reference. The existence of a model that relates salt marsh habitat to commercial fishing indicates the existence of the relationship between these nodes. If no relationship existed, there would be no need for a model.

Source

Natural Capital Project. n.d. "InVEST Fisheries." Natural Capital Project. http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/fisheries.html.

2e: Wildlife Populations → Culture and Heritage
Description of Relationship

Certain wildlife species contribute to the persistence of human culture. As wildlife populations increase, certain communities' cultures can benefit because of the connection between that species and some aspect of culture or heritage.

Summary of Evidence

Certain wildlife species have strong links to specific aspects of human culture, and increased wildlife populations can contribute to the cultural benefit they provide. The species could be animals especially meaningful for a tribe or indigenous group's customs or animals tied to heritage or a community's sense of place. Measuring species' contribution to culture is notoriously hard to quantify. Examples are described below.

Salmon are a key aspect of many northwestern Native American cultures and are heavily integrated into cultural practice (NRC 1996). For example, the Quinault Indian Nation (QIN) culture is tied distinctly to blueback salmon (*Oncorhynchus nerka*). Cultural practices involving food preparation depend directly on these fish, and important traditions are maintained because of salmon availability (Biedenweg et al. 2014). Salmon are seen as a symbol of these tribes' cultural and religious identity (Amberson 2013).

The blue crab, Atlantic blue crab, or Chesapeake blue crab (*Callinectes sapidus*) has been harvested for centuries but plays a distinct role in Maryland's culture today. Not only does the blue crab industry provide livelihoods for many fishermen, but those involved in the crab fishery are part of a cultural community that has value for fishermen as well as for the wider population (Paolisso 2007a; Paolisso 2007b). Cultural activities such as festivals highlight crab, and some see the blue crab as a symbol of the Chesapeake Bay (Chesapeake Bay Program 2012).

Many marine species play an important role in culture through the strong role that food has in creating and maintaining culture. Other marine species, and some food species, are cultural icons and symbols. Examples include the Pacific Northwest Tribe totem animals, which are often whales and seabirds (including Haida, S'Klallam, Makah, Quinault, Hoh), and the blue crab, which the Chesapeake Bay Program had stamped on sewer drains to remind people to protect the bay. Notably, relationships between wildlife and culture are not necessarily constant over time.

Other Factors

In some cases, cultural perception is important. In the case of the Maryland blue crab, Paolisso (2007b) points out that for some people, the idea of eating a crab on the Maryland coast is more important than the actual type of crab they are eating. Each individual cultural case will be different, and substitutability of services provided by a species must be considered.

Strength of Evidence

Low. No generalizable evidence explains the relationship between wildlife and cultural importance. Each culturally important species must be evaluated on a case-by-case basis within the context of local culture(s).

Predictability: This link is highly context dependent. The link between wildlife and culture depends on the species, the community that values that species, and the importance of that species to that community. Studies relating wildlife species to culture will have to be performed on a case-by-case basis. As seen above, the link is very strong at some sites, but it may be irrelevant at others.

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2f: Wildlife Populations→ Recreation (Birding, Whale Watching, Scuba Diving and Snorkeling, Hunting, Fishing, Photography, and Wildlife Viewing)

Description of Relationship

Wildlife populations create and contribute to recreational opportunities. With an increase in wildlife populations, there will be additional wildlife-related recreational opportunities, resulting in increased recreating, increased recreation quality, or increased types of recreational opportunities.

Summary of Evidence

An increase in wildlife populations will (most likely) increase the likelihood that a person recreating will have a viewing opportunity of a particular species. Wildlife-based recreation includes birding, whale watching, scuba diving and snorkeling, hunting, fishing, photography, and wildlife viewing. Depending on the species available at the site of interest, recreational opportunities will differ. Determining the relationship between wildlife populations and recreational opportunities requires data on wildlife population sizes and number of people recreating or survey results on quality of recreation experiences.

Many factors determine the decision to engage in wildlife-based recreation and satisfaction with that activity (see below). Wildlife have to exist for these recreational opportunities to be possible. "Non-consumptive use of wildlife requires a predictable occurrence of the target species within a fairly small spatial area" (Duffus and Dearden 1990).

No studies were found that directly link wildlife population numbers to recreation outcomes, but studies do report data related to wildlife-related recreational/tourist activities. The International Fund for Animal Welfare (IFAW) report on global whale watching presents data on whale watching tourist numbers for Alaska, Hawaii, Washington, Oregon, California, New England, the Eastern Seaboard, and Florida/Gulf of Mexico (O'Connor et al. 2009). A case study for Johnstone Strait showed that a population of 190 Orca whales resulted in 10,000 whale-related visits in 1980 and 15,000 visits in 1989 (Duffus and Dearden 1993). It has also been found that there is a positive relationship between rarity of birds seen at a site and visitor numbers to that site (Booth et al. 2011). It appears that the evidence available makes the link between recreation outcomes and the *presence* of wildlife populations, rather than specific population sizes. The marginal benefit of an additional animal is not clear from the evidence and would likely differ on the basis of wildlife taxa.

There appear to be more studies that link wildlife populations directly to economic outcomes (tourism revenues or willingness to pay for visits to engage in wildlife-based recreation) than studies involving recreation-based non-monetary indicators. See link 2g,10g for more details.

Tool: The InVEST Recreation and Tourism model can estimate both current recreational patterns as well as future patterns of use under alternate scenarios (which could include increased salt marsh habitat). The model predicts person-days of recreation on the basis of natural habitat locations in relation to other features that factor into decisions about recreation location. The model does not predict specific types of recreation, only person-days at a site. Users can input their own variables that they consider important for predicting recreational use of a site—spatial data on wildlife presence or abundance could be used as an input data type to attempt to predict person-days. Person-days are estimated using

geo-tagged photos from the website flikr. Data requirements include shapefiles or rasters of user-determined visitation predictors and data on the future state of those predictors for scenario analysis. (Download the model here: https://www.naturalcapitalproject.org/invest/).

Other Factors

Laws, Permits, Permissions: Some forms of wildlife-based recreation depend on certain permits or permissions. There are often regulations on hunting and fishing, so it does not necessarily follow that an increase in a species' population will directly relate to the number of people able to harvest that species recreationally. Restrictions are sometimes applied to habitat or nesting areas for threatened or endangered species, so members of the public are not able to view them even if the population of that species is increasing.

Population Size Versus Diversity: For some recreational activities, species diversity matters more than number of animal individuals present. Diversity is especially relevant for birders.

Population Size Versus Visibility: For most species, visibility is a key aspect of wildlife-based recreation. If a species is particularly camouflaged, lives in dense habitat, is nocturnal, or is generally hard to see, it will be hard to link that species to wildlife-based recreation.

Facilities: A site may become more attractive for wildlife-based recreation if it provides facilities designed to provide services to visitors (Duffus and Dearden 1990).

Negative Feedbacks: Though increased populations of wildlife species can yield recreation or tourism benefits, increased recreational activities could negatively affect wildlife. Increased disturbance, noise, interactions with people, and facility construction can alter wildlife behavior and potentially decrease population numbers (Green and Giese 2004). If recreation negatively affects wildlife to the extent that populations die off or migrate, the original recreational benefits from those wildlife species will also disappear.

Strength of Evidence

Low. Though there are many logical connections to be made between wildlife populations and wildlife-based recreation, there are few studies that report data linking wildlife populations to recreational visits or tourist numbers. Site-specific information and data will be necessary to make estimates of recreational outcomes related to wildlife populations. Individual studies and reports have made the existence of the link between aspects of wildlife populations and visitors to a site clear; however, the relationship will vary on the basis of the wildlife species and the site (O'Connor et al. 2009; Booth et al. 2011).

Predictability: InVEST models are convenient tools to predict ecosystem services-related outcomes; however, they inherently simplify certain processes and make assumptions. These assumptions are well described, and the user can run this model fully aware of its limitations. However, model limitations mean output limitations. The recreation model has been tested in multiple cases, and the natural capital project website links to practical applications of the model for reference. Spatial data on wildlife presence or abundance are needed to use this model to predict wildlife impacts on recreation person-days. Other types of models can also be used to predict how visitation or recreation will change on the basis of wildlife species (see example below).

Example. Studies such as that performed by Booth et al. (2011) relating species rarity to the number of visitors to a site can provide site-specific data that will enable detailed descriptions of the connection between wildlife and recreation indicators. Though this study used species rarity as a predictor, it would be possible to also use data on population numbers, species diversity, or some other wildlife indicator as a predictor variable.

Sources

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2f,10g: Wildlife Populations → Recreation Value

(This link represents a combination of two links in the model—wildlife populations → recreation → recreation value—because there is evidence that combines these two links.)

Description of Relationship

Wildlife populations provide a resource that people are willing to pay to enjoy. These recreational activities also bring tourism dollars to a community.

Summary of Evidence

There are multiple ways to estimate recreational value, including revealed preference methods (based on travel cost or direct spending on a recreational activity related to a certain species) and stated preference studies. Although most valuation studies attempt to put a value on recreational use as a whole (sometimes with a few distinctions among habitat types, but more often with factors related to the user, such as household income), a few have examined the value of individual species to recreation.

Tool: The U.S. Geological Service (USGS) Benefit Transfer Toolkit provides regression functions that estimate the value of hunting, fishing, and wildlife watching opportunities on the basis of the type of species involved. The wildlife viewing function gives separate values for birds, charismatic megafauna, and general wildlife; the hunting function gives a separate value for waterfowl; and the fishing function gives separate values for tuna, and salmon. These values can provide a general idea of the recreational value of a particular species' presence in a region.

The studies listed below provide examples of the type of research that can be done to illustrate the recreation value of certain wildlife species or taxa. A revealed preference (travel cost) study on birders coming to see migratory shorebirds in Delaware Bay found that these birders were willing to spend $32–$142/trip/household or $131–$582/season/household to view migratory species (Edwards et al. 2011). A contingent valuation study conducted at the same site found that people were willing to pay $66–$96/trip/household for a day trip, and $201–$428/trip/household for an overnight trip (Myers et al. 2010). A study gathering data on Orca watching on Vancouver Island found that visitors were willing to pay $370 per trip to see orcas in 1986, and $400 per trip in 1989. However, it is not clear which method was used to collect that data (Duffus and Dearden 1993). An International Fund for Animal Welfare (IFAW) report on global whale watching provides data on trip expenditures and jobs provided by whale watching for Alaska, Hawaii, Washington, Oregon, California, New England, the Eastern Seaboard, and Florida/Gulf Coast (O'Connor et al. 2009). A follow-up IFAW report lists the economic value of a single whale in various regions of Australia, and this value ranges from AUD$32,000 to $1.25 million (present value at a 2.65% discount rate) (Knowles and Campbell 2011).

Other Factors

Marginal Value: The marginal value of an individual animal is likely different depending on total population size and the wildlife taxa. For example, the marginal value of one mackerel (a schooling fish) is most likely very different from the marginal value of one whale (a charismatic marine mammal). No studies could be found that discuss this issue, but it should be kept in mind when valuing wildlife populations.

Presence Versus Population Numbers: There is a distinction between putting a recreational value on the *population size* of a species and the *presence* of a species. Many studies do not take population numbers into account when collecting information on value, rather they collect data related to the opportunity to view/catch/photograph the species. Exact wildlife population numbers as an input are sometimes less important than species presence. However, once a population drops below a certain threshold that makes the opportunity to view/catch/photograph the species less likely, the recreational value may be affected.

Strength of Evidence

Moderate. There is clearly a link between wildlife populations and recreational value—people are willing to pay for wildlife-based recreation experiences. The specific economic value of those recreational experiences will be site specific and will depend on the species and recreational activity being examined. Many individual valuation studies for recreation have been conducted across the United States and Canada; these studies are captured in the USGS Benefit Transfer Toolkit database and are used to calculate average regional values and to create meta-regressions for valuation of certain recreational activities. The database appears to be fairly complete, but it is unclear how recently it has been updated; some more recent valuation studies may not be included. The strength of evidence should ultimately be determined by how the toolkit is used; application of meta-regressions is much better than use of average and point estimates. Users should fully consider that the average values from the USGS Benefit Transfer Toolkit were calculated using studies that were not necessarily done with coastal recreational resources in mind, and this fact may affect valuation.

Predictability: The USGS Benefit Transfer Toolkit or other benefit transfer methods make it possible to predict the relationship between wildlife and recreational value.

Example. Using travel cost or contingent valuation methods, it is possible to carry out studies that examine the monetary value of recreational experiences related to wildlife. Though they have limitations (see link 10g for more detail), these methodologies are widely accepted as a reliable way to assign economic values to recreational activities.

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2g: Wildlife Populations → Research and Education

Description of Relationship

Wildlife populations provide opportunities for research and outdoor education. Increased wildlife populations allow for a greater number of research studies as well as for opportunities for experiential outdoor learning.

Summary of Evidence

This link will be highly dependent on nearby educational and research resources. For National Estuarine Research Reserve (NERR) sites, it is likely that scientific and educational resources will be accessible. At the most basic level, for research on or experiential education about wildlife populations to exist, the wildlife populations must exist at the research or education site.

Though it is possible to educate students about wildlife without viewing wildlife species, experiential learning has been shown to provide additional benefits. Experiential learning is "learning from the real-world," "characterized by variability and uncertainty," and engages the student in an environment with high active involvement (Gentry 1990). Experiential learning has been linked to a deeper understanding of subject matter than understanding acquired in a typical classroom learning (Eyler 2009). Viewing wildlife while learning about it will therefore enhance a student's learning experience. NOAA already values the benefits of experiential learning by providing resources such as the Bay Watershed Education and Training (B-WET) program, which funds "locally relevant, authentic experiential learning for K-12 audiences." A review of the Chesapeake B-WET program showed linkages between student B-WET participation and environmental stewardship and literacy (NOAA, n.d.).

Measuring or predicting the research or education benefits provided by wildlife populations requires data on wildlife populations as well as on the number of scientific papers, reports, theses, educational programs, or students attending the programs related to wildlife at a marsh site.

Other Factors
If there are many substitute options for studying wildlife populations or teaching about them, the link between wildlife populations at a site of interest and research or education may not be strong. The link may exist, but substitutability should be considered.

Strength of Evidence
None. Though many logical connections exist between wildlife populations and research/education benefits, there is little to no published evidence that generally supports those connections. Site-specific data may be available at a local level to improve the evidence grade of this link for a particular site.

Sources
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2h: Wildlife Populations (Shellfish) → Nutrients in Estuary Water
Description of Relationship
As shellfish feed, they draw water in and filter nitrogen from water as organic particulates are drawn over the cilia along their gills. Impacts of eutrophication can be reduced due to shellfish filtration. This link is valid only if there is a significant shellfish population at the site of interest.

Summary of Evidence
Bivalves can remove nitrogen from the water (and thus help to reduce effects of eutrophication) in three separate pathways: (1) assimilation of nitrogen (N) into shells and tissues, (2) burial of nitrogen into sediments, and (3) enhancement of denitrification in bottom sediments. Bivalves feed by filtering particulates from the water column, and they expel waste as feces and pseudofeces. These waste biodeposits fortify bottom sediments with organic matter, part of which is nitrogen. This process allows bacterially driven denitrification to occur in the bottom sediment, removing biologically available nitrogen from the system and allowing it to escape in the form of nitrogen gas, which is released to the atmosphere (Newell 2004; zu Ermgassen et al. 2017). Denitrification rates can be affected by overall bivalve biomass and reef structure—though factors affecting these rates are not yet well understood and it is difficult to accurately predict the amount of nitrogen removed (zu Ermgassen et al. 2017).

Though water filtration rates can differ depending on shellfish species and other abiotic factors (see below), for Eastern oysters (*Crassostrea virginica,* a common North American species) the rate can be represented by this equation:

$$\text{Filtration rate (L hr}^{-1}\text{ m}^{-2}) = N(8.02W^{0.58}e^{[-0.015*(T-27)^2]})$$

Where N is the density of oysters per m², W is the dry tissue weight in grams, and T is temp in °C

(zu Ermgassen et al. 2013).

For a more detailed summary of water filtration services provided by oysters specifically, see zu Ermgassen et al. (2017).

Tool: The Nature Conservancy "Mapping Ocean Wealth" oyster calculator can be used to roughly calculate current bay water filtration rates by oysters (M L/h) for a select number of bays across the United States. With data on bay volume, bay residence time, water temperature, current oyster reef area, mean oyster lengths, and mean oyster densities, users can calibrate the model to a new site. (http://oceanwealth.org/tools/oyster-calculator/)

Tool: The FARM model (Ferreira et al. 2007) is available from NOAA. In addition to modeling expected shellfish production from aquaculture, the model can also calculate the estimated water quality changes resulting from aquaculture oysters (Bricker et al. 2014). Among the water quality changes that can be evaluated are nitrogen removal, chlorophyll a levels, and dissolved oxygen levels. Required data inputs include culture practices (farm layout, species, stocking density) as well as environmental parameters such as shellfish food particles in the water column. The model provides a variety of outputs, including the mass of nitrogen removed by shellfish filtration (Bricker et al. 2014).

A study applying the FARM model in 14 locations across 9 countries and 4 continents found that nitrogen removal by shellfish farms ranged from 105 lbs/acre/year to 1356 lbs/acre/year. Mean removal was 520 lbs/acre/year (Rose et al. 2015). Technically, shellfish grown in aquaculture may not be considered wildlife, but they are a wildlife species that contributes to nutrient removal.

Note: The FARM model is applicable only to aquaculture shellfish. Therefore, the tool should be used only if aquaculture sites are established in the salt marsh site of interest, or if the intervention to change the quantity or quality of salt marsh creates conditions conducive to new or enhanced aquaculture sites outside of or adjacent to the marsh. In short, FARM is applicable only if the intervention to restore/protect a salt marsh somehow establishes new or enhanced aquaculture sites.

Other Factors
Species and Size of Shellfish: Different species of bivalves filter water at different rates (Moehlenberg and Riisgaard 1979). Age and size of the bivalve will also determine how much water they are able to filter; larger bivalves filter more water (Moehlenberg and Riisgaard 1979; Gerdes 1983).

Temperature: For some species of bivalves, water filtration rates increase with increasing temperature (Hutchinson and Hawkins 1992; Haure et al. 1998).

Salinity: Bivalve filtration rates can be affected by salinity level. At low salinity, filtration rates have been seen to drop, and filtration rates generally increase with increasing salinity—up to a point of physiological stress (Hutchinson and Hawkins 1992).

Strength of Evidence
High. Multiple sources, models, and tools are available to estimate and describe the filtration services provided by shellfish. Numerous studies discuss the ability of shellfish to remove nutrients from estuary water.

Predictability: The predictability of this relationship is high due to the availability of multiple models and tools to estimate filtration services provided by shellfish. However, the relationship is context specific, and these models and tools require site-specific data to establish an accurate prediction. Tools such as FARM can provide numerical predictions of the amount of nutrients removed by shellfish.

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2i: Wildlife Populations (Shellfish) → Turbidity

Description of Relationship

As shellfish feed, they draw water in and filter out particles, improving water clarity and reducing the impacts of eutrophication. This link is valid only if there is a significant shellfish population at the site of interest.

Summary of Evidence

It is understood that filter-feeding bivalves can help to mitigate estuarine eutrophication through their filtration abilities (Wall et al. 2011). Bivalves can filter particles greater than 5µm in diameter with high efficiency. Some particles are used as food, and others are biodeposited as waste (zu Ermgassen et al. 2017). Experimental treatments have shown that bivalves can lower chlorophyll a and particulate concentrations as well as increase light penetration into the water column (Wall et al. 2011). A mass-balance equation model developed for the Chesapeake Bay predicted that a tenfold increase in oyster biomass would reduce light attenuation (i.e., increase light penetration into the water column) by 20% (Cerco and Noel 2007).

Though water filtration rates can differ depending on shellfish species and other abiotic factors (see below), for Eastern oysters (a common North American species) the rate can be represented by this equation:

$$\text{Filtration rate (L hr}^{-1}\text{ m}^{-2}) = N(8.02\,W^{0.58}e^{\,[-0.015*(T-27)^2]})$$

Where N is the density of oysters per m^2, W is the dry tissue weight in grams, and T is temp in °C

(zu Ermgassen et al. 2013).

For a more detailed summary of water filtration services provided by oysters specifically, see zu Ermgassen et al. (2017).

Tool: The Nature Conservancy "Mapping Ocean Wealth" oyster calculator can be used to roughly calculate current bay water filtration rates by oysters (M L/h) for a select number of bays across the United States. With data on bay volume, bay residence time, water temperature, current oyster reef area, mean oyster lengths, and mean oyster densities, users can calibrate the model to a new site (http://oceanwealth.org/tools/oyster-calculator/).

Tool: The FARM model (Ferreira et al. 2007) is available from NOAA. In addition to modeling expected shellfish production from aquaculture, the model can also calculate the amount of particulates/detritus filtered by shellfish, allowing for calculations of reduced turbidity. Required data inputs include culture practices (farm layout, species, stocking density) as well as environmental parameters such as shellfish food particles in the water column.

Note: The FARM model is applicable only to aquaculture shellfish. Therefore, the tool should be used only if aquaculture sites are established in the salt marsh site of interest, or if the intervention to change the quantity or quality of the salt marsh creates conditions conducive to new or enhanced aquaculture sites outside of or adjacent to the marsh. In short, the FARM model is applicable only if the intervention to restore/protect the salt marsh somehow establishes new or enhanced aquaculture sites.

Other Factors

Species and Size of Shellfish: Different species of bivalves filter water at different rates (Moehlenberg and Riisgaard 1979). Age and size of the bivalve will also determine how much water they are able to filter; larger bivalves filter more water (Moehlenberg and Riisgaard 1979; Gerdes 1983).

Temperature: For some species of bivalves, water filtration rates increase with increasing temperature (Hutchinson and Hawkins 1992; Haure et al. 1998).

Sediment Load: There appear to be sediment load thresholds above which bivalve filtration becomes less effective (Barille et al. 1997).

Salinity: Bivalve filtration rates can also be affected by salinity level. At low salinity, filtration rates have been seen to drop, and filtration rates generally increase with increasing salinity—up to a point of physiological stress (Hutchinson and Hawkins 1992).

Strength of Evidence

High. Multiple sources, models, and tools are available to estimate and describe the filtration services provided by shellfish. Numerous studies discuss the ability of shellfish to reduce turbidity/increase light availability in the water column.

Predictability: Predictions of this relationship are possible due to the availability of multiple models and tools to estimate filtration services provided by shellfish. However, the relationship is context specific, and these models and tools require site-specific data to establish an accurate prediction. Tools such as the FARM model can provide numerical predictions of the amount of particulates/detritus removed by shellfish; however, translating filtered detritus to changes in turbidity levels can prove somewhat difficult.

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Links 3a-g

3a: Sediment Accumulation → Marsh Elevation

Description of Relationship

Sediment accumulation affects marsh elevation. As salt marshes accumulate sediments, the elevation of the marsh rises. The mean rate of elevation change for high-elevation marshes is 3.0 mm/year and for low-elevation marshes, 6.9mm/year; however, this rate is closely linked to sediment accretion, so changing accretion rates will alter elevation change rates (Kirwan et al. 2016).

Summary of Evidence

Sediment accumulation in a marsh is often measured in mm/year, which corresponds to an elevation change. A meta-analysis of 179 accretion or elevation measurements in the United States, Canada, the United Kingdom, France, and Spain showed that the average rate of elevation change is 3.0 mm/year for high-elevation marshes and 6.9mm/year for low-elevation marshes (Kirwan et al. 2016). However, this rate will vary depending on sediment accumulation by the marsh. The meta-analysis shows that sediment accretion measurements and elevation change rates are not statistically significantly different from one another, indicating that accretion rates and elevation change are directly related (i.e., one measurement could be substituted for another) (Kirwan et al. 2016). If sediment accumulation rates are known to be changing in the marsh, it could be assumed that marsh elevation changes will vary in a similar manner.

Tool: The Marsh Equilibrium Model (MEM) uses physical and biological inputs to predict a variety of outcomes related to salt marshes, including elevation. The model assumes that a marsh site's elevation will depend on rate of sea level rise and local sediment supply. By calibrating the model to local estuary data and accretion rates of the marsh, the model can output elevation change estimates over a period of 100 years (DCERP 2013; University of South Carolina 2010). The model can be accessed on the web here: http://marsh.baruch.sc.edu/.

Other Factors

Inorganic sediments are not the only sources of elevation change in a marsh. Organic plant matter in marsh soil also provides an important source of elevation gains (Neubauer 2008).

Strength of Evidence

Moderate. The Kirwan et al. (2016) meta-analysis provides evidence that sediment accretion rates and marsh elevation changes are closely linked. However, other factors like sea level rise and other sources of elevation change (like organic plant material) make the connection between these two variables less certain. For rough estimations, it is possible to use sediment accretion rates to approximate marsh elevation changes.

Predictability: Using tools like the Marsh Elevation Model, it is possible to predict marsh elevation changes, given local data on physical and biotic factors. The model takes into account many of the factors that produce elevation changes. It has been used in the scientific literature to estimate marsh elevation changes (Schile et al. 2014). Rough predictions of elevation change can be estimated on the basis of sediment accumulation rates.

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3b: Marsh Elevation → Wave Attenuation

Description of Relationship

Marsh elevation plays a role in how well a marsh is able to attenuate wave energy. Coastal bathymetry and water depth influence wave energy.

Summary of Evidence

Wave energy dissipation in a salt marsh environment results from (1) changes in bathymetry to shallower depths, (2) bed friction, and (3) vegetation friction. This link focuses on factor 1, changes in bathymetry to shallow depths, but also incorporates factor 2, bed friction.

Marsh elevation will determine water depth throughout the marsh. Higher elevations correspond with shallower water. In shallow water, bed friction attenuates wave heights as waves travel toward land (Gedan et al. 2011). The maximum height of a wave is proportional to the depth of water between the bed surface and sea level, so shallower water means that maximum wave height is lowered. It has also been found that wave-induced shear stress declines with bed elevation in intertidal environments, such as salt marsh (Gedan et al. 2011). In a meta-analysis of shoreline protection services provided by salt marshes, it was discovered that increased marsh elevations increased wave attenuation (Shepard et al. 2011).

Tool: The Wave Height Analysis for Flood Insurance Studies (WHAFIS) model developed by the Federal Emergency Management Administration (FEMA) uses representative transects to compute wave crest elevations in a particular study area (FEMA 2007). The model can be used to predict storm surge elevations, floodplain extent, and details of wave propagation over flooded areas (Conner et al. 2011).

Other Factors

Biotic Marsh Features: Various features of a salt marsh can affect the level of wave attenuation it is able to provide. These features include vegetation density (Möller 2006; Shepard et al. 2011; Anderson and Smith 2014), species composition (Möller 2006; Yang et al. 2012), vegetation height (Möller 2006; Marsooli et al. 2017), and biomass production (Shepard et al. 2011). These features affect the level of friction and drag that the marsh extends on passing water, and they can determine the percentage of wave energy that is absorbed by the marsh.

Abiotic Marsh Features: Abiotic features that influence the level of wave attenuation by a marsh include vegetation submergence and water depth (Koch et al. 2009; Gedan et al. 2011; Anderson and Smith 2014), elevation gradient (Yang et al. 2012), marsh width (Möller and Spencer 2002; Shepard et al. 2011), and configuration of the coastline (Möller and Spencer 2002). All of these features influence either the nature of the incoming waves or the ability of the marsh to absorb that wave energy.

Strength of Evidence

Fair. It is understood that marsh elevation affects wave attenuation capability. Models that estimate wave attenuation consistently use bathymetry inputs, because elevation is considered an important factor in wave energy dissipation.

Predictability: It is difficult to isolate the influence of elevation on wave attenuation in a salt marsh environment, because there are multiple aspects of a marsh (including frictional effects of vegetation) that determine the amount of wave attenuation. However, multiple models using elevation inputs do exist for predicting wave attenuation.

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3c: Change in Salt Marsh Quantity or Quality (Vegetation) → Wave Attenuation

Description of Relationship

Salt marsh vegetation attenuates incoming wave energy and reduces the height and force of waves reaching areas behind the marsh.

Summary of Evidence

Wave energy dissipation in a salt marsh environment results from (1) changes in bathymetry to shallower depths, (2) bed friction, and (3) vegetation friction. This link focuses on factor 3, vegetation friction. The basic relationship between vegetation friction and wave energy dissipation is summarized succinctly in Arkema et al. (2017): "at the most fundamental level, a plant stem immersed in a moving fluid will experience viscous and form drag forces. These forces, in turn, cause the flexible plants to move, further perturbing the surrounding fluid and promoting turbulence, which dissipates energy."

It is widely accepted that coastal salt marshes act as a buffer for inland areas and provide coastal protection from incoming waves; meta-analyses show a positive effect of salt marshes on wave attenuation (Shepard et al. 2011; Narayan et al. 2016). The amount of wave attenuation provided by a particular marsh varies widely depending on multiple biotic and abiotic features of the site (Arkema et al. 2017). The majority of wave attenuation studies focus on wind-produced waves, not waves produced under storm surge conditions. However, storm surge protection is often of higher importance to coastal managers (Shepard et al. 2011). (Note: a modeling approach for approximating wetland reduction of storm surge can be found in Wamsley et al. 2010). In one of the few meta-analyses that examined storm waves, it was found that salt marshes have wave attenuation rates (proportion of wave height reduced per meter of land traversed) equal to 0.0001 for storm waves and attenuation rates of 0.018 for normal wind-produced waves (Gedan et al. 2011). Observational studies of storm surge attenuation show that for 1m of storm surge reduction, between 4km and 25km of marsh are needed (Shepard et al. 2011). One study that used an experimental wave flume tank to examine storm surge conditions found that across 40m of marsh, a salt marsh dissipated wave energy 11.9%–19.5% for waves with heights 0.2m–0.4m and dissipated wave energy by 13.8%–16.9% for waves with heights 0.6m–0.9m (Möller et al. 2014). Many studies highlight the non-linear nature of the relationship between salt marsh vegetation and wave energy dissipation, meaning that finding an average % reduction in waves per meter of marsh makes little sense (Koch et al. 2009; Shepard et al. 2011; Möller et al. 2014).

Tool: SWAN (Simulating Waves Nearshore) models do just what their name implies, simulate waves in nearshore areas. These models can be adapted to model wave energy dissipation over vegetated areas, and they are considered to be some

of the most detailed wave dissipation 2D numerical models available (Suzuki et al. 2011). Due to the complex nature of these models, expertise in their use is likely required. SWAN models are Eulerian flow models, and they calculate wave attenuation given vegetation characteristics (height, density), wave parameters (height, frequency), site bathymetry and topography, and the bulk drag coefficient (Suzuki et al. 2011; McIvor et al. 2012). It is possible to account for horizontal variation in vegetation characteristics of a site if vegetation species differ throughout a marsh (McIvor et al. 2012). Using SWAN models as Suzuki et al. (2011) did to calculate wave dissipation over vegetated areas has been shown to be successful on the basis of validations against other model simulations as well as applications of the model for field measurement (Suzuki et al. 2011). These models do not account for vegetation motion and plant flexibility.

Tool: XBeach is a two-dimensional model for wave propagation, long waves and mean flow, sediment transport, and morphological changes of the nearshore area, beaches, dunes, and back barrier during storms (Roelvink et al. 2009). This model was originally built for sandy shorelines, but recent additions include vegetation modules that incorporate the effects of coastal vegetation on wave heights and erosion (see online description here: https://publicwiki.deltares.nl/display/ VegMod/XBeach-VEG). See van Rooijen et al. (2016) for an application of the XBeach model used to estimate wave heights at vegetated sites. Another application of the XBeach model to estimate wave attenuation in salt marsh areas can be found in Songy (2016). Download the model here: https://oss.deltares.nl/web/xbeach/10.

Tools: See Chapter 4, Table 4.1, Appendix 4.1, and Appendix 4.2 of the WAVES technical report "Managing Coasts with Natural Solutions" for a summary of additional coastal engineering models and tools. Depending on the local site, type of wave(s), and technical expertise available, different modeling approaches may be more or less applicable (Kroeker et al. 2016).

Other Factors

Biotic Marsh Features: Various features of salt marsh vegetation can impact the level of wave attenuation it is able to provide. These features include vegetation density (Möller 2006; Shepard et al. 2011; Anderson and Smith 2014), species composition (Möller 2006; Yang et al. 2012), vegetation height (Möller 2006; Marsooli et al. 2017), and biomass production (Shepard et al. 2011). These features affect the level of friction and drag that the marsh extends on passing water, and they can determine the percentage of wave energy that is absorbed by the marsh.

Abiotic Marsh Features: Abiotic features that influence the level of wave attenuation by a marsh include vegetation submergence and water depth (Koch et al. 2009; Gedan et al. 2010; Anderson and Smith 2014), elevation gradient (Yang et al. 2012), marsh width (Möller and Spencer 2002; Shepard et al. 2011), and configuration of the coastline (Möller and Spencer 2002). All of these features influence either the nature of the incoming waves or the capacity of the marsh to absorb that wave energy.

Seasonal Variation: Because vegetation characteristics can change seasonally, there is temporal variation in the capacity of salt marsh vegetation to provide wave buffering services (Möller 2006; Shepard et al. 2011; Marsooli et al. 2017).

Spatial Variation: Many studies make reference to the fact that marsh wave attenuation is a non-linear relationship and varies over space. In a meta-analysis focused on the non-linearity of coastal wetland ecosystem service provision, it was found that % wave attenuation by salt marsh decreases non-linearly with increasing distance from the marsh edge (Koch et al. 2009). That result was also found in a study that reports the most seaward 10m of a 310m wide marsh averaged a 1.1%– 2.1% wave height attenuation per meter of marsh, whereas the entire marsh averaged only 0.5% wave height attenuation per meter of marsh (Möller and Spencer 2002).

Strength of Evidence

Moderate. Estimates for amount of wave attenuation differ widely among sources; however, they agree that intact marshes do provide wave attenuation services. The literature does not provide a singular numerical estimate of wave attenuation rates (measured in % wave energy reduced per meter of marsh traversed) due to high variation in attenuation resulting from biotic and abiotic factors specific to each marsh environment. Therefore, the applicability of each study is not very wide; results are closely tied to the marsh where the data were collected. Multiple types of evidence support marsh wave attenuation, including observational field studies, experimental field studies, meta-analyses, and modeling efforts. Multiple sources cite the need for more field studies that examine marsh wave attenuation under storm surge conditions.

Predictability: Some generalizations about wave attenuation by salt marsh habitat are available; however, synthesis studies often emphasize the variation in attenuation based on the marsh context. Modeling salt marsh wave attenuation using models such as SWAN or XBeach are possible; however, these models are data intensive and relatively complex to use. SWAN simulations have been validated against other simulations, but further validation using field data, especially during storm surge conditions, would be beneficial (McIvor et al. 2012). XBeach has been validated using data from flume tank experiments (van Rooijen et al. 2016), but validation of the vegetation module using field data has not occurred to our knowledge.

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3d: Wave Attenuation → Marsh Erosion
Description of Relationship
Wave attenuation by salt marsh vegetation prevents marsh sediment erosion.

Summary of Evidence
A reduction in wave energy means a subsequent reduction in water velocity, water turbulence, and shear stress along the marsh bed (Gedan et al. 2011). All these factors can lead to reductions in erosion of the salt marsh sediment. Shear stress (the force applied by a flowing liquid to its boundary, in this case the marsh bed) was found to rarely reach a level high enough to cause sediment entrainment (the incorporation of sediment into a fluid flow) in a marsh under tidal flow or wind wave conditions (Gedan et al. 2011). In a comparison of erosion rates at vegetated marsh sites and those at sites with exposed sediment, it was found that erosion rates in marshes were 33%–82% lower, depending on the species of marsh vegetation (Coops et al. 1996). This last study was done in a wave tank and not *in situ*, so there may be some limitations to the applicability of its results.

A review and meta-analysis by Shepard et al. (2011) also discusses the potential for reduced erosion in salt marsh environments. The analysis found that reduced erosion was positively correlated with the presence of marsh vegetation in the majority of studies. Shepard et al. (2011) examine vegetation characteristics most often associated with both wave attenuation and shoreline stabilization, finding that vegetation density, biomass production, and large marsh size all positively affect both of these services. The authors note that "this overlap in significant drivers suggests that large marshes that contain dense and productive vegetation will attenuate wave energy and stabilize shorelines more effectively than deteriorating or severely altered marshes" (Shepard et al. 2011). However, this analysis does not examine the mechanisms for reduced erosion; it finds only that marsh vegetation is associated with lower erosion rates than sites with no vegetation. It is unclear whether wave energy attenuation by marsh vegetation is reducing erosion or whether some other aspect of marsh vegetation is causing the effect (Shepard et al. 2011).

In a review of the coastal protection services provided by coastal wetlands, Gedan et al. (2011) make it clear that erosion control by these wetlands is effective only up to a point. They note that coastal wetlands are successful at limiting erosion in low-wave-energy environments, but they are less effective in instances of high wave energy (Gedan et al. 2011). Large waves are capable of eroding wetland sediments, though the review also notes that these large waves are often sources of sediment delivery, which may offset erosion losses to some extent (Gedan et al. 2011).

Various methods are used to measure coastal erosion at a specific site; however, predicting changes in coastal erosion mediated by vegetation changes is somewhat difficult. Erosion in marsh habitat is often predicted using some form of coastal vulnerability index, but the outputs of a vulnerability index do not indicate numerical values of coastal erosion, only relative risk. A short review of relevant coastal vulnerability indices can be found on the European Climate Adaptation Platform site (Climate ADAPT 2011). Integrated modeling approaches to establish numerical (not indexed) estimates of erosion, based on the presence of coastal vegetation, have been developed, but they still incorporate relatively high levels of uncertainty (Guannel et al. 2015). Other new models such as the XBeach vegetation module are capable of estimating erosion at vegetated sites, but these models require further testing (see below).

Tool: XBeach is a two-dimensional model for wave propagation, long waves and mean flow, sediment transport, and morphological changes of the nearshore area, beaches, dunes, and back barrier during storms (Roelvink et al. 2009). This model was originally built for sandy shorelines, but recent additions include vegetation modules that incorporate the effects of coastal vegetation on wave heights and erosion (see online description here: https://publicwiki.deltares.nl/display/VegMod/XBeach-VEG). Two published applications of the XBeach vegetation module for erosion were found; see van Rooijen et al. (2017) for an application of the XBeach vegetation model used to estimate erosion at a beach fronted by eelgrass and Hu et al. (2018) for an application of the XBeach vegetation model used to estimate erosion and morphological change of a salt marsh. Download the model here: https://oss.deltares.nl/web/xbeach/10.

Soil Properties: Soil properties have a large impact on erosion rates. Properties found to influence erosion rates in salt marsh environments include soil bulk density, percent organic matter, percent water, percent coarse particles (Feagin et al. 2009), soil type (and therefore particle size), clay-silt fraction, and amount of soil carbon (Ford et al. 2016).

Shoreline Structure: The structure or shape of a shoreline will influence erosion at a particular site. Bathymetry and microtopography have been shown to influence erosion rates at salt marsh edges (Feagin et al. 2009). Shoreline shape can influence incoming wave energy and creation of longshore currents, which in turn affect erosion (Feagin et al. 2009; NOAA 2017).

Strength of Evidence

Moderate. Multiple lines of evidence point to the erosion-reducing capabilities provided by reduced wave action in salt marsh environments. However, not all studies found consistent results and not all studies measured the same variables, making comparison of results between studies difficult. Two reviews of the protective services provided by salt marshes found trends in the literature that point to reduced erosion based on low wave energy, but causality was not necessarily proven (i.e., it is not always clear that reduced wave action is the cause of reduced erosion).

Predictability: Quantitatively predicting the amount of erosion is data intensive (using models like XBeach). Less data-intensive models are available, but they will result in indexed erosion risk levels, which only provide relative indications of risk.

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3e: Wave Attenuation → Flooding

Description of Relationship

Reduced wave energy can limit wave travel over land and reduce overtopping of beaches, dunes, and other barriers. Attenuation of energy by salt marsh vegetation can also potentially reduce storm surge heights, which could reduce flooding under storm conditions.

Summary of Evidence

As waves, storm surge, or both travel over marsh vegetation, their energy is attenuated by the marsh (see evidence for link 3c). Coastal marsh vegetation increases resistance to landward-propagating flood waves, and it can thus defend against flood events (reduce peak flood levels) (Temmerman et al. 2012). Much attention is being paid to the potential for marshes to act as a natural flood control, especially under conditions of climate change (and potential increased storm activity). Marshes are understood to attenuate waves and reduce storm surge (Wamsley et al. 2010; Temmerman et al. 2013); however, field-based observations of flood surge reduction and associated flooding are scarce and widely variable (Wamsley et al. 2010; Temmerman et al. 2012; Stark et al. 2015). Temmerman et al. (2012) and Stark et al. (2015) cite storm surge reductions by marshes commonly ranging from 4cm to 25cm of flood level reduction per 1 km of marsh, whereas Wamsley et al. (2010) cite surge attenuation ranges from 40cm to 250cm per 1 km of marsh. Measurements of storm surge attenuation by marshes is rare, but Stark et al. (2015) provide an example of how these measurements can be made and analyzed. Modeling efforts to estimate storm surge attenuation by marshes is still developing, but they are more common than in-situ observations and measurements. Temmerman et al. (2012) and Wamsley et al. (2010) provide examples of modeling techniques to estimate surge reduction by wetlands.

Tool: XBeach is a two-dimensional model for wave propagation, long waves and mean flow, sediment transport, and morphological changes of the nearshore area, beaches, dunes, and back barrier during storms (Roelvink et al. 2009). This model was originally built for sandy shorelines, but recent additions include vegetation modules that incorporate the effects of coastal vegetation on wave heights and erosion (see online description here: https://publicwiki.deltares.nl/display/VegMod/XBeach-VEG). Model outputs of wave heights can be translated into flood levels. Download the model here: https://oss.deltares.nl/web/xbeach/10.

Additional resources:

Tool: Though not specific to salt marsh protection from flooding, the NOAA Coastal Inundation Toolkit can act as a primer to coastal inundation modeling. This resource is an online, self-paced course that covers topics such as causes of coastal inundation, community risks, inundation visualization options, risk and vulnerability communication, and links to other resources. Take the course here: https://coast.noaa.gov/digitalcoast/training/coastal-inundation-toolkit.html.

Tool: The NOAA Mapping Coastal Inundation Primer introduces coastal inundation modeling/mapping and is a good resource for users who may be less familiar with the considerations that must be taken into account when using coastal inundation models. Download the primer here: https://coast.noaa.gov/data/digitalcoast/pdf/coastal-inundation-guidebook.pdf.

Other Factors

Biotic Marsh Features: Vegetation intactness and pattern/size of vegetated patches has been shown to influence storm surge reductions by marsh areas in a simulation study (Temmerman et al. 2012).

Abiotic Marsh Features: The structure or shape of a shoreline will influence the capacity of a marsh to attenuate storm surge and reduce flooding at a particular site. Marsh geography and bathymetry are considered major factors that influence peak flood levels (Wamsley et al. 2010; Stark et al. 2015). Geography and bathymetry include marsh elements such as channel width and marsh platform elevation (Stark et al. 2015).

Storm Conditions: The intensity, duration, and track of a particular storm will influence the nature of the incoming waves and surge and therefore the capacity of a marsh to attenuate that incoming energy (Wamsley et al. 2010; Stark et al. 2015). Duration of the surge has been highlighted as particularly important. Lengthy surges will decrease or eliminate a marsh's capacity to provide reductions in surge height (Wamsley et al. 2010; Startk et al. 2015).

Spatial Variation: Temmerman et al. (2012) found that the amount of storm surge that a marsh was able to attenuate differed from one area of a marsh to another. Their modeling simulations indicated that the first 10km of marsh attenuate flood levels by 6cm to 9cm per km of marsh, whereas the next 5km of marsh attenuate flood levels by 15cm to 23cm per km of marsh. This attenuation capacity is applicable only to very large (wide) marsh areas.

Strength of Evidence

Moderate. Estimates of amount of storm surge attenuation differ widely; however, all sources examined agree that intact marshes do provide flood prevention services. The literature does not provide a singular numerical estimate of surge attenuation rates or of flood reduction amounts due to high variation resulting from biotic and abiotic factors specific to each marsh environment. Therefore, the applicability of each study is not very wide; results are closely tied to the marsh where the data were collected. Multiple types of evidence support marsh surge attenuation and flood reduction, including observational field studies (though few) and modeling efforts.

Predictability: Modeling salt marsh storm surge attenuation using models such as XBeach, DELFT3D FLOW (Temmerman et al. 2012), or ADCIRC combined with other wave models (Wamsley et al. 2010) is possible; however, these models are data intensive and relatively complex to use. Model validation against field data is difficult, because so few field measurements of storm surge reduction by marshes are available.

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3f: Marsh Erosion→ Change in Salt Marsh Quantity or Quality

High erosion rates can result in reduced size of a marsh.

Description of Relationship

Marsh erosion rates are highly variable; however, in many locations in the United States, sediment accretion rates offset sediment loss by erosion, meaning that marsh erosion does not result in reductions in salt marsh area (Kennish 2001). There are exceptions, and salt marsh erosion should not be ignored.

Summary of Evidence

Sediment erosion rates must be compared to sediment accretion rates in order to predict whether a salt marsh's area or elevation will change. In much of the United States, erosion rates are offset by sediment accretion (Kennish 2001). Despite this fact, multiple review articles have cited erosion as a cause of reduction in salt marsh area in certain locations (Kennish 2001; Gedan et al. 2009). Certain parts of the Gulf of Mexico (namely the Mississippi, Louisiana, and southern Texas shorelines) tend to have relatively high rates of erosion (Kennish 2001). At sites with high erosion, significant losses in marsh area have been seen; a 1,000-acre reduction in salt marsh area (representing a 25% loss of salt marsh for the county) was reported over a 25-year period in Essex, England, and this loss was attributed to erosion (Cooper et al. 2001). If a reduction in salt marsh size does occur because of erosion, the result may be cascading effects on all of the services that the marsh provides .

Sediment Accumulation: Site-specific factors will be key in determining whether erosion causes a significant loss in salt marsh area (or elevation). Most notably, erosion must be compared with sediment accretion rates to determine whether there is net gain or loss of sediment at a certain site.

Sea Level Rise: Marsh erosion combined with sea-level rise can result in increased loss of salt marsh area due to loss of area, elevation, and submergence (Kennish 2001).

Strength of Evidence
Fair. Consistency of results for this link is limited. In many cases, this link is negligible because sediment accumulation offsets sediment erosion; however, erosion should not be discounted entirely because in some cases it can result in loss of salt marsh area.

Predictability: Predictability for this link is entirely site specific. Erosion and sediment accretion for the site will have to be compared in order to understand if or how a marsh's area might change due to erosion.

Sources
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3g: Flooding → Coastal Protection
Flooding causes damage to coastal structures, infrastructure, and other resources.

Description of Relationship
Predicted flood height and flood frequency can be translated into estimated damages caused by a flood. Less flooding will result in less damage.

Summary of Evidence
It is a given that where coastal development exists and flooding occurs, damages will result (FEMA 2018). Damages caused by coastal flooding are entirely dependent on the structures and infrastructure along the coast that are at risk of such damages. If flood risks, heights, or extents have been calculated, it is possible to estimate which man-made and natural resources are at risk from floods. Resources document multiple ways that flood height and frequency estimates can be translated into coastal protection measures (Schuster and Doerr 2015; Abt Associates 2015). These measures include number of affected homes or structures, change in damage costs (see link 10f for more on such costs), number of days a particular structure is flooder per year, number of days businesses are closed due to flooding, number of people affected by a particular flood (living in homes that were flooded), and miles of transportation infrastructure affected by a flood event.

Tool: The NOAA Coastal Flood Exposure Mapper is an online mapping tool that can be used to begin visualizing different types of flood risks with various population and infrastructure data overlays. Use the mapper here: https://coast.noaa.gov/ digitalcoast/tools/flood-exposure.html.

Other Factors
None.

Strength of Evidence
Moderate. It is clear that coastal flooding causes damage and that when less flooding happens less damage occurs. How flooding information is transformed into damage measures is dependent on what would be most useful to the community of interest.

Predictability: If flood heights and extents have been predicted, spatial overlays of buildings and infrastructure offer an easy tool for predicting numbers of structures affected by a flood event. Methods outlined in Dutta et al. (2003) show

how flood damage losses can be calculated using stage-damage functions, which define the relationship between flood parameters (depth) and possible damage (percent of property value that gets damaged). This model outputs economic losses, which are discussed more fully in link 10f. Other modeling tools such as FEMA HAZUS are available for making detailed predictions of flood damages (FEMA 2018b). However, FEMA HAZUS requires user expertise and training.

Sources

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3c,d,e: Change in Salt Marsh Quantity or Quality → Marsh Erosion and Flooding Risk

(This link represents a combination of three links in the model: Change in salt marsh quantity or quality → wave attenuation → marsh erosion AND Change in salt marsh quantity or quality → wave attenuation → flooding. These links are combined here because they are combined in a model.)

Description of Relationship

Salt marsh habitats can help protect coastal areas from erosion and inundation. Salt marshes provide a buffer that can intercept incoming waves and storm surge as well as stabilize soils to help prevent erosion.

Summary of Evidence

Tool: The InVEST Coastal Vulnerability model can provide a qualitative index of coastal exposure to both erosion and inundation in comparison to human settlement and population locations. Though it does not produce quantitative values of inundation level or erosion amount, it can give the user an index value of areas most and least vulnerable under different scenarios. Model outputs include an exposure index, which ranks relative exposure of different coastline areas to inundation and erosion as well as a coastal population raster. The model requires data inputs for geomorphology, relief, spatial extent of natural habitats, net sea-level change estimates, wind and wave exposure, and a surge potential depth contour (details on data requirements can be found here: http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/coastal_vulnerability.html#limitations-and-simplifications). Importantly, this model does not take into account some key habitat characteristics such as density, width, and height of marsh shoots on small scales. But spatial variation in these factors could be reflected qualitatively in the model. This model will be most useful for establishing relative estimates of vulnerability as a baseline or for modeling scenarios of marsh restoration in which new portions of shoreline will be turned into marsh habitat. (Download the model here: https://www.naturalcapitalproject.org/invest/). See Arkema et al. (2013), Arkema et al. (2017), or Cabral et al. (2017) for examples of published studies using the InVEST Coastal Vulnerability Model.

Additional resources:

Tool: Though not specific to salt marsh protection from flooding, the NOAA Coastal Inundation Toolkit can act as a primer to coastal inundation modeling. This resource is an online, self-paced course that covers topics such as causes of coastal inundation, community risks, inundation visualization options, risk and vulnerability communication, and links to other resources. Take the course here: https://coast.noaa.gov/digitalcoast/training/coastal-inundation-toolkit.html.

Tool: The NOAA Mapping Coastal Inundation Primer introduces coastal inundation modeling/mapping and is a good resource for users who may be less familiar with the considerations that must be taken into account when using coastal inundation models. Download the primer here: https://coast.noaa.gov/data/digitalcoast/pdf/coastal-inundation-guidebook.pdf.

Tool: The NOAA Coastal Flood Exposure Mapper is an online mapping tool that can be used to begin visualizing different types of flood risks with various population and infrastructure data overlays. Use the mapper here: https://coast.noaa.gov/digitalcoast/tools/flood-exposure.html.

Other Factors

The InVEST model simplifies complex coastal processes, including storm surge, wave fields, natural habitat quality, interactions among variables included in the model, and geomorphic ranking of the coast. See the "limitations and simplifications" section of the model user guide for more detail (http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/coastal_vulnerability.html#limitations-and-simplifications).

Strength of Evidence

Predictability: InVEST models are a convenient way to estimate natural capital and ecosystem services; however, they inherently simplify certain processes and make assumptions. These assumptions are well described, and the user can run this model fully aware of its limitations. However, model limitations mean output limitations. The coastal vulnerability model provides an index of flooding and erosion risk, not numerical estimates.

Sources

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3f,g: Marsh Erosion and Flooding Risk → Shoreline Protection

(This link represents a combination of two different links in the model: marsh erosion → change in salt marsh quantity or quality AND flooding → shoreline protection. These links are combined here because they are combined in a model.)

Description of Relationship

Reduced erosion and flooding can help protect onshore structures, infrastructure, and other resources. Overlap between onshore structures/infrastructure and areas of high/low flooding and erosion risk can be compared from one scenario to another.

Summary of Evidence

Using the output from the InVEST Coastal Vulnerability Model (link 3c,d,e), it is possible to find the overlap between onshore structures or infrastructure and relative risk of flooding and erosion. It would then be possible to compare the overlap of high-risk areas in different scenarios run by the model (example scenarios would include no/reduced salt marsh versus restored/additional salt marsh along the shoreline). It would then be possible to examine where reduced risk to

structures/infrastructure could be promoted by increased salt marsh along the shoreline or to examine the difference in percentage of highly vulnerable areas intersecting with structures/infrastructure and other resources.

Other Factors
Again, the InVEST model quantifies reduction in risk due to change in habitat across a seascape due to habitat restoration or degradation. It will be less useful for exploring site-scale scenarios involving changes in the density or height of vegetation, although these changes could potentially be reflected qualitatively through the habitat ranking input.

Strength of Evidence
Predictability: The limitations of the InVEST model (see link 3c,d,e) limit the predictability for these following links (3f and 3g).

Source
Natural Capital Project. n.d. "InVEST Coastal Vulnerability Model." Natural Capital Project. http://data. naturalcapitalproject.org/nightly-build/invest-users-guide/html/coastal_vulnerability.html.

3c, 3e, 3g, 10f: Change in Salt Marsh Quantity or Quality → Cost of Coastal Damages
(This link represents a combination of four links in the model: Change in salt marsh quantity or quality → wave attenuation → flooding → shoreline protection → cost of damage. These links are combined here because they are combined in a model.)

Description of Relationship
Salt marsh habitat reduces the damages (and associated costs) of coastal flooding due to storm events. Coastal wetlands were found to reduce hurricane flooding damages an average of 10% in counties with wetland coverage. During normal storm years, properties behind salt marshes were found to have on average 20% lower annual flood costs.

Summary of Evidence
Tool: There are storm surge and flooding risk models more complex than InVEST; however, these integrated hydrodynamic, socioeconomic models are complicated and often require expert use (and models developed primarily for insurance purposes, such as the one described below, may be expensive to use). One such model is the Risk Management Solutions (RMS) storm surge flood model, which was used in a study that examined the cost savings that salt marshes could provide during storm conditions (Narayan et al. 2016). The study modeled wetland benefits during an extreme storm event, Hurricane Sandy, as well as benefits of salt marshes during one year of average storm conditions. The model outputs reveal that, during Hurricane Sandy, counties with coastal wetlands had an average of 10% lower storm damages than if those wetlands were removed. The RMS model can output the change in storm surge heights along the coast during Sandy in the absence of those wetlands. On average, storm surge heights were shown to increase if wetlands were removed from the model scenario. The second part of the study that examined annual storm damages found that in Ocean County, New Jersey, properties located behind a salt marsh had on average 20% lower annual property damage than properties without salt marsh coverage (Narayan et al. 2016).

The RMS flood model is a hydrodynamic model that calculates propagation of storm surges from the coastal shelf on to land and that accounts for differing storm-surge dissipation capacities by land cover types using a friction coefficient (Manning's n). It has many input data requirements, including wind fields, property values, bathymetry, elevation, and land cover. In this particular study, the RMS model was validated with Hurricane Sandy flood heights measurements.

Other Factors
None.

Strength of Evidence
Predictability: The RMS flood model is a complex hydrodynamic simulation, and its validation with data collected during Hurricane Sandy displays its relative accuracy. Although it is widely recognized that coastal wetlands such as salt marshes provide risk reduction services, the quantitative assessment of these services is not always common practice (Narayan et al. 2016). A model such as the RMS flood model allows for complex flooding simulations, using the presence or reduction of wetlands as input scenarios for calculating estimated flood damages. Narayan et al. (2016) acknowledge that using a

static friction coefficient for all wetlands in the model likely does not account for slight differences in wetland frictional resistance, and it may in fact underestimate vegetation effects on reducing flood heights.

Source

Narayan, Siddharth, Michael W. Beck, P. Wilson, C. Thomas, A. Guerrero, Christine Shepard, Borja G. Reguero, G. Franco, C.J. Ingram, and D. Trespalacios. 2016. "Coastal Wetlands and Flood Damage Reduction: Using Risk Industry-Based Models to Assess Natural Defenses in the Northeastern USA." Lloyd's Tercentenary Research Foundation, London. http://www.conservationgateway.org/ConservationPractices/Marine/crr/library/Documents/CoastalWetlandsandFloodDamageReductionReport.pdf.

Link 4a
4a: Change in Salt Marsh Quantity or Quality → Aesthetics
Description of Relationship
Salt marsh habitat will contribute to an area's aesthetic beauty. The community around and visitors to the habitat will benefit from its aesthetic beauty.

Summary of Evidence
Aesthetics are known to play a key role in people-landscape interactions and in perceptions of a place (Gobster 1999). "Scenic resource management" has entered into land management decisions (Gobster 1999); however, no cases of salt marsh scenic management were identified. Environmental preference research as well as evolutionary anthropology suggest that humans prefer the aesthetics of natural environments that resemble the African savannah because that is where humans evolved (Balling and Falk 1982; Kaplan 1987). It could be said that salt marsh habitats have many of the same features as the savannah—open grassy surfaces with interspersed water features. Though there is no research to support this comparison, it is understood that humans enjoy certain aesthetic features represented in the salt marsh environment.

Because aesthetic value is hard to measure and quantify, it is difficult to find evidence linking salt marshes to aesthetics. However, data could be collected to examine this relationship. Ideas to quantify or monitor aesthetic changes include counting the number of photos tagged at the marsh site on social media sites such as flikr or the number of local marsh-related artists or artworks.

Other Factors
Health of the Marsh: If salt marsh quality is being improved through the focal intervention in a conceptual model, meaningful aesthetic changes may not occur. Depending on the state of the marsh prior to the intervention, people may or may not have aesthetic preferences for a restored marsh over a degraded one, depending on what the ecological degradation looked like.

Type of Marsh: People value the aesthetics of marshes, but some marshes are preferable to others due to the dominant plants and the types of views they offer (e.g., a smelly, muddy marsh is not as appreciated as an emergent, grassy marsh).

Strength of Evidence
Low. Though there is evidence that says humans prefer the aesthetics of natural environments, there is essentially no evidence that assesses the aesthetic benefits of salt marsh habitat. Any connection between salt marshes and aesthetics must be made by extrapolation.

Predictability: Predictability is dependent on the marsh context, the type of restoration, and the values of the near-marsh community.

Sources
Balling, John D., and John H. Falk. 1982. "Development of Visual Preference for Natural Environments." *Environment and Behavior* 14 (1): 5–28.

Gobster, Paul H. 1999. "An Ecological Aesthetic for Forest Landscape Management." *Landscape Journal* 18 (1): 54–64.

Kaplan, Stephen. 1987. "Aesthetics, Affect, and Cognition: Environmental Preference from an Evolutionary Perspective." *Environment and Behavior* 19(1): 3–32.

Link 5a
5a: Change in Salt Marsh Quantity or Quality → Culture and Heritage
Description of Relationship
Salt marsh habitats are of particular cultural importance to certain communities. An increase in salt marsh area will maintain and reinforce the cultural importance of salt marshes.

Summary of Evidence
Each marsh site can have a unique cultural importance. A stakeholder analysis should be performed to identify who places cultural importance on marsh habitats in an area of interest. Cultural importance is particularly hard to quantify due to the difficulty (and perhaps impropriety) of putting a number on cultural values and traditions. Examples of groups with cultural ties to salt marshes are presented below.

Many southeastern Native American tribes, including the Seminole, Guale, and Yemassee utilized salt marshes for resources. Cultural practices including fishing, medicinal plant collection, and shellfish harvesting are tied to these ecosystems (Sanger and Parker 2016). The Gullah/Geechee (descendents of central and western African cultures forcibly brought to the United States by the slave trade in the 1600s) live along the sea islands of central North Carolina to northern Florida, and salt marshes play an important role in their traditional cultural practices. Multiple Gullah/Geechee sacred ceremonies take place in salt marsh environments, and many Gullah/Geechee still practice traditional fishing techniques that depend on healthy marshes. The Gullah/Geecheee also produce crafts such as baskets that are created from salt marsh grasses (Sanger and Parker 2016).

Other Factors
Unique Cultural Traditions: There will be other factors unique to each scenario, and specific cultural practices and traditions should be considered when considering salt marsh restoration or protection.

Access to Marshes: In some locations there are tradeoffs between salt marsh protection and use of marshes by groups who have cultural ties to the location. Accessibility of marshes for these groups should be considered when examining cultural importance.

Strength of Evidence
Low. There are limited resources directly connecting salt marshes to cultural importance. This link may not be relevant in many circumstances, but it may be highly important in the specific locations where there are cultural ties to salt marshes.

Predictability: Site-specific data and stakeholder analysis will provide information for the local importance and strength of this link.

Example. In the locations where there is a community that places high cultural value on salt marshes, this link is clear and very important. A prominent example is the Gullah/Geechee communities that still rely on and place high value on the salt marshes of the southeastern United States (Sanger and Parker 2016). Studies will have to be performed on a case-by-case basis to determine the local connection between salt marsh habitat and culture.

Sources
Sanger, Denise, and Catharine Parker. 2016. "Guide to the Salt Marshes and Tidal Creeks of the Southeastern United States." Southeastern Regional Taxonomic Center, Marine Resources Research Institute: South Carolina Department of Natural Resources. http://www.saltmarshguide.org/wp-content/uploads/2016/11/SaltMarshTidalCreekGuide.pdf.

Link 6a
6a: Change in Salt Marsh Quantity or Quality → Recreation (User Days)
Description of Relationship
Salt marsh habitat provides recreational resources. An increase in salt marsh habitat will provide additional recreational opportunities and increase user-days at the site.

Summary of Evidence
Many sources cite the recreational benefits provided by salt marsh systems, including federal agencies like the National Parks Service (NPS n.d.), state agencies (City of New York Parks and Recreation 2017), academic literature (Gedan et al.

2009), and global organizations like the United Nations (UNEP 2006). Recreation at salt marshes can include swimming, hiking, boating, fishing, hunting, wildlife watching, scenic viewing, birding, biking, and so on. The quality of a site is important—the UN notes that aesthetically pleasing, intact coastal ecosystems are important contributors to the amount of recreational services that a site provides (UNEP 2006). Studies and surveys on individual recreational activities can highlight the links between a site and specific activities.

Tool: The InVEST Recreation and Tourism model can estimate both current recreational patterns as well as future patterns of use under alternate scenarios (which could include increased salt marsh habitat). The model predicts person-days of recreation on the basis of natural habitat locations in relation to other features that factor into decisions about recreation location. The model does not predict specific types of recreation, only person-days at a site. Users can input their own variables that they consider important factors for predicting recreational use of a site, such as natural habitat location, accessibility, built features, and other site attributes. Person-days are estimated using geo-tagged photos from the website flikr. Data requirements include shapefiles or rasters of user-determined visitation predictors and data on the future state of those predictors for scenario analysis. (Download the model here: https://www.naturalcapitalproject.org/invest/). See Reddy et al. (2015) for an example of the InVEST model used to analyze a marsh habitat and Sessions et al. (2016) for an example of the InVEST model used to estimate national park visitation.

Other Factors

Opinions About Predictor Variables: The InVEST model assumes that people's opinions about various predictors will not change in the future (i.e., their attraction or aversion to certain features of the landscape will not change).

Accessibility: A site's accessibility may affect the number of people who use the site. If the site is very difficult to get to or accessible only at certain times of year, the number of recreational user-days will be lowered.

Facilities: A site may become more attractive for recreation if facilities related to recreation (e.g., boardwalks, changing facilities, canoe/kayak rental sites, and so on) are built. These types of facilities should be included as variables in the InVEST model.

Substitutability: If there are many substitute options for visiting salt marshes in the region of interest, there may not be a strong link between an increase in salt marsh at a site and recreation there. The link may still exist, but substitutability should be considered.

Strength of Evidence

Moderate. It is clear that salt marsh habitats provide recreational benefits. However, whether user-days increase due to an increase in salt marsh size or habitat quality is entirely dependent on the individual site and its context.

Predictability: InVEST models are a convenient way to estimate natural capital and ecosystem services; however, they inherently simplify certain processes and make assumptions. These assumptions are well described, and the user can run these models fully aware of their limitations. However, model limitations mean model output limitations. The recreation model has been tested in multiple cases, and the natural capital project website links to practical applications of that model for reference. See Reddy et al. (2015) for an example of the InVEST model used to analyze a marsh habitat and Sessions et al. (2016) for an example of the InVEST model used to estimate national park visitation.

Sources

Natural Capital Project. n.d. "InVEST Recreation and Tourism." Natural Capital Project. http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/recreation.html#visitation-recreation-and-tourism.

Reddy, Sheila M.W., Gregory Guannel, Robert Griffin, Joe Faries, Timothy Boucher, Michael Thompson, Jorge Brenner, Joey Bernhardt, Gregory Verutes, and Spencer A. Wood. 2016. "Evaluating the Role of Coastal Habitats and Sea-Level Rise in Hurricane Risk Mitigation: An Ecological Economic Assessment Method and Application to a Business Decision." *Integrated Environmental Assessment and Management* 12 (2):328–344.

Sessions, Carrie, Spencer A. Wood, Sergey Rabotyagov, and David M. Fisher. 2016. "Measuring Recreational Visitation at US National Parks with Crowd-Sourced Photographs." *Journal of Environmental Management* 183: 703–711.

UNEP (United Nations Environment Programme). 2006. "Marine and Coastal Ecosystems and Human Well-Being: A Synthesis Report Based on the Findings of the Millennium Ecosystem Assessment." https://www.millenniumassessment.org/documents/Document.799.aspx.pdf.

Link 7a

7a: Change in Salt Marsh Quantity or Quality → Carbon Storage

Description of Relationship

Salt marshes accumulate and store carbon in vegetation and soils. Table 11, which reflects a global review of carbon accumulation rates in salt marsh soils, summarizes carbon accumulation for different regions of the coastal United States (Ouyang and Lee 2014).

Table 11. Carbon accumulation rates for salt marsh habitats in three U.S. regions

Region	C accumulation rate g m^{-2} yr^{-1} (±SE)
NW Atlantic (East Coast)	Mean: 172.2 (±18.1) (n=64 sites) Range: 21- 928
NE Pacific (West Coast)	Mean: 173.6 (±45.1) (n=8 sites) Range: 43-385
Gulf of Mexico	Mean: 293.7 (±60.9) (n=32 sites) Range: 18-1713

Source: Ouyang and Lee (2014).

Salt marsh soils also store carbon. In the first meter of salt marsh sediments, soil organic carbon averages 917 t CO_2e/ha (Murray et al. 2011).

Summary of Evidence

Table 1 in Ouyang and Lee (2014) provides a full list of carbon accumulation rates for 143 marsh sites. It may be helpful to examine the full table and choose a carbon accumulation rate on the basis of a site close to or similar to the site of interest. Salt marshes store carbon in three places: aboveground biomass, belowground biomass, and soils. It is understood that in a salt marsh system, soils contain the bulk of carbon and do not become carbon saturated because of vertical sediment accretion (Chmura et al. 2003; Mcleod et al. 2011; Ouyang and Lee 2014).

Salt marshes store the carbon that they accumulate. In the first meter of salt marsh sediments, soil organic carbon averages 917 t CO_2e/ha (Murray et al. 2011). Of total salt marsh carbon stocks, 95%–99% are stored in the soil, but when the carbon stored in living biomass is accounted for, salt marshes are known to store (on average) 949 t CO_2e/ha (Murray et al. 2011).

Tool: The InVEST Blue Carbon model can be used for more detailed, site-specific evaluations of blue carbon storage scenarios; however, it has relatively high data requirements. The model takes into account specific biotic and abiotic features of a salt marsh and estimates carbon sequestration, storage, and market value of stored carbon (if desired). This model will be especially useful if the user wants to compare alternative future scenarios based on development, land cover changes, and sea level rise. Model outputs include total carbon stock, carbon accumulation, carbon emissions, net carbon sequestration, and net present value of stored carbon. Data requirements include land cover maps for present and future scenarios, carbon pool initial values (for biomass, soil, and litter), and information on the accumulation rates, percent disturbance, and half-lives of carbon in the system. (Download the InVEST model here: http://natcap.wpengine.com/invest/.)

Other Factors

Climate Change: Simulations by Kirwan and Mudd (2012) suggest that the net impact of climate change will increase carbon burial rates in the near term, but as sea-level rise quickens, carbon burial rates might decrease over the long term. Other studies agree and note that if marshes stay above sea level that carbon accumulation rates may be maintained or may increase with increasing temperatures (Langley et al. 2009).

Salinity: Salinity changes soil properties, which can lead to different carbon accumulation rates. Carbon accumulation in marsh soil has been negatively correlated with salinity (Loomis and Craft 2014).

Strength of Evidence

High. It is widely accepted that salt marsh habitats accumulate and store carbon (Chmura et al. 2003; Mcleod et al. 2011; Ouyang and Lee 2014; Fennessy and Nahlik 2016). There are global reviews and meta-analyses that summarize carbon accumulation rates.

Predictability: Models such as InVEST can perform scenario analysis for current and future carbon storage. Generalized predictions can be made for this link due to the existence of robust review studies, such as the one by Ouyang and Lee (2014).

Sources

Chmura, Gail L., Shimon C. Anisfeld, Donald R. Cahoon, and James C. Lynch. 2003. "Global Carbon Sequestration in Tidal, Saline Wetland Soils." *Global Biogeochemical Cycles* 17 (4). http://onlinelibrary.wiley.com/doi/10.1029/2002GB001917/full.

Nahlik, Amanda, and Siobhan Fennessy. 2016. "Carbon Storage in US Wetlands." *Nature Communications* 7. https://www.nature.com/articles/ncomms13835.

Kirwan, Matthew L., and Simon M. Mudd. 2012. "Response of Salt-Marsh Carbon Accumulation to Climate Change." *Nature* 489 (7417): 550–553.

Langley, J. Adam, Karen L. McKee, Donald R. Cahoon, Julia A. Cherry, and J. Patrick Megonigal. 2009. "Elevated CO2 Stimulates Marsh Elevation Gain, Counterbalancing Sea-Level Rise." *Proceedings of the National Academy of Sciences* 106 (15): 6182–6186.

Loomis, Mark J., and Christopher B. Craft. 2010. "Carbon Sequestration and Nutrient (Nitrogen, Phosphorus) Accumulation in River-Dominated Tidal Marshes, Georgia, USA." *Soil Science Society of America Journal* 74 (3): 1028–1036.

Mcleod, Elizabeth, Gail L. Chmura, Steven Bouillon, Rodney Salm, Mats Björk, Carlos M. Duarte, Catherine E. Lovelock, William H. Schlesinger, and Brian R. Silliman. 2011. "A Blueprint for Blue Carbon: Toward an Improved Understanding of the Role of Vegetated Coastal Habitats in Sequestering CO2." *Frontiers in Ecology and the Environment* 9 (10): 552–560.

Murray, Brian C., Linwood Pendleton, W. Aaron Jenkins, and Samantha Sifleet. 2011. *Green Payments for Blue Carbon: Economic Incentives for Protecting Threatened Coastal Habitats.* NI R 11-04. Durham, NC: Duke University. https://nicholasinstitute.duke.edu/environment/publications/naturalresources/blue-carbon-report.

Natural Capital Project. n.d. "InVEST Coastal Blue Carbon." Natural Capital Project. http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/coastal_blue_carbon.html.

Ouyang, Xiaoguang, and S.Y. Lee. 2014. "Updated Estimates of Carbon Accumulation Rates in Coastal Marsh Sediments." *Biogeosciences* 5057.

Link 8a

8a: Change in Salt Marsh Quantity or Quality → Habitat Persistence

Description of Relationship

An increase in salt marsh area or quality will result in persistence of the salt marsh habitat. Protecting or improving salt marsh habitat will allow it to persist in a specific area. Areas of salt marsh contribute to the overall salt marsh habitat patch mosaic in a certain region/estuary.

Summary of Evidence

Without protection, salt marshes may be filled in and developed, especially in highly desirable coastal regions where waterfront property is very valuable. Salt marsh habitats have been targets for human development for hundreds of years (Gedan et al. 2009). A review by Gedan et al. (2009) describes in detail the multitude of threats that salt marshes face—threats that can be partially mediated by protection or restoration. These threats include resource exploitation and extraction, development, invasive species, hydrologic alterations, pollution, climate change, and sea-level rise.

Other Factors

The persistence of salt marsh at a particular site is not substitutable, but it can be for a particular region or estuary. If there are many salt marsh sites in an area, the persistence of one site does not mean as much for the persistence of that habitat type overall in the region.

Strength of Evidence

Fair. It is clear that salt marsh protection or restoration is needed for the existence/persistence of salt marsh habitats. These habitats face many threats, and restoration and protection will extend their existence—however, persistence over the long term is less clear.

Predictability: This link is highly variable at the site-specific level, depending on the threats the marsh was facing before protection or restoration as well as on the larger habitat mosaic of which the marsh is a part and the substitutability of salt marsh habitat sites. Continued threats to a salt marsh habitat even after protection or restoration (sea-level rise, invasives, pollution, and so on) must also be taken into account.

Sources

Gedan, K. Bromberg, B.R. Silliman, and M.D. Bertness. 2009. "Centuries of Human-Driven Change in Salt Marsh Ecosystems." *Annual Review of Marine Science* 1 (1): 117–41. doi:10.1146/annurev.marine.010908.163930.

Link 9a

9a: Change in Salt Marsh Quantity or Quality → Research and Education Value

Description of Relationship

Salt marsh habitats provide opportunities for research and outdoor education. Increased salt marsh area allows for a greater number of research studies as well as opportunities for experiential outdoor learning.

Summary of Evidence

This link will be highly dependent on nearby educational and research resources. For Natural Estuarine Research Reserve Sites (NERRS), it is likely that scientific and educational resources will be accessible.

Though it is possible to educate students about marshes in a classroom setting, experiential learning has been shown to provide additional benefits. Experiential learning is "learning from the real-world," "characterized by variability and uncertainty," and engages the student in an environment with high active involvement (Gentry 1990). Experiential learning has been linked to a deeper understanding of subject matter than understanding that occurs in typical classroom learning (Eyler 2009). Being at a salt marsh site while learning about it will therefore enhance a student's learning experience. NOAA already values the benefits of experiential learning by providing resources such as the Bay Watershed Education and Training (B-WET) program, which funds "locally relevant, authentic experiential learning for K-12 audiences." A review of the Chesapeake B-WET program showed linkages between student B-WET participation and environmental stewardship and literacy (NOAA n.d.).

Measuring or predicting the research or education benefits provided by salt marshes requires data on the number of scientific papers, reports, theses, educational programs on marshes or the number of students attending programs related to marshes. There will likely be a distinction between indicators and analysis used for measuring educational opportunities provided for K-12 students and research opportunities provided for college or graduate students and researchers.

Other Factors

If there are many substitute options for studying salt marshes or teaching about them, there may not be a strong link between the salt marsh at a site of interest and research or education. The link may still exist, but substitutability should be considered.

Strength of Evidence

None. Though many logical connections exist between salt marsh sites and research/education benefits, there is little to no published evidence that supports this generalized link. Site-specific data may be available at a local level to improve the evidence grade of this link for a particular site.

Sources
Eyler, Janet. 2009. "The Power of Experiential Education." *Liberal Education* 95 (4): 24–31.
Gentry, James W. 1990. "What Is Experiential Learning." *Guide to Business Gaming and Experiential Learning* 9: 20.
NOAA. n.d. "Bay Watershed Education and Training: Impacts." National Oceanic and Atmospheric Administration. http://www.noaa.gov/office-education/bwet/impacts.

Links 10a-l

10a: Dredging → Cost Savings
Description of Relationship
Dredging costs money. Dredging less frequently or dredging lower amounts of sediment will result in cost savings.

Summary of Evidence
Tool: The U.S. Army Corps of Engineers U.S. Waterway Data Dredging Information System contains information on dredging contracts. Examining contracts similar to those that would need to be completed in an area of interest can help determine previous dredging amounts, frequency, and costs. It is then possible to calculate avoided costs on the basis of updated dredging frequencies and estimated sediment levels.

Other Factors
None.

Strength of Evidence
High. It logically follows that a change in dredging amount or frequency will result in a change in cost of dredging.

Predictability: Site-specific data will be required to make this evidence useful, but using the U.S. Waterway Data Dredging Information System database in combination with site data should enable generation of estimated avoided dredging costs on the basis of sites similar to the one of interest. Some extrapolation of costs will be required.

Sources
U.S. Army Corps of Engineers. 2016. "U.S. Waterway Data: Dredging Information System." Navigation Data Center. http://www.navigationdatacenter.us/data/datadrg.htm.

10b: Health → Cost to Society (Disability-adjusted Life Years Lost)
Description of Relationship
Adverse physical health outcomes create a burden to society. The societal effect of adverse physical health outcomes is captured by the burden of disease, which can be quantified using several indicators. The use of one indicator, disability-adjusted life years (DALYs), is described below.

Summary of Evidence
The DALY metric of the environmental burden of disease is widely used by governmental and non-governmental organizations as a policy evaluation tool. Required data to calculate DALYs from a particular cause include population exposure to the cause, an exposure-response relationship for each outcome, an estimate of the proportion of disease that is caused by the source, an estimate of the prevalence of each outcome, and a disability weight value for each outcome (Theakston et al.. 2011). The equations used to calculate DALYs lost to adverse health outcomes is as follows:

$$\text{DALYs lost} = \text{years lived with disability (YLD)} + \text{years lost to premature mortality (YLL)}$$

$$\text{YLD} = \text{number of cases (I)} * \text{disability weight (DW)} * \text{average duration of disability, years (L)}$$

$$\text{YLL} = \text{number of deaths (N)} * \text{standard life expectancy at the age at which death occurred (L)}$$

Disability weights for a variety of outcomes have been assessed by the World Health Organization (see Table 12).

Table 12. Disability weights for physical health outcomes potentially affected by salt marsh habitats

Outcome	Disability weight[a]
Diarrheal diseases (episodes)	0.105
Poisoning (short term)	0.611 (age 0-14), 0.608 (age ≥ 14)
Respiratory (episodes)	0.279 (lower respiratory), 0 (upper respiratory)
Protein (energy, malnutrition)	0.053 (wasting), 0.002 (stunting), 0.024 (developmental disabilities)
Cancer	Depends on cancer type

Source: World Health Organization (2004).
[a] Disability weights are on a scale from 0 to 1, with 0 corresponding to a healthy person.

Other Factors
Exposure to causes of disease and availability of treatment can vary by location and can affect the number of cases or deaths and disability weights used to calculate DALYs.

Strength of Evidence
Moderate. The DALY is a widely used nonmonetary metric for the environmental burden of disease. Its accuracy in estimating the burden of a particular disease depends on the evidence available for the components needed to calculate DALY (see description of relationship and summary of evidence sections) for that disease.

Predictability: Prediction of DALY is standardized and well-documented.

Sources
Mathers, Colin. 2008. The Global Burden of Disease: 2004 Update. World Health Organization. https://books. google. com?hl=en&lr=&id=xrYYZ6Jcfv0C&oi=fnd&pg=PR5&dq=World+Health+Organization +(WHO).+2004.+Global+burden +of+disease+2004+update:+Disability+weights+for +diseases+and+conditions &ots=tbRA4f86Ao&sig=InJhPGEXafGf07XZR8Z6yG7TA9k. [See this link for shortcut to DALY tables: http://www.who.int/healthinfo/global_burden_disease/GBD2004_DisabilityWeights.pdf].

World Health Organization, and others. 2011. "Burden of Disease from Environmental Noise: Quantification of Healthy Life Years Lost in Europe." In *Burden of Disease from Environmental Noise: Quantification of Healthy Life Years Lost in Europe,* 126–126.

10c: Species Persistence → Existence Value (Species)
Description of Relationship
People (or households) are willing to pay $X for the existence of a certain species.

Summary of Evidence
Using contingent valuation (CV) (stated preference methods), it is possible to determine the value that a person or household is willing to pay (WTP) for the existence of a species. These survey methods ask people to imagine various hypothetical future states of a species or population and to specify how much they would be willing to pay to attain that state. Future states are often listed in terms of avoided loss, increasing population size, improving the species' status, or increasing chances of survival (Wallmo and Lew 2011). A good example of stated preference methods used to estimate WTP values for marine species can be seen in Wallmo and Lew (2011). This paper reports the methods used to estimate WTP values for Chinook salmon, Hawaiian monk seals, and the Smalltooth sawfish. Unique studies must be found or conducted for each species of interest.

Other Factors
Valuation Question: The way that a CV question is structured can influence the price that a person is willing to pay for species existence. Variables such as expected change in threat level or size of a species' population, payment frequency, and mode of the survey have been shown to influence willingness to pay (Richardson and Loomis 2009).
Species Characteristics: Species characteristics have been found to influence how much people are willing to pay for a

species' existence. People are often willing to pay more for charismatic megafauna. A species' taxa has also been linked to WTP price (Richardson and Loomis 2009).

Substitutability: Existence of a species at a certain site may be influenced by the substitutability of that site. Does the species exist only at a few sites or at many? Willingness to pay for existence in a certain place can change depending on perception of how important that location is to the species' survival overall.

Survey Population: Existence value is quantified using survey methods. However, when examining WTP studies and interpreting WTP prices, the population surveyed should be considered carefully. What are the population's primary concerns? Were underrepresented communities included? Are there other confounding factors?

Strength of Evidence

Fair. Multiple sources provide evidence to show that people are willing to pay for the existence of certain wildlife species. WTP will depend on the species and the community that places (or doesn't place) value on it.

Predictability: There is little available evidence that generalizes the numerical relationship between persistence of a species and existence value of the species because each species has a unique value to people. It is possible that a WTP study on a species of interest (or a similar species) has been done and that a benefit transfer approach could be used to approximate willingness to pay at a different site. WTP surveys are used extensively, and although they have limitations, they are considered to be the primary method for developing monetary values for species existence.

Example. Studies such as the one by Wallmo and Lew (2011) will have to be performed for each wildlife species of interest at a specific location to determine the most accurate WTP value. Extrapolations are possible (using a benefit transfer function); however, they should be used with caution.

Sources

Richardson, Leslie, and John Loomis. 2009. "The Total Economic Value of Threatened, Endangered and Rare Species: An Updated Meta-Analysis." *Ecological Economics* 68 (5): 1535–1548.
Wallmo, Kristy, and Daniel K. Lew. 2011. "Valuing Improvements to Threatened and Endangered Marine Species: An Application of Stated Preference Choice Experiments." *Journal of Environmental Management* 92 (7): 1793–1801.

10d: Commercial Fishing → Value of Fish Caught

Description of Relationship

Fish harvested commercially have economic value and can be sold on the market.

Summary of Evidence

The National Marine Fisheries Service (NMFS) commercial fish landings database contains total revenue data for fish species caught in the United States (http://www.st.nmfs.noaa.gov/commercial-fisheries/commercial-landings/annual-landings/index). Data can be organized by species and geography, and they can be used to estimate monetary value of fish caught in a certain area. Local governments or other fishery-related organizations may host more specific data for a region of interest.

Tool: The InVEST Fisheries model can estimate harvest volume of single-species fisheries. Model outputs include economic value of fish harvest. See a more detailed description of the model in the entry for link 2a,d. (Download the model here: https://www.naturalcapitalproject.org/invest/.)

Other Factors

The NMFS data include only dockside value. In many/most fisheries, additional value chains are associated with processing and distribution (i.e., crab cakes from blue crabs, shucking oysters, filleting finfish, and dog food from offal). The IMPLAN model (http://www.implan.com/) accounts for some of this added value by incorporating landside fishery businesses, but it is often considered only a first step in incorporating added value.

Strength of Evidence

High. Because fish have a market value, it is relatively easy to link the amount of fish caught commercially to a monetary value on the basis of market prices.

Predictability: Predictability of the value of fish caught will depend on access to up-to-date market data. If additional values (further along the value chain or related to local businesses) are included, additional site-specific data will be needed in order to make predictions.

Sources

Natural Capital Project. n.d. "InVEST Fisheries." Natural Capital Project. http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/fisheries.html.

NOAA National Marine Fisheries Service. 2015. "Commercial Fisheries Statistics." NOAA Office of Science and Technology. http://www.st.nmfs.noaa.gov/commercial-fisheries/commercial-landings/annual-landings/index.

10e: Aesthetics → Property Values

Description of Relationship

Aesthetics provided by the salt marsh will increase property values. There is a price premium for properties that have views of or proximity to a salt marsh.

Summary of Evidence

No evidence was found that directly links property values to salt marsh habitat. However, there are multiple cases of the hedonic method being used to attribute property price premiums to natural features such as lakes or trees. A good summary of the hedonic method is provided in Lansford and Jones (1995), but in simple terms this method uses a statistical model to develop a relationship between housing price data and property features, such as distance to waterbody, waterfront access, and percent tree canopy. It is then possible to estimate how housing prices might respond to variations in those property features. Using the hedonic method, it would be possible to estimate house price premiums supplied by aesthetic features of a salt marsh. Examples of research using the hedonic method can be found for estimating the influence of aesthetics due to lake and river views (Kulshreshtha and Gillies 1993; Lansford and Jones 1995) and to greenways (Nicholls and Crompton 2005).

Other Factors

In some cases, the property features used in hedonic pricing models do not necessarily isolate the influence of aesthetics on property values. For example, Lansford and Jones (1995) examined features such as lakefront access and proximity to lake; however, it is hard to distinguish whether price premiums associated with those variables are due solely to aesthetics or whether, perhaps, they integrate recreational use as well. Similarly, the study by Nicholls and Cropton (2005) on greenways uses variables such as proximity to greenway, but it is unclear whether it is the beauty of the greenway or the recreational opportunities it provides (or both) that are contributing to the price premium.

Strength of Evidence

Low. Multiple sources note the link between natural features and property price premiums. However, no evidence was found to directly support the link between salt marsh aesthetics and property prices. It is understood that property values are linked to natural features, and extrapolations between other natural features and salt marshes could be made.

Predictability: Example studies are available that demonstrate methods that could be used to examine the relationship between salt marshes and property value. But it is difficult to quantify and isolate aesthetic value, even when doing a study to examine this service.

Sources

Kulshreshtha, Suren N., and Jon A. Gillies. 1993. "Economic Evaluation of Aesthetic Amenities: A Case Study of River View." *Journal of the American Water Resources Association* 29(2): 257–266.

Lansford Jr, Node H., and Lonnie L. Jones. 1995. "Recreational and Aesthetic Value of Water Using Hedonic Price Analysis." *Journal of Agricultural and Resource Economics*, 341–355.

Nicholls, Sarah, and John L. Crompton. 2005. "The Impact of Greenways on Property Values: Evidence from Austin, Texas." *Journal of Leisure Research* 37(3): 321.

10f: Shoreline Protection → Cost of Potential Damage

Description of Relationship

Coastal erosion and flooding result in economic costs. It is possible to measure the economic costs of coastal damage (erosion and flooding). These costs include damage to structures and infrastructure, loss of business, and loss of tourism.

Each area will have unique attributes that lead to differing costs of coastal damage. Local real estate and property value data as well as data on infrastructure and public facilities costs are needed to estimate the value of structures/infrastructure at high risk for coastal flooding, erosion, or both.

Studies of economic damages caused by previous flooding events can help estimate potential costs of future events. For example, the Department of Commerce report *Economic Impact of Hurricane Sandy* details the varying long-term economic costs of hurricane damage (Henry et al. 2013). Costs include loss of business, loss of tourism revenue, and limits on travel and transport. These are the types of costs that must be considered when scoping the full economic costs of coastal damage.

A meta-analysis of 34 hurricanes in the United States found that coastal wetlands reduce the damaging economic impacts of these storms (Costanza et al. 2008). Creating a regression model using hurricane damages per unit of GDP in the hurricane swath, wind speed, and wetland area in the hurricane swath resulted in predictions of the value of coastal wetlands in reducing hurricane damage. The model predicts that losing 1ha of wetland corresponds to an average of $33,000 additional damage from a specific storm (data on marginal values of additional hectares of wetland for each individual hurricane are also provided in Table 2 of Costanza et al. 2008). When using the model to map the value of coastal wetlands in the United States at the km^2 scale, it was found that 1km^2 of wetland ranged in value from $250/ha/year to $51,000/ha/year for storm protection (average value = $3,230/ha/year). Notably, this study includes all coastal wetlands on the U.S. East Coast and does not distinguish among habitats like salt marsh and mangroves.

Other Factors
None.

Strength of Evidence

Moderate. It is clear that coastal damages have a cost, so the relationship between coastal protection and damage costs does exist.

Predictability: Some generalizable evidence regarding the economic costs of coastal damage and the part that salt marshes have to play in those costs is available (Costanza et al. 2008). Finding the economic value of vulnerable houses, structures, and infrastructure will depend on availability of local data. Predicting other long-term economic costs related to storm damage is more difficult, and the evidence for that aspect of this link is weakest.

Sources

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10g: Recreation → Recreational Value

Description of Relationship

People (or households) are willing to pay $X for recreational opportunities provided by salt marsh habitats.

Summary of Evidence

Using surveys, it is possible to estimate people's willingness to pay for recreational resources associated with coastal habitats. These surveys are most useful when they are conducted specifically for the habitat, site, or region of interest. A good example of this type of survey can be found in Leeworthy et al. (2017), a National Marine Sanctuaries report that details the process of valuing recreational resources on the Washington coast. The study estimates households' willingness to pay for recreational activities such as wildlife watching, tide pool exploration and access, beach use, and scenic viewing.

The travel cost method is another common way to put a monetary value on recreation. This method examines the resources (gas, time, plane tickets, and so on) that people use to travel to a recreational site and places a monetary value on

those resources to estimate how much people are willing to pay to recreate there. Examples of the travel cost method used in the marine recreation context can be found in Milon (1988), Park et al. (2002), and Carr and Mendelsohn (2003).

Existing datasets on tourism and recreation can be useful for making generalizations about tourism use and values. A report completed for NOAA in 2013 details these datasets and may be useful when determining data needs for valuing tourism resources (ERG 2013).

Tool: The USGS Benefit Transfer Toolkit provides several options for estimating economic value for a variety of recreational activities within the United States, using a database of more than 2,000 individual nonmarket valuation estimates. The most useful tools are the meta-regression calculators for values of fishing, hunting, trail use, and wildlife viewing, which take into account the region, type of wildlife species involved, and land ownership type. These regression calculators can be used to estimate the value of a day of recreation in a certain location. All values are in 2014 dollars.

For recreation types for which no meta-regression is available, benefit transfer (using results from a primary study at one site to estimate benefits at another site) may be the best available alternate approach. But users need to be aware that it can introduce significant uncertainty and that using meta-regressions (described above) or developing site- and context-specific estimates is a better approach if this valuation is an important component in the decision (Wainger et al. 2016). If a study exists within the USGS Nonmarket Valuation Database that closely matches the recreation type and site being evaluated, it can be used for a point estimate benefit transfer. Average values for studies in a particular geographic region can also be used for a benefit transfer if no individual study is a good match. Table 13 lists average values for several types of relevant recreational activities in different regions of the United States. Importantly, many of the studies included in this database were not conducted specifically for coastal recreational resources—a fact that should be taken into consideration and acknowledged if doing a benefit transfer.

Table 13. Average values (value per person, per day) for recreational activities in U.S. regions

Activity	Alaska	Pacific Coast	Southeast	Northeast
Hiking	$159.61	$50.30	$100.82	$59.49
Boating (motorized/ non-motorized)	$433.36/---	$21.97/---	$23.02/ $85.73	$100.07/ $17.83
Fishing (saltwater)	$224.61	$141.15	$115.77	$62.36
Hunting (waterfowl)	---	$53.16	$67.91	$39.60
Beach use	---	$56.42	$75.83	$35.49
Wildlife viewing	$83.05	$94.02	$60.66	$61.84
Diving	---	$11.05	$163.83	---
Swimming	---	$31.30	$14.18	$27.64

Source: USGS Benefit Transfer Toolkit.

Tool: The GecoServ database, hosted by the Harte Research Institute, provides a collection of valuation studies relevant to the Gulf of Mexico. The valuation database can be searched by ecosystem service type as well as by habitat type. In this case, search selections could be limited to the intersection of studies about "recreation" and "saltwater wetlands" to provide relevant studies for recreation values in salt marsh areas. As of September 2017, 52 results using those search criteria had emerged.

The database does include studies that were done outside of the Gulf of Mexico region, and each search result dataset includes an economic value (in 2012 USD), units, region of study, the method used to determine an economic value, and a citation for the original study.

Other Factors

Attributes and preferences of the sample population surveyed can influence the results of a WTP survey for recreational use. Leeworthy et al. (2017) note that survey respondents in their study were limited to those who already visit the coast for recreation; however, other groups who do not visit may hold other values for coastal recreation. This study also examined respondents' ecological world view and income to examine if these attributes affect willingness to pay for coastal recreation.

Moderate. Many individual valuation studies for recreation have been conducted across the United States and Canada; these studies are captured in the USGS Benefit Transfer Toolkit database and are used to calculate average regional values and to create meta-regressions for valuation of certain recreational activities. The database appears to be fairly complete, but it is unclear how recently it has been updated; some more recent valuation studies may not be included. The strength of evidence should ultimately be determined by how the toolkit is used, because application of meta-regressions is much better than use of average and point estimates. Users should fully consider that the average values from the USGS Benefit Transfer Toolkit were calculated using studies that were not necessarily done with coastal recreational resources in mind, a fact that may affect valuation. GecoServ studies are relevant to coastal recreational resources; however, benefit transfer (which introduces uncertainty and error) is the only way to utilize the data presented in that database.

Predictability: WTP surveys and the travel cost method are used extensively, and although these methods do have limitations, they are considered to be the primary methods for developing monetary values for recreation. Studies such as the ones by Leeworthy et al. (2017) or Carr and Mendelsohn (2003) would ideally be performed at your specific location in order to develop the most accurate WTP value. Extrapolations are possible (using a benefit-transfer function); however, this type of extrapolation should be used with caution.

Sources

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10h: Carbon Storage → Social Cost of Carbon

Description of Relationship

Changes in carbon stored by salt marsh habitat correspond to avoided economic costs resulting from climate change or other disturbances, and these avoided costs can be represented (i.e., monetized) by the social cost of carbon (SCC). The social cost of carbon represents long-term damages to society from the release of 1 ton of carbon dioxide equivalent (CO_2e). Each additional ton of carbon dioxide equivalent represents $42 (in 2007 USD) of avoided damages to society, according to the SCC estimate for 2020, using a 3% discount rate.

Summary of Evidence

The social cost of carbon is the calculated total economic cost of an additional ton of carbon dioxide released into the atmosphere. This cost incorporates many different impacts of climate change. To account for these diverse impacts, the Interagency Working Group (IWG) on the Social Cost of Carbon (2016) bases its estimates on the DICE, PAGE, and FUND integrated assessment models (IAMs). The social cost of carbon for five-year intervals up to 2050 based on different discount rates are displayed in Table 14 (IWG 2016).

Table 14. Social cost (2007 $) of emitting 1 metric ton of CO_2, by year of emission and discount rate

Year	Discount Rate		
	5%	3%	2.5%
2015	$11	$36	$56
2020	$12	$42	$62
2025	$14	$46	$68
2030	$16	$50	$73
2035	$18	$55	$78
2040	$21	$60	$84
2045	$23	$64	$89
2050	$26	$69	$95

Source: IWG (2016).

Important note: the current U.S. presidential administration has dissolved the Interagency Working Group on the Social Cost of Carbon, and although the group's reports are still available, the administration has made it clear that SCC values provided in the 2016 report are not meant to be used for federal policy making or for cost-benefit analysis at this time.

Other Factors

Model Uncertainty: Data gaps and uncertainty about predicting a wide variety of climate change outcomes make predicting the social cost of carbon inherently difficult. Section IV of the IWG report details these uncertainties (IWG 2016). The integrated assessment models used to calculate that cost have omitted certain negative impacts of climate change, and estimates reported by the models should be considered low estimates of the "true" economic cost of carbon (Institute for Policy Integrity 2017).

Discount Rate: Different discount rates allow for the weighting of future damages, with higher discount rates putting lower weights on damages relative to lower discount rates. Depending on the discount rate chosen, there could be up to a five-fold difference in the social cost of carbon, as seen in the Table 12.

Strength of Evidence

Fair. Though the concept of the social cost of carbon is widely acknowledged, there are active discussions in the literature about what this cost should be and which discount rate should be used (Watkiss and Downing 2008; Tol 2011; Ackerman and Stanton 2012; Johnson and Hope 2012; Greenstone et al. 2013). Though the social cost of carbon may not perfectly reflect the "true" economic cost of a carbon dioxide equivalent in the atmosphere, the IWG's SCC estimates represent current estimations that have been used in cost-benefit analyses and for policy making in the United States. The value of the social cost of carbon will likely evolve in the future. The IWG's current SCC estimates are based on generally accepted integrated assessment models, but the specific cost projections vary among the models, and some of the assumptions and techniques used in these estimates have been identified as inaccurate or in need of improvement (NASEM 2017).

Predictability: If you accept and use a certain SCC value and discount rate is accepted and used, predicting the estimated cost savings related to reduced CO_2 emissions is simple. However, predicting whether that value truly represents the cost of 1 ton of CO_2e to society is another matter and is under debate in the literature.

Sources

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10h: Carbon Storage → Value of Carbon Credits

Description of Relationship

In theory, carbon stored by salt marsh habitat can be turned into commodities as carbon credits and sold on the voluntary carbon market. The average price for credits sold on that market in 2016 was $3.0/tonne CO_2e (Hamrick and Gallant 2017).

Summary of Evidence

Carbon credits created by reducing emissions or sequestering carbon at wetland sites have been added to the project types available for listing on the voluntary carbon market (RAE 2016). Forest Trends' Ecosystem Marketplace conducts yearly evaluations of the voluntary carbon market; its most recent report listed the average price of a carbon credit sold as $3.0/tonne of CO_2e. However, the most relevant carbon credit prices are forestry and land-use, REDD+, afforestation/reforestation, improved forest management, and grassland/rangeland management credits, which traded at $5.1, $4.2, $8.1, $9.5, and $6.9, respectively (Hamrick and Gallant 2017). As of February 2018, avoided emissions from salt marshes and carbon credits from coastal marine ecosystems in general had not been traded on the voluntary carbon market at scale.

Almost all offsets sold on the voluntary market were verified by a third-party standard, most often the Verified Carbon Standard (Hamrick and Gallant 2017). Multiple organizations have released guidelines and standards for creating and monitoring wetland carbon credits, as shown in Table 15.

Table 15. Guidelines and standards for creating and monitoring wetland-based carbon credits

Organization: Guideline title	Notes
Verified Carbon Standard: Methodology for Tidal Wetland and Seagrass Restoration	This methodology outlines steps to quantify net greenhouse gas emission reductions and removals resulting from project activities implemented to restore tidal wetlands.
Verified Carbon Standard: Methodology for Coastal Wetland Creation	This methodology outlines steps to quantify the greenhouse gas benefits of wetland creation activities.
Restoring America's Estuaries: A Manual for Using the VCS Methodology for Tidal Wetland and Seagrass Restoration	This manual provides guidance on use of the Verified Carbon Standard methodology.
Commission for Environmental Cooperation: Greenhouse Gas Offset Methodology Criteria for Tidal Wetland Conservation	This methodology is meant to supplement the Verified Carbon Standard methodology.
American Carbon Registry: Restoration of Degraded Wetlands of the Mississippi Delta	This methodology is specific to wetlands of the Mississippi Delta.
American Carbon Registry: Restoration of California Deltaic and Coastal Wetlands	This methodology is specific to wetlands of California.

Other Factors

Price Uncertainty: Prices for carbon credits are variable and change from year to year, so stable predictions of the value of carbon credits are unlikely (Hamrick and Gallant 2017).

Oversupply: Though demand for carbon credits has been increasing, credit suppliers reported that there were many unsold credits available. Creating carbon credits does not guarantee that the credits will be bought on the market (Hamrick and Gallant 2017).

Buyer Preferences: Almost all carbon credit buyers consider standard use (such as the Verified Carbon Standard) to be an essential prerequisite for purchasing a credit. Buyers are often searching for a credit that fits their organization's mission. This fit is determined by factors such as prices, project location, and additional co-benefits of the project. Buyers tend to want the credit-creating project to be in a location near the buyer's suppliers, operations, headquarters, or customers. In terms of desired co-benefits, buyers are most often interested in biodiversity or community benefits. North America and Europe represent the regions from which most buyers originate, and U.S. buyers tend to prefer local projects (Hamrick and Goldstein 2016).

Strength of Evidence

Fair. The creation of a carbon credit does not guarantee that it will be sold on the voluntary market. The average carbon credit price is variable, and changes in market dynamics could alter price trends. Carbon credits from coastal ecosystems are not currently being sold at a large scale on the voluntary carbon market.

Predictability: Until carbon markets and carbon crediting systems become more established and standardized, predictability for this link will be low due to uncertainty and market fluctuations.

Sources

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10i: Habitat Persistence → Existence Value (Habitat)

Description of Relationship

People (or households) would be willing to pay $X to maintain the existence of salt marsh habitat.

Summary of Evidence

A study from the Peconic Estuary System in New York state completed a WTP survey of residents and tourists about the existence of salt marsh habitats in the area (Johnston et al. 2002). However, it should be noted that because residents and

tourists were surveyed, the respondents could be expected to use or enjoy the resources, so the survey does not completely isolate willingness to pay for *existence* of the marsh even though that is what the survey asked about. The study found the annual willingness to pay for salt marsh existence is $0.066/acre/household/year (Johnston et al. 2002). This study was performed in 1996, so any interpretation of this value should reflect proper inflation rates.

It is hard to identify a true WTP value for all salt marshes on the basis of one study, though benefit-transfer methods are available to translate that value to other locations. Experts in environmental valuation are cautious with benefit transfer, and they note the benefit of doing benefit-transfer meta-analysis to get the best possible transfer function (Johnston et al. 2005; Wilson and Hoehn 2006). No meta-analyses of willingness to pay for salt marsh existence were found.

Tool: The GecoServ database, hosted by the Harte Research Institute, provides a collection of valuation studies relevant to the Gulf of Mexico. The valuation database can be searched by ecosystem service type as well as by habitat type. In this case, search selections could be limited to the intersection of studies about "habitat" and "saltwater wetlands" to provide relevant studies for habitat values of salt marsh areas. As of September 2017, 32 results using those search criteria had emerged.

The database does include studies that were done outside of the Gulf of Mexico region, and each search result dataset includes an economic value (in 2012 USD), units, region of study, the method used to determine an economic value, and a citation for the original study.

Other Factors
Existence value is quantified using survey methods. However, when examining WTP studies and interpreting WTP prices, the population surveyed should be considered carefully. What are the population's primary concerns? Were underrepresented communities included? Are there other confounding factors?

Strength of Evidence
Fair. Multiple studies have been performed to establish that there is a connection between salt marsh habitat persistence and existence value, but some studies do not completely isolate for *existence* of the marsh (rather than utility).

Predictability: There is little evidence available that generalizes the numerical relationship between salt marsh persistence and existence value of salt marsh habitat. Multiple values for salt marsh habitat are available through the GecoServ database, but limitations of these studies and their methods should be considered if they are used for benefit transfer. Extrapolations are possible (using a benefit-transfer function); however, this type of extrapolation should be performed with caution.

Example. WTP surveys are used extensively, and although they do have limitations, they are considered to be the primary method for developing monetary values for existence. One of the major limitations of the surveys is the difficulty of isolating the value of habitat existence. Studies such as the one by Johnston et al. (2002) would ideally be performed at a specific location of interest in order to develop the most accurate WTP value.

Sources
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10j: Salt Marsh Restoration →Local Jobs and Income
Description of Relationship
Restoration projects create jobs.

Summary of Evidence

Many restoration projects require relatively extensive site construction activities, creating jobs and income. Whether the jobs and income will be local depends on availability of expertise in the region to perform restoration activities. Each project will be different, so it is hard to generalize the number of jobs created by a specific restoration project, though previous studies and reports have documented economic activity generated by restoration work. For example, the U.S. Fish and Wildlife Service (FWS) released a 2014 report showing that for each million dollars spent on coastal restoration projects, between 5 and 55 jobs were created (depending on the state) and that FWS coastal restoration programs alone generated more than $15 million in income in 2011 (Laughland et al. 2014). Another study examined the economic impact of the American Recovery and Reinvestment Act of 2009, which funded NOAA coastal restoration projects across the United States. The study shows that in 1.5 years, 50 restoration projects were funded, creating 1,409 jobs and, on average, 17 jobs per million dollars spent (Edwards et al. 2013).

Other Factors

Job longevity is another factor. Construction will take place at the beginning of a restoration project and will last for a few months to a few years, depending on the size and complexity of the project. After initial restoration construction is completed, the number of construction jobs supported directly by a salt marsh is limited. Intermittent management (i.e., adding dredged material to a site, removing invasives etc.) may require labor; however this management will likely not require full-time employment. After its establishment, a marsh might indirectly create jobs by supporting local fisheries and coastal tourism (Edwards et al. 2013; see links 10k and 10l).

Strength of Evidence

Moderate: Restoration requires labor, which will create jobs—this is clear. The number of jobs created and the amount of income generated is less clear and will depend on the scale and complexity of the restoration project. In addition, there is no guarantee that the jobs created will be local.

Predictability: Though the number of jobs created will depend wholly on the project size and the type of restoration required, it is possible to predict this number by comparing a new restoration project to previous restoration projects. Restoration professionals can provide a general sense of the number of jobs that a project is likely to create.

Sources

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10k: Commercial Fishing →Local Jobs and Income
Description of Relationship
Commercial fishing creates jobs.

Summary of Evidence

It has been predicted that fisheries supported by salt marsh restoration could lead to job creation in the commercial fisheries sector (Edwards et al. 2013). NOAA reports that commercial fisheries supported 1.18 million jobs and generated $39.7 billion dollars in income in the United States in 2015 (NMFS 2017). The number of fisheries jobs supported by a single restoration project will depend on how much wildlife populations change due to nursery habitat provided for commercial species. Multiple restoration projects would likely have a more noticeable impact on job creation than single projects in the fisheries industry, but one large project could have a measurable job outcome.

Tool: The NOAA ENOW Explorer allows the user to explore the wages and number of jobs created by commercial fishing in each coastal U.S. county. It should be noted that data on commercial fishing jobs are combined with data on jobs related to fish hatcheries, aquaculture, seafood processing, and seafood markets. Find the tool here: https://coast.noaa.gov/digitalcoast/tools/enow.html.

Resource: The National Marine Fisheries Service (2017) provides data tables that break down jobs in the fisheries sector for each U.S. state.

Other Factors

Job location is another factor. Jobs created in the fisheries industry might not be local. If the commercial species supported by the nursery habitat of the marsh is migratory or wide ranging, the locations where these species are caught (and therefore the location of the jobs created) might not be close to the site of the restoration project.

Strength of Evidence

High. Clearly, commercial fishing supports jobs, and an increase in commercial fishing would support more jobs. Although the evidence for this particular link is high, it is unclear whether a single salt marsh restoration project will increase fisheries stocks enough to increase local jobs in the commercial fishing sector.

Predictability: If available, an estimate of changes in fish landings could be translated into a number of additional jobs provided by the commercial fishing industry. The number of jobs created would likely depend on the type of fishing and effort required, the species being harvested, and the scale of the fishing operation.

Sources

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10l: Recreation → Local Jobs and Income

Description of Relationship

Recreation opportunities support jobs.

Summary of Evidence

Jobs in the recreation industry include tourist guides, tour operators, gear rental providers, and service industry jobs supported by increased visitation by tourists and locals who are recreating. NOAA reports that 2.3 million people were employed in the ocean tourism and recreation sector in the United States in 2015 and that on average the sector was generating $24,000 per year in wages for each employee (NOAA OCM 2018).

Tool: The NOAA ENOW Explorer allows users to explore the number of jobs created by the tourism and recreation industry in each coastal U.S. county. Data on recreation and tourism jobs include data on jobs in eating and drinking establishments, hotels, marinas, boat dealers and charters, campsites and RV parks, scenic water tours, manufacture of sporting goods, amusement and recreation services, recreational fishing, zoos, and aquariums. Find the tool here: https://coast.noaa.gov/digitalcoast/tools/enow.html.

Other Factors

Seasonality: Many jobs associated with recreation are seasonal; these jobs are not supported year round (NOAA OCM 2018).

Recreation Type: Recreation type will determine how many jobs can be supported. For example, an increase in recreational fishing might create jobs by supporting fishing gear rentals and fishing charters, whereas the creation of a boardwalk to attract walkers and runners might not directly create new jobs in the recreation industry.

Strength of Evidence

High. It is clear that recreation and tourism supports jobs, and that an increase in people recreating would support more jobs. While the evidence for this particular link is high, it is important to emphasize that it is not clear whether a single salt marsh restoration project will increase recreation in ways that would increase local jobs in this sector.

Predictability: If available, an estimate of changes in visitors (number of people recreating) could be translated into the number of additional jobs provided by the recreation and tourism sector. That number would likely depend on the type of recreation available.

Source

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National Ecosystem Services Partnership

The National Ecosystem Services Partnership (NESP) engages both public and private individuals and organizations to enhance collaboration within the ecosystem services community and to strengthen coordination of policy and market implementation and research at the national level. The partnership is an initiative of Duke University's Nicholas Institute for Environmental Policy Solutions and was developed with support from the U.S. Environmental Protection Agency and with donations of expertise and time from many public and private institutions. The partnership is led by Lydia Olander, director of the Ecosystem Services Program at the Nicholas Institute, and draws on the expertise of federal agency sta , academics, NGO leaders, and ecosystem services management practitioners.

Conceptual Model Series

The NESP Conceptual Model Series provides a collection of resources explaining why ecosystem services conceptual models (ESCMs) are useful for decision making, providing guidance for building ESCMs, and describing NESP's initial efforts to standardize and apply these models with federal agency partners. It includes application examples of ESCMs and associated evidence libraries. The series aims to provide practical guidance for those who wish to apply ESCMs as a tool for incorporating ecosystem services considerations into their decisions.

NESP Conceptual Model Series Publications:
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