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Managing estuaries for ecosystem function

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Abstract

Estuary management is limited by lack of consensus on operational tools for handling multiple conflicting management objectives. One critical step to this goal is a shift from individual problems to a focus on maintaining ecosystem functions that benefit humans. If function is maintained, then the ecosystem is said to be functionally equivalent to its unimpacted state, which is sufficient for management. We propose an adaptation of a functional equivalency (FE) assessment approach from marine fishery management and use a case study demonstration to address how this approach can be integrated into existing ecosystem assessment tools. The functional equivalency framework has three components for implementation: definition of target ecosystem functions, measurable metrics of ecosystem functions, and policy-based thresholds for each metric that indicate when functional equivalency is lost and must be restored. Each case study is an application of available data, models, and management policy to define these ecosystem function components. We intend to foster discussion and future work on integrating the FE approach into existing ecosystem assessment tools. Data requirements are high, as is the necessary integration between science and policy. The results can be a more integrated management approach focused on maintenance of ecosystem functions most beneficial to humans.

Keywords

Ecosystem services; Assessment; Modeling; Habitat; Nutrients; Production

1. Introduction

1.1. Background

Resource management at the level of ecosystems has long been a goal for promoting sustainable human use of multiple ecosystem resources (Hein, 2010; O’Higgins et al., 2014).

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Declaration of competing interest
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Appendix A. Supplementary data
Supplementary data to this article can be found online at https://doi.org/10.1016/j.gecco.2019.e00892.

Yet, this goal remains elusive due largely to the complexity of ecosystems and the lack of consensus on an operational approach that effectively connects policy objectives to the available scientific information on ecosystem dynamics. Nowhere is this issue more apparent than in coastal estuaries where disparate and often conflicting management issues, such as nutrient management, habitat alteration, and fishery production, must be considered (Yanez-Arancibia et al., 2013). In most cases, these are managed as independent issues with the goal of reversing long-term degradation (Greening and Janicki, 2006; Lewis et al., 2016). The interrelationships among these issues have been evaluated as a part of integrated management approaches (Batiuk et al., 2009; Harvey et al., 2017), but this effort has not yet reached its full potential (EPA, 2004; Fulton et al., 2011), and one of the primary reasons for this is a need to transition from a paradigm of solving problems to one of achieving and maintaining functional targets that measure how ecosystems benefit people. Our goal here is to consider how to operationalize ecosystem functional targets, based on an existing assessment framework currently used in marine fishery management, and ask whether this framework can help shift management towards a more integrated ecosystem approach to decision making.

Ecosystem-based management (EBM) does not imply individual ecosystems are the management unit, rather it acknowledges that any management decision (e.g., nutrient loading limits) will have impacts on the entire ecosystem. In an EBM framework, management objectives and assessment metrics directly address this reality by seeking trade-offs among multiple parts of the ecosystem with the broad goal of maximizing ecosystem services (Harvey et al., 2017; Weijerman et al., 2018). Ecosystem-based management objectives are typically set based on one of three principles (Table 1). The first and most common is the use of historical reference points (Fulford et al., 2007; Greening and Janicki, 2006; Wilberg et al., 2013), which are easy to understand, but the realized benefits and achievability of historic targets can be secondary considerations (EPA, 2000). The second type of EBM objective is 'no/reduced harm' standard that seeks to re-establish a more pristine condition of the system and are usually based on important relationships, such as between nutrient loads and eutrophication (Greening and Janicki, 2006) or incidence of hypoxia (Breitburg et al., 2009), and water clarity or chl-*a* concentration (EPA, 2010a; Janicki and Wade, 1996). No harm targets are more closely aligned with the present state of the managed system than historic targets, but there remains a gap between the goal of reducing apparent harm and the desired outcome of maintaining ecosystem services.

The third type of EBM objectives are functional management objectives that are designed to maintain a set of conditions defined as optimal to preserve a desired ecosystem service (e.g., sustainable harvest). We will refer to such a managed ecosystem service as an 'ecosystem function' to clearly delineate it as a management target. For example, in marine fishery management, sustaining the ecosystem function (fishery harvest) has overtaken the concept of 'no harm' to the harvested population as the standard for management action (NOAA, 2007). Allowable harvest is based on the population replacement rate adjusted for reasonable uncertainty about future events (Methot, 2009). In a sense, a reasonable amount of harm is allowed as a necessary part of fishery exploitation, the managed system is said to be functionally equivalent to an unfished ecosystem if the harvest is sustainable. This management objective is not based on equivalency of the entire ecosystem. The concept of 'functional equivalency' (FE) is a compromise of sorts that acknowledges the ecosystem has changed and management goals based on achieving a previous state of ecosystem health, while desired, are not practical due to the dynamic nature of ecosystems. We then replace the objective of a 'previous state' with the objective of an 'equivalent state' in defining decision rules. The FE approach is then based on managing the ecosystem to preserve a specific set of functional endpoints while allowing for general change in the ecosystem. If the functional endpoints are achieved then the system is said to be equivalent, which is the policy-based objective. The FE concept has been explored in evaluating habitat restoration (Kentula, 2000; La Peyre et al., 2007), evaluating the health of aquatic food webs (Kang et al., 2008; Llewellyn and La Peyre, 2011), as well as marine fishery management. The FE approach has two equally important parts: a measure of function and a threshold beyond which functional equivalency is said to be lost. Together, these open the management question to more integration between scientific information and policy in that science is directly involved in both the resource assessment, as well as defining management thresholds that preserve the desired ecosystem function.

A focus on a desired ecosystem function, as well as the development of management thresholds based on functional equivalency, has clear advantages over both the historical and the no harm models described above; not because this approach is radically different, rather because both the decision rules and the assessment of ecosystem state are linked directly to the desired outcome of maintaining ecosystem services. The function is tied to ecosystem services and therefore the endpoint of management is maintenance of human benefit derived from the ecosystem and this is far easier to justify to stakeholders. Further, the use of an equivalency threshold ensures that multiple aspects of the ecosystem are taken into consideration in setting targets, including current conditions of the ecosystem and interactions with other management targets (e.g., fishery harvest in the context of shoreline development and nutrient loading). For instance, managers may be successful in reducing or eliminating hypoxia (i.e. harm) from excess nutrient loading, but if that outcome comes with the functional cost of reducing fishery production (i.e., function) then that should be considered as a part of an ecosystem approach to management. The functional equivalency framework currently used in marine fishery management provides a clear roadmap for setting functional targets, as well as a suite of quantitative tools (Methot, 2009) designed to use the resulting information to make decisions. This framework can also be a viable approach for broader application in other components of ecosystem management, such as water quality management or habitat protection, if the appropriate assessment models, metrics, and thresholds can be identified. Our objective here is to examine how available tools and data can be used in a functional equivalency framework and expanded to other management sectors in estuaries, and to suggest that function provides a better currency for management that either historical or no-harm standards.

1.2. Functional equivalency approach

A functional equivalency (FE) approach to management has clear advantages over other alternatives for target setting because it is based on ecosystem services to people (e.g., harvest, recreational use). Both the 'historic' and 'no harm' management objectives are generally favored because they are based on readily available data about iconic ecosystem structures (e.g., reduce hypoxia, increase SAV coverage). A focus on preservation of an ecosystem function is more complicated, but potentially more versatile, as it sets a functional target that is not dependent on recreating the past (e.g., Maintain target level of fishery harvest not restoring fish population). The FE approach also maximizes the utility of mechanistic models, which can potentially provide estimates of ecosystem function under a suite of conditions and under a wider range of data availability (Fulford et al., 2014; Wu et al., 2015).

Functional equivalency assessments have two important components (Box 1: items i and ii): Assessment metrics that define system state for the target function, and associated thresholds that define when function has dropped sufficiently that managment action should to be taken (i.e., equivalency has been lost). We use the term ‘equivalency’ here to draw a connection to some previous desired state of the system, while keeping assessment focused on the ecosystem function we wish to maintain. The ecosystem is managed to maintain an equivalent functional state, and when equivalency is lost, then action must be taken to recover it. Metrics of a desired ecosystem function then become the measure of interest for assessment. In fisheries management, fishing mortality and spawning stock biomass are metrics used to maintain the function of a stable, fishable population (Southeast Data Assessment and Review; SEDAR, 2016). For assessment in other sectors, such as nutrient management, functional metrics would likewise be chosen based on tradeoffs between the functional costs and benefits of nutrients (e.g., excess nitrogen removal vs. carbon production). This expansion to other multiple sectors will be explored in detail in the two examples.

Decision thresholds that define when equivalency is lost are the second key component for ecosystem assessment as they define actionable change in ecosystem function. For example, a common ecosystem assessment metric is biological condition gradients (BCG, Davies and Jackson, 2006). A BCG is a compilation of data that measure ecosystem state and can be used to track degredation or restoration progress (Gerritsen et al., 2017; Shumchenia et al., 2015). Based on the FE framework, this is only half the needed information for managment. We would also use available data, model simulations, and a policy-driven level of acceptable risk to establish the BCG value at which desired ecosystem functions, such as primary production from seagrass, are considered impaired and use that as a threshold for management action. The BCG then shifts from its typical role of a ‘report card’ for the ecosystem to a formal assessment tool tied to desired functional outcomes. A BCG typically has no absolute meaning, which makes functional thresholds hard to define. We shall endeavor to expand on this approach and look for more suitable metrics in our case study examples. We will use two well-established case studies to explore how the functional equivalency approach taken from a fishery sustainability framework can be used for setting meaningful functional metrics and equivalency thresholds for assessment. This assessment will be based on the target of functional equivalency for multiple management sectors as defined in each case. The message in both case studies is that while current management has been viewed as successful this view is strongly related to either their choice of structural assessment metrics or the deliberate treatment of multiple management issues as independent. The desired outcome is a roadmap for formal assessment with functional metrics, integrated across management sectors, and based on both science and policy.

2. Functional equivalency case studies

2.1. Fishery assessment framework

The concept of functional equivalency is highly developed in the arena of marine fisheries stock assessment (MFSA). In MFSA the target mandated by U.S. law (NOAA, 2007) is a sustainable fishery, meaning the stock can replace biomass lost to fishing through reproduction and growth of individual fish, and this target is reevaluated regularly as a part of the assessment following a structured process (Box 1). There have been extensive efforts to broaden the scope of fishery management from assessments of single stocks to an ecosystem-based fishery management framework (Fulton et al., 2011) including the development of the integrated ecosystem assessment process (IEA; Levin, 2013) and ecosystem level fishery indicators (Link, 2018; Link et al., 2002). These efforts have borne important fruit including a focus on ecosystem function that we advocate here, yet the legal mandate for fishery decision making remains at the scale of single stock and we begin at that scale as most relevant for a practical expansion of the functional approach to other important areas of estuarine assessment. For fishery assessment, sustainable harvest is the target function, a loss of functional equivalency occurs when the harvest becomes unsustainable for any reason. This is a complicated objective as managers typically have control over only one life history process (i.e., harvest mortality), but must account for variation in other processes (recruitment, growth, natural mortality) as a part of setting functional thresholds. For this purpose, quantitative models are highly useful to account for other measurable facets of functional variability (Methot, 2009). However, even model output is seldom useful for making management decisions without the input of policy makers to set decision thresholds for model-derived functional metrics. These equivalency thresholds represent an understanding of fish stock dynamics, collective uncertainty in the available data, and a policy-derived level of acceptable risk regarding the loss of harvest sustainability. All three pieces: models, indices, and decision thresholds must be clearly defined and accepted for management to succeed. Historical targets might be easier to define and measure but may not be achievable under current ecosystem conditions, which is a consideration of assessment under the FE approach.

Marine fishery assessment models, measurable indices of sustainable harvest, and policy thresholds all vary stock to stock, but generally focus on two broad assessment metrics: measures of spawning stock size, and measures of harvest mortality rate (Miller and Swanson, 2001). For example, in the South Atlantic Red Snapper Fishery (South Atlantic Fishery Management Council; SAFMC, 2009), decision thresholds are set by comparing the current and projected spawning stock size to a Minimum Spawning Stock Threshold (MSST) and likewise the harvest mortality rate to a Maximum Fishing Mortality Threshold (MFMT) (Fig. 1). Both values are derived from quantitative analyses of population dynamics adjusted for uncertainty and a level of acceptable risk of overfishing, and are vetted in a public process (SEDAR, 2016). In the case of Red snapper, a clear trend exists from the 1950’s to the 2000’s that crosses over both the threshold for fishing mortality and for spawning stock biomass (Fig. 1) indicating this fishery is currently far from functional equivalency and management decisions are made annually to reverse this situation (Cowan, 2011). The South Atlantic Red Snapper fishery has a long assessment history and is a good example of the challenges faced by single-species management strategies. Yet, the current focus on functional metrics combined with stakeholder engagement (Fig. 1 inset) has borne fruit and shows that assessment can provide a clear decision roadmap to sustainability. Policy implementation, however, is a separate challenge that is beyond the scope of this paper. Marine fishery management is also a good example of how challenges faced in single-sector management are being addressed through efforts to broaden the scope of management to include community-level and biotic-abiotic interactions (Link, 2002). For that reason, it represents a good place to start for an examination of transferable assessment approaches. We will next explore how this functional equivalency assessment approach, well-established in fishery management, can be translated to other management questions, such as the assessment of coastal nutrient loading and conservation/restoration of coastal habitat.

2.2. Chesapeake Bay ecosystem

The first example explores adaptations to the ‘No harm’ standard for decision making (Table 1). Eutrophication is a common problem in estuaries (Howarth, 2008) often caused by excess nutrient loading (Rabalais et al., 2010). Eutrophication leads to higher primary production, which can stimulate production in higher trophic levels, but also yields unstable dynamics yielding episodic hypoxia, and in extreme cases increased mortality of fish and invertebrates (Breitburg et al., 2009). Chesapeake Bay, U.S. (CB; Fig. 2) has experienced long standing issues with eutrophication resulting from increased nutrient loading from it’s watershed (Boynton et al., 2014), as well as documented reductions in grazing by oysters from over-harvesting and disease (Wilberg et al., 2011). In response, federal managers, as well as managers from the five states in the CB watershed, have developed plans to limit nutrient loading into the Bay and restore oyster biomass (EPA, 2000). Human benefit from increases in water quality and oyster biomass include multiple ecosystem services such as habitat improvement, fishery productivity, aesthetics, and the economic value of the Bay, both for residential property values and as a recreation destination (Phillips and McGee, 2014). Targets exist for reducing loading of nitrogen and phosphorus into CB (EPA, 2010b), concentrations for *chl a* (EPA, 2010a), and oyster biomass (EPA, 2000). Extensive work has also been done in defining integrated water quality metrics based on a suite of designated uses and derived from a combination of aquatic life requirements, a “no harm” construct, and historical habitat use (Batiuk et al., 2009). The current designated use approach is focused on reducing harm via hypoxia and phytoplankton blooms and provides a clear benchmark for improvement, but it is not explicitly connected to other management sectors or directly to benefits to people.

In a review of interrelated impacts from management in estuaries, the relationships between reductions in nutrient loading/hypoxia and key services, such as fishery productivity, were sometimes negative suggesting important tradeoffs may exist (Breitburg et al., 2009; Nixon and Buckley, 2002; Riemann et al., 2016). This is a particularly important consideration for the oyster fishery in Chesapeake Bay, which is both highly valuable and directly dependent on pelagic primary productivity (Wilberg et al., 2013). A provision of services approach, such as FE, would focus on these trade-offs to maximize both water quality and secondary production to support high priority services like fisheries, recreational use, and economic value. A comprehensive ecosystem model with functional endpoints can be used as part of an FE framework to assess nutrient loading and oyster biomass based on their impact on ecosystem function. As in the fishery assessment model, we must identify the target functions, functional metrics, and decision thresholds for those metrics useful for decision making. Our target functions will be a sufficient level of both primary and secondary productivity, and our first metric is the rate of primary production (g C . d^{-1}). The managment goal is to maintain sufficient primary productivity to support optimal secondary production, which in turn supports multiple ecosystem services. The threshold of phytoplankton replacement rate supports a target of reducing pelagic primary productivity below potentially harmful levels while maintaining sufficient production (Fulford et al., 2007). In this case, we define functional equivalency as a stable but non-blooming phytoplankton population, which reduces the likelihood of eutrophication while not sacrificing pelagic primary productivity necessary to secondary production. A reasonable first-order decision threshold for phytoplankton primary productivity in this example is to 'remove' sufficient biomass to limit production to 35% of phytoplankton standing stock, which is the annual mean for daily replacement rate (g C . g C . d^{-1}) of phytoplankton in CB (Harding et al., 2002). The removal of phytoplankton biomass necessary to achieve this rate can occur either as reduction in the daily replacement rate via nutrient limitation or through management of secondary foraging (e.g., oysters). This definition is highly complementary to existing standards for *chl* a concentration, but by focusing on a sustainable rate of pelagic primary productivity, this decision threshold considers influence on multiple functional objectives, and nutrient loading can be examined in models alongside phytoplankton grazing and important measures of secondary production.

The second FE metric, based on secondary production impacted by eutrophication, is biomass and distribution of oyster populations in CB (Brumbaugh et al., 2000; Newell et al., 2005). Current oyster biomass is less than 1% of historic levels and oyster restoration is a major fishery management objective in the Bay (Wilberg et al., 2011). Restoration of oysters, in addition to benefiting the oyster fishery, impacts water clarity, stimulates important benthic secondary production, and has potentially significant impacts on pelagic primary production through the direct filtration of phytoplankton from the water column. For this example, we deviate from the year by year approach in the fishery assessment example and adopt a spatial-based assessment for proposed management actions by mechanistically examining different spatial decision options at a fixed timepoint. This assessment makes use of the two functional metrics (phytoplankton daily replacement rate and oyster biomass) and decision thresholds. The target for oyster restoration Bay-wide is a 10-fold increase in biomass but this is not implicitly tied to a functional index. We therefore will use it as a general decision threshold, but we will use the model results to evaluate the oyster biomass needed to achieve our functional equivalency threshold of reducing phytoplankton production at or below annual mean replacement. A full FE evaluation would include a formal equivalency threshold for oyster biomass as well, but this is the subject of future work.

This example assessment was based on a mass balance model for examining the functional equivalency of phytoplankton production in the Bay under different oyster population scenarios currently under consideration. This is an FE threshold-based comparison of different approaches to directed increases in oyster filtration and a comparison between oyster and nutrient management. The assessment employed a model-based food web analysis that included a full suite of phytoplankton grazing effects, as well as physical habitat features such as hypoxia (Fulford et al., 2007, 2010). Models such as these are critical tools for informing managers on the relative impacts of multiple potential decision options, and allow for the inclusion of a wide array of scientific data into the decision process beyond direct measures of nutrient concentration, phytoplankton biomass, or concentrations of chlorophyll *a*. The analytical outcome was that eutrophication can be contained (i.e., stable pelagic primary production) with either a 50% reduction in nutrient load or a 25-fold increase in oyster biomass. Yet, the reduction in pelagic primary productivity at least partially through grazing is more likely to support functional endpoints for secondary productivity (Fig. 3).

The current target for oyster restoration is a 10-fold increase but this increase will not impact all CB sub-estuaries the same way. Nutrient load reduction is more effective at achieving the FE target Bay wide, but also results in large predicted reductions in secondary production that may negatively impact secondary production important to achieving functional targets (Fig. 3; Fulford et al., 2010). Evaluation of oyster management options shows restoration at the 10-fold biomass target can achieve our functional threshold for phytoplankton removal overall under limited harvest conditions and locally in only three sub-estuaries of CB. A 25-fold increase is needed in most sub-estuaries of CB to achieve the threshold (Fig. 4), however in several sub-estuaries 25X increases are too high in terms of oyster filtration and far exceed the functional target for limiting primary production suggesting a reduction in filtration benefit and possibly negative production effects similar to nutrient load reductions. The results of oyster restoration are spatially specific in that comparable restoration in certain sub-estuaries of Chesapeake Bay will result in different optimal functional responses based on current state of the local oyster population and water exchange dynamics (Fulford et al., 2007). Sub-estuaries predicted to achieve functional equivalency can be a focal point for functional restoration, and productivity dynamics in these sub-estuaries can be studied to explore how to achieve similar results in other parts of CB.

Examination of management actions within an FE-based assessment allows managers to compare Bay-wide nutrient reduction targets to restoration targets for oysters on a local scale and evaluate where and how management decisions are most likely to contribute to functional equivalency for primary and secondary production. This is an improvement over existing approaches as oyster harvest and nutrient load reduction are assessed as interrelated issues and the dependencies can be integrated into decisional guidance. Using our assessment approach, specific sites would be recommended for oyster restoration investment based on functional targets and an integrated model-based evaluation of effects on both primary and secondary production. Looking Bay wide, nutrient load reduction combined with successful oyster restoration with specific spatial targets is more likely to support achievement of functional thresholds. Further, limited oyster harvest scenarios do not improve functional outcomes, suggesting reef location is more important than reef quality to production of ecosystem services. Overall, an FE assessment framework for CB would be most useful if it included both nutrient loading and oyster biomass functional thresholds and evaluated the collective effect of these two actions on the ecosystem function of stable primary and secondary productivity.

2.3. Tampa Bay ecosystem

The second example explores adaptations to the ‘historic target’ approach to decision making. Tampa Bay is a shallow estuary located on the central Gulf Coast of Florida and the Tampa area is one of the fastest growing communities in the U.S. (Fig. 2). The watershed for Tampa Bay is comparatively small (5617 square km, 5 times the Bay surface area; Greening and Janicki, 2006) and as a result freshwater delivery into the Bay is highly variable and strongly correlated with recent rainfall (Yates et al., 2011). Since the late 1970’s, Tampa Bay has been in recovery from years of high nutrient loading from both point and non-point sources resulting in reduced water clarity, numerous harmful algal blooms, and large-scale loss of submerged vegetation. High priority ecosystem services in Tampa Bay include water quality and clarity, recreational fishing opportunities, and economic impacts of tourism and coastal property values. From this point of view, ecosystem function can be best defined based on a balance between Bay productivity and water quality. Full recovery of the lost seagrass areal coverage was declared in 2014 (Greening et al., 2014). This recovery involved reductions in nutrient load into the Bay and was accomplished through extensive efforts to engage the public in reducing nutrient load sources and monitoring the recovery of the Bay. Monitoring centered on several structural metrics for a healthy estuary developed by experts, namely mean *chl* a level in the Bay and total light predicted to reach the bottom. These are indices of recent health rather than measures of function so represent a ‘report card’ approach for gauging near-term health based on an historical target. Decision thresholds associated with these indices exist but are not used directly for management. Rather, evaluation takes the form of reporting progress through time, which represents assumptions about how these indices are related to functional recovery. This is not to say the report card approach cannot have an effect. In fact, structural recovery targets set based on historic reference points for the Bay, including restoration of submerged vegetation, have largely been met in recent years (Tampa Bay Estuary Program; TBEP, 2017). Nonetheless, a focus on structural indices rather than function has two important limitations. First, with no formal connection between indices and human benefit, the value of achieving these targets can be hard to relate to benefits for stakeholders. Second, and more importantly, the optimal value of these indices is hard to communicate once restoration targets have been met making them less informative for gauging system resilience and sustainability over the long term. In particular, the known delay in seagrass response to changes in eutrophication means that functional targets are needed to anticipate impacts before they are observable (Cardoso et al., 2010; Petraitis and Hoffman, 2010). Functional metrics are an improvement over structural or empirical metrics because they can be directly linked to manageable services provided by the ecosystem and allow for identification of potential trade-offs between management goals such as the impact of reduced pelagic productivity on fishery production (Hondorp et al., 2010; Riemann et al., 2016). In the case of Tampa Bay, we maintain the core goal of improving both water quality and aquatic vegetation as a key driver of overall productivity, but focus on a key function provided by submerged vegetation: removal of anthropogenic nitrogen via denitrification (Seitzinger et al., 2006). Submerged vegetation has many other values, such as fostering secondary fishery production and stabilization of bed sediment, however the functional equivalency approach does not require all services to be included to effectively inform management. As with the fishery example, we are interested in measuring function in multiple ways that are linked directly to manageable elements of the system and have clear function-relevant thresholds to inform decision making.

In the case of Tampa Bay, the function targeted for assessment is processing excess nitrogen delivered into the Bay, and the proposed functional metrics are annual mean anthropogenic nitrogen load and an assessment of the amount of this annual N load removed naturally by sea grass present in the Bay (Fig. 5). We could approach the choice of metrics in TB in the same way we did in CB, but here SAV restoration goals have been achieved and the future management objective is to maintain function rather than how to restore it, and this is best achieved with a slightly different set of metrics. Seagrass denitrification rates were estimated based on a nitrogen budget model for the Bay with published rates of nitrogen exchange assembled by benthic habitat type (Table 2). The model output is not a restatement of SAV coverage in different units, but accounts for spatial distribution, as well as abiotic factors such a salinity and temperature. Nonetheless, while it is sufficient for this example, the estimates of denitrification are a first cut estimate that can and should be improved in practice through application of available models (Fulford et al., 2016; Rogers et al., 2017; Tomasko et al., 2001). As in the functional assessment framework, the first metric measures the direct outcome of a management action (i.e., controlling nitrogen load), and the second measures its longer-term outcome in terms of ecosystem function (i.e., seagrass functional benefits). A decision threshold for these two functional measures is the target proportion of annual nitrogen load removed by seagrass (Fig. 5). An examination of a functional equivalency plot over the period from 1976 to 2006, shows that the primary change over that period has been in annual nitrogen load with a distinct reduction from 1976 to 2006. However, if, we examine the estimated proportion of the load removed by seagrass we see that the estimated value was lowest in 1976 at less than 10% and stabilized out during the recovery period (2001–2006) between 20 and 30%.

This is still well below an estimated historical reference point of 46% for seagrass coverage and nutrient loading in the 1950’s (Russell and Greening, 2015; Tomasko et al., 2005) compared to mean nitrogen loading rates 1996–2006, but the Bay is currently considered to be functionally equivalent to historical conditions, so achieving historic maximums may be an unnecessarily conservative restoration target similar to achieving an unfished biomass level in an exploited fish population. Neither is necessary to achieve management goals for the system. In this example, we propose an annual functional target based on estimates of nitrogen load and sea grass nitrogen removal using a lower decisional threshold of 20% removal. This can be achieved through a reduction in nitrogen loading as is indicated by the data in Fig. 5, but this is a low productivity solution as indicated by the location of the data relative to the overall functional equivalency threshold. Potential exists to structure recovery of seagrass so to achieve functional equivalency at a higher level of nitrogen loading (Fig. 5), which would allow for higher pelagic secondary productivity supported both by higher nutrient and habitat availability. Secondary productivity in Tampa Bay is historically low in comparison to other similar estuaries (Breitburg et al., 2009) and this suggests that reductions in nutrient delivery into the Bay may have an important impact on pelagic production and recreational yield, particularly for pelagic species (Hondorp et al., 2010). In this case, the goals shift away from a water clarity metric towards a broader focus on achieving and sustaining desired levels of ecosystem services from seagrass restoration based on a functional metric. This would be a clear improvement to current assessments, as it better informs long-term sustainability of function. A shift to the ‘right’ inside the functional equivalency zone (Fig. 5) might be achieved by shifting the target from total seagrass coverage to site specific restoration in places where nitrogen removal is predicted to be maximized. This shift would be a viable trade-off to overall productivity in Tampa Bay and spatially explicit models are a valuable tool for this sort of multi-sector restoration planning.

Nitrogen removal by sea grass is a value that is best estimated with a model (e.g., Giordani et al., 2008) and such tools exist, as do the data to inform their use. Application of functional thresholds require model-based synthesis, and this is a more effective long-term choice than structural indices that lack clear decisional guidance for longer term sustainability. The chosen threshold is an example of unifying scientific information with policy. Assessment models allow for the comparison of possible threshold values in the context of desired outcomes and the resources necessary to achieve them. A full FE assessment in this case would include model-based evaluation of the sea grass-nutrient load balance based on alternative management actions. This better informs the goal of balancing nutrient loading with Bay-wide productivity to support other services like recreational fishing and abundance of iconic species.

3. Discussion

In these two examples, we demonstrate an approach for operationalizing integrated ecosystem assessment based on functional endpoints. A functional equivalency approach can be contrasted with historic and no-harm standards and opens the assessment process to better consider trade-offs among management outcomes by keeping management objectives focused on the desired benefits. The differences between the FE approach and more traditional methods for defining management thresholds can be subtle and are not represented in a simple change in metrics. The differences are reflected in the emphasis on a policy-driven functional outcome for decision rules, increased use of model-based prediction, and the examination of broader suite of data as a part of restoring or maintaining function. This is a shift towards the concept of ecosystem services (Engle, 2011; Heck et al., 2008) as a management endpoint and is easily integrated into ecosystem assessment frameworks like the NOAA Integrated Ecosystem Assessment (Harvey et al., 2017), as well as more general decisional frameworks like Structured Decision Making (Gregory et al., 2012). In their description of marine ecosystem assessment in a fisheries management context, Link et al. (2002) developed clear examples of ecosystem metrics for assessment, but also highlighted the need for “less arbitrary” ecosystem reference points so that clear actions can be associated with these reference points. The functional approach we advocate here as described in Box 1 is intended to do exactly that.

The fishery assessment framework provides a transparent approach, but more importantly includes identification of meaningful thresholds for decision making that are amenable to a model-based assessment. The limitations of marine fishery management based on the current single species approach is apparent in Fig. 1 and efforts to improve the process are a large part of the model for assessment we showcase in this study. Link et al. (2002) in their discussion of operationalizing ecosystem-based fisheries management highlight the need for reference points clearly tied to policy, the value of mechanistic modeling, and the need to take a cross-disciplinary approach. Reliance on complex model-based assessment has been criticized based on the perceived increase in uncertainty resulting from model output and this issue has been well-studied in the application of models in fishery stock assessment (Carmichael, 2010). The consensus view is that such uncertainty exists as risk in policy setting, whether it is quantified or not, and model-based tools are an important aspect of increasing transparency of this issue and focusing effort on reducing its impact on management decision making (Lehrter and Cebrian, 2010; Martin and Johnson, 2019). The use of FE as described in Box 1 in marine fishery assessment has demonstrated that the acceptance of uncertainty associated with model-based assessment by policy makers and stakeholders is strongly tied to transparency and early stakeholder engagement. These are two steps we strongly advocate in application of a functional equivalency approach to multi-sector assessment.

These case studies are meant to be informative examples, but the application of the functional framework to multi-sector assessment is highly flexible to other potential metrics. It will be most informative if these metrics are quantifiable either empirically or through model-based assessments, sensitive to management actions under consideration, and measure change in functional endpoints important to people (e.g., secondary productivity can be linked to improvements in iconic species biomass). Such metrics are best established through a collaboration of science and policy considerations, such as practiced in marine fishery assessment (e.g., SEDAR, http://sedarweb.org/cooperator/gmfmc). The data needs are high, and this approach will not apply equally in all systems and for all issues, yet the use of model-based assessments aids in the process where direct empirical data are limited. In both cases described here, management decisions had to be made based on the best information available, and a formal assessment based on the functional equivalency paradigm that makes full use of a model output allows for management continuity across management sectors, while also getting the most value from all the available data.

More specifically, the functional metrics chosen in both cases include a dynamic metric sensitive to inter-annual variability and a more stable metric responsive to longer term, more permanent change in the ecosystem. This approach is consistent with the base approach in that fishing mortality measures annual behavior towards sustainability and spawning stock biomass reflects the functional change over multiple years. Both metrics are individually informative about the state of the managed stock, but combined they provide an integrated picture of how management affects stock sustainability. The two ecosystem examples both include management of nutrient loads into the respective estuaries as a metric with annual variability, but also consider a suite of long-term functional metrics (e.g., seagrass/oyster services) that reflect how nutrient load and these other managed ecosystem elements combine to provide functional value to human stakeholders including delays and feedback issues such as hysteresis. These more stable metrics consider the functional value of a reduced nutrient load and in some cases, may suggest a loading rate higher than expected is needed to preserve biological or ecological function of the system.

Multiple frameworks have been developed for coastal ecosystem management specifically and complex decision making more generally (Gregory and Keeney, 2002; Harvey et al., 2017). These frameworks have been referred to as 'organized common sense' as they greatly aid the process of organizing disparate scientific information and linking it to important management questions. Yet, it has been noted that operationalizing these frameworks for decision making requires a clear translation of science into the language of policy (Harvey et al., 2017). This is an area where the FE approach can greatly contribute to the utility of these frameworks in real-world applications by defining metrics and decision thresholds strongly supported by both scientific data and public policy objectives. Such an integration has already occurred in Marine Fishery Management through the SEDAR framework (SAFMC, 2009) and we have demonstrated here that it can be expanded to consider other multiple management issues.

4. Conclusions and way forward

A functional equivalency approach to estuarine ecosystem assessment has the potential to advance the sophistication of decision making beyond single-sector assessment and allow for the maximum use of available scientific information in the decision process. This is an improvement over more traditional assessments, which employ approachable indices (e.g., spatial coverage) but fail to assess improvement in sustainable ecosystem services (e.g., nitrogen sequestration and productivity) or important trade-offs between management sectors (e.g., nutrients and fisheries). The data requirements of the FE approach are high, yet the use of assessment modeling greatly aids in estimation. Model-based assessments are commonly used and accepted for decision making in marine fishery management, and while the same is not generally true for nutrient management, the modeling tools are available (Hagy and Murrell, 2007; Stow et al., 2003). The dependence of an FE approach on stakeholder input for determining thresholds is also high, but this provides an opportunity for improving both management outcomes and stakeholder acceptance. The functional equivalency approach includes five important steps that we see as critical to a successful transfer to ecosystem management (Box 1). Future work will involve an application of the approach and a full vetting of both FE metrics and the associated functional thresholds with stakeholders. The application of the functional equivalency approach to sustaining the ecosystem services of estuaries can help to operationalize existing ecosystem management frameworks for decision making.

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Box 1

Steps of a functional equivalency approach to ecosystem management:

i. Choose assessment metrics that reflect chosen functional objectives (e.g., resource sustainability).

ii. Choose decision thresholds for these metrics that are sensitive to loss of function, reflect uncertainty, and reflect policy-based levels of acceptable risk for loss of function.

iii. Choose assessment models that are sensitive to management-based alterations of the system, estimate chosen metrics, and directly inform on decision thresholds.

iv. Consider how to use the framework at varying levels of data availability and quality [decisions always must be made].

v. Engage stakeholders in both the development of functional metrics and associated thresholds, as well it's application for management.

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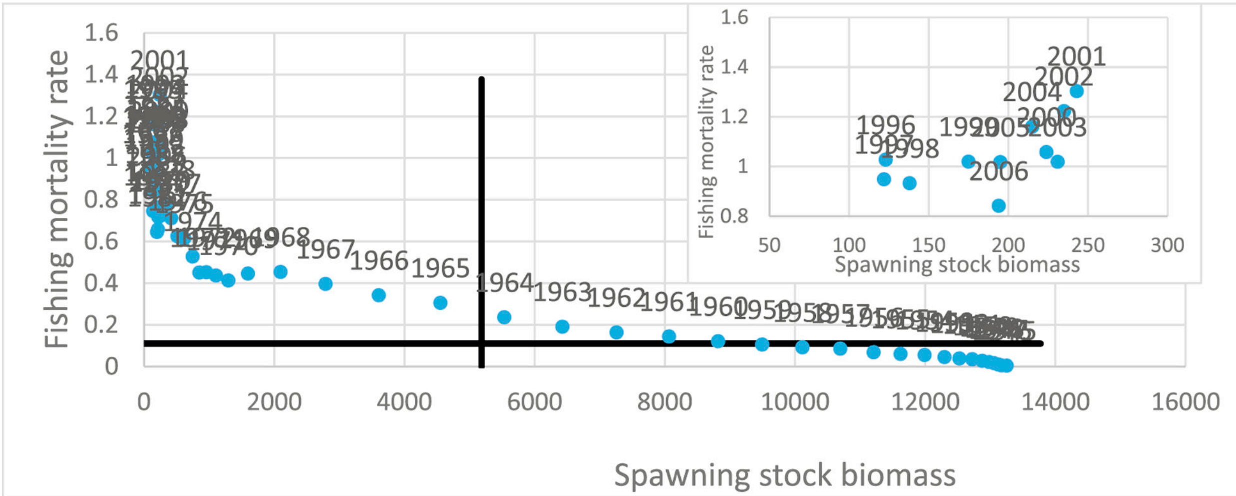

Fig. 1.

Time series figure for Red snapper (*Lutjanus campechanus*) assessment showing condition of the stock by year (1945–2006) based on two functional metrics. Inset graph shows most recent ten years in this timeseries (1996–2006), which is the period of functional assessment as described in this paper. The assessment model indices are spawning stock biomass and fishing mortality rate and the functional thresholds are Minimum Stock Size Threshold (MSST) and Maximum Fishery Mortality Threshold (MFMT). Thresholds are based on a model estimate of maximum sustainable fishery yield (SEDAR, 2016).

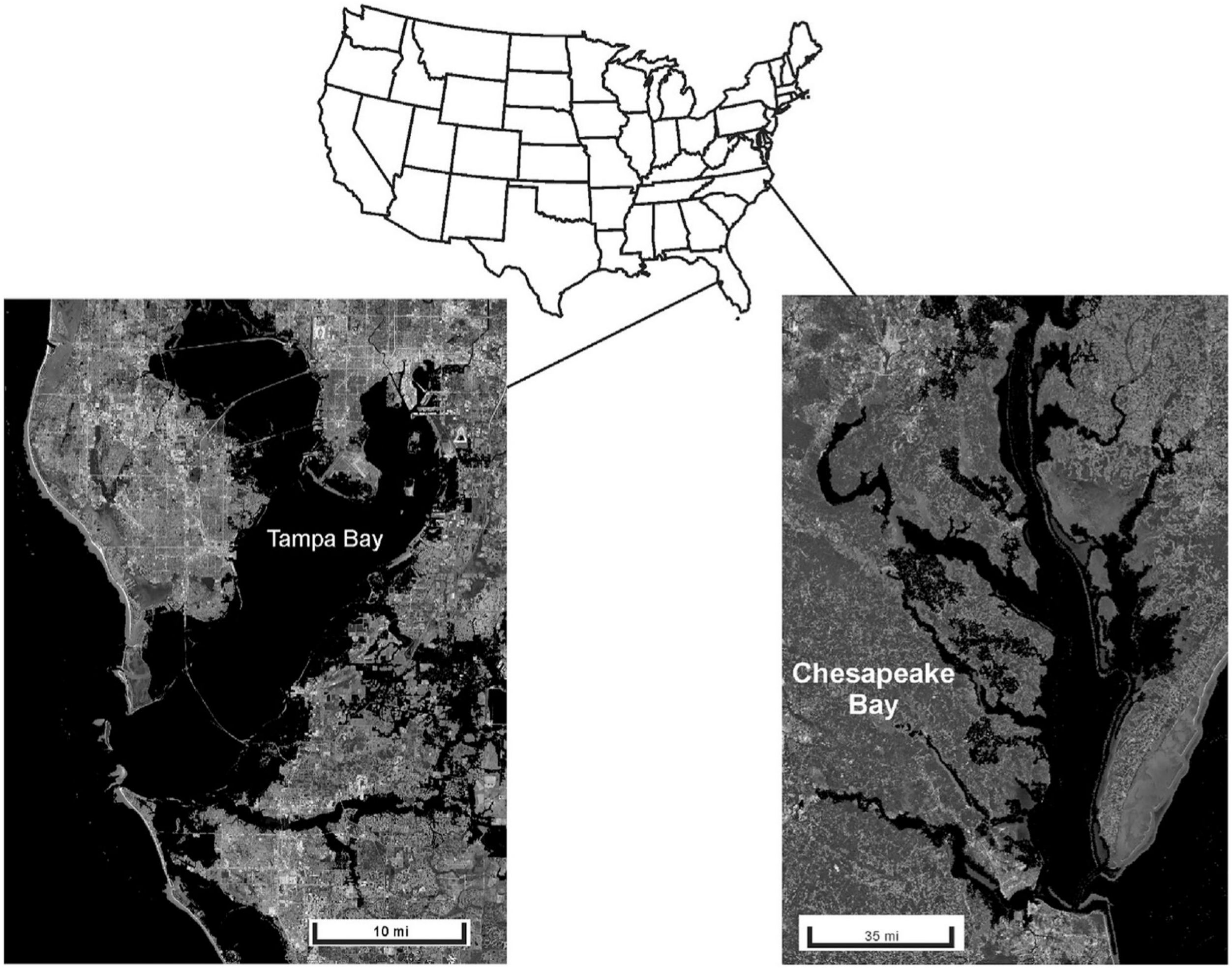

Fig. 2.
Location of two example estuarine ecosystems: Chesapeake Bay and Tampa Bay, FL.

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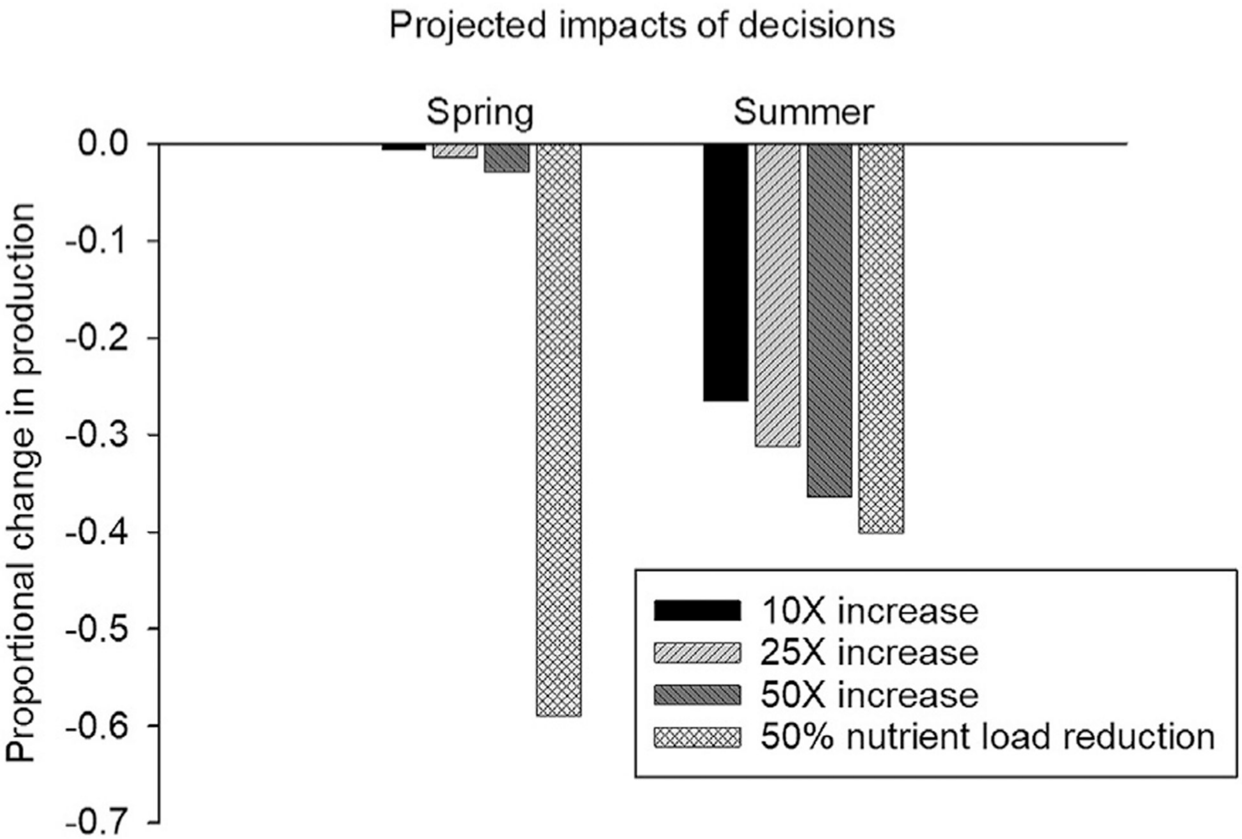

Fig. 3.
Comparison of proportional change in pelagic primary production from three levels of Chesapeake Bay wide oyster restoration and a 50% reduction in Bay wide nutrient loading rate. Oyster biomass is increased from a baseline value specific to individual sub-estuaries and monitoring segments of the main Bay. Results are summarized from Fulford et al., (2010).

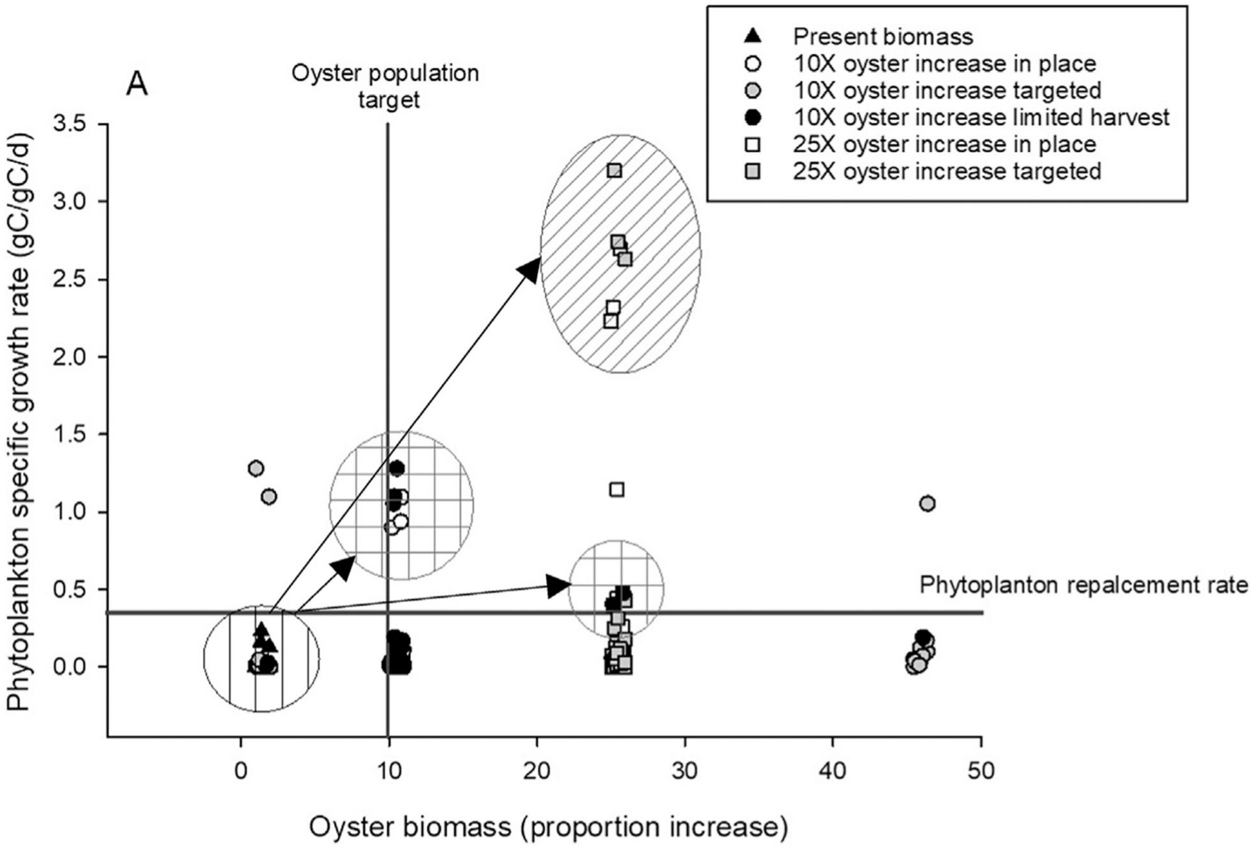

Fig. 4.

Functional equivalency (FE) plot for nutrient-oyster management question in Chesapeake Bay (CB). The chosen FE indices (Mean annual phytoplankton specific growth rate rate/ oyster biomass increase) are plotted and the target thresholds are indicated by horizontal (phytoplankton replacement rate) and vertical (oyster population restoration target) solid lines. Points are for projected impacts in specific individual sub-estuaries of CB. Symbol fill indicates targeted Bay wide oyster restoration and shape indicates one of three restoration scenarios examined (1) Restoration in place, 2) Restoration only in targeted estuaries, 3) Restoration with harvest restrictions to maximize oyster mean size on reefs). This plot shows example localities with predicted state under current conditions (large circle, vertical hatching) and transitions that meet functional targets (large circles, box hatching) or far exceed targets (large circle, slanted hatching) to show how projections can be used to identify functionally equivalent scenarios.

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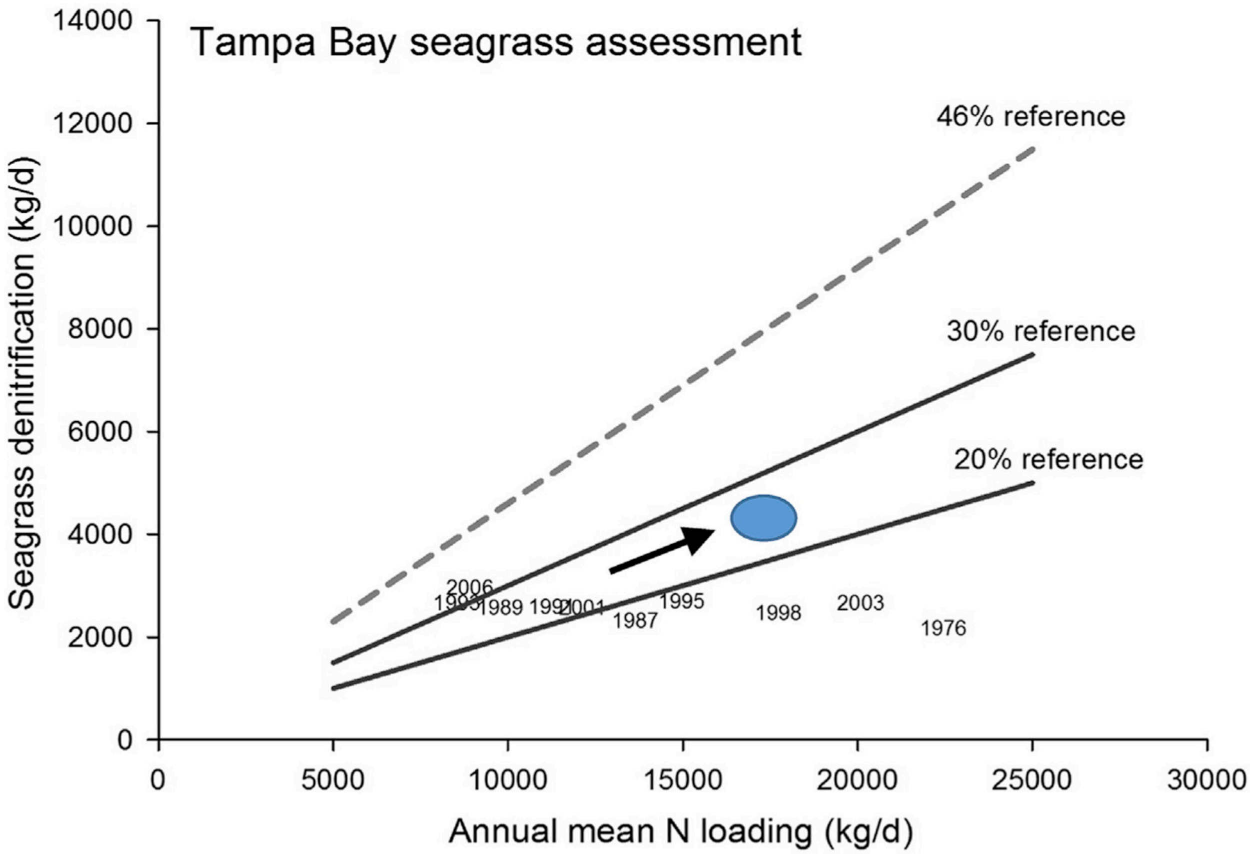

Fig. 5.

Functional equivalency (FE) plot for nutrient-seagrass management question in Tampa Bay, FL. The chosen FE indices (nutrient loading rate and proportion of annual N load removed by seagrass) are plotted and the thresholds are indicated by linear relationships (solid lines) of constant proportion. Estimated historical patterns are shown (hatched line) for reference. A ‘right’ shift from the current management strategy emphasizing reductions in bay nutrient load to one that balances seagrass coverage with overall system productivity is indicated by the arrow leading to an example of a functionally equivalent target (solid circle) farther along the nutrient axis.

Table 1

Example contrast of fishery management and nutrient management under the three described methods for setting management criteria. The term TMDL refers to Total Maximum Daily Load.

Management criteria strategy	Fishery management example	Generic nutrient loading example
Historic target — primary goal is to restore system to known historic state ofa reference ecosystem component.	Fishing mortality set to be no higher than estimates from Year X when the population was viewed as healthy.	TMDL set to past, known loading rate when estuary was viewed as healthy.
No harm - primary goal is to reduce observed harm to one or more ecosystem components.	Fishing mortality minimized or eliminated to remove negative impacts on an iconic species (e.g., sturgeon)	TMDL targets reduction in loading rate to level associated with a target reduced harm such as minimizing or eradicating hypoxia.
Functional equivalency — primary goal is to restore and maintain specific ecosystem function(s) to a policy-based level set based on acceptable risk.	Fishing mortality set at maximal rate that allows for pre-determined level of population stability through time (e.g., five-year review cycle)	TMDL set as a part of an estuary nutrient budget to a target associated with maintaining secondary production established as sufficient to sustain both desired fish community biomass and water quality.

Table 2

Nitrogen removal as denitrification by estuarine habitat type used in nitrogen budget model for Tampa Bay, FL. Denitrification is converted to function by restating as percentage of total nitrogen removal and percentage of the known nitrogen loading rate from anthropogenic sources in a reference year (2006).

Habitat type	Annual mean denitrification rate (kg N/ha/d)	Areal coverage (ha)	% removal	% loading (2006)
SAV	0.246	11998	0.17	0.39
Wetlands	0.046	7906	0.02	0.05
Sediment	0.05	68296	0.2	0.46
*Burial	0.002	*88200	0.01	0.023

*Burial is included as a component of all habitat types to distinguish this from denitrification. Data were compiled specifically for Tampa Bay (Ranade et al., 2015).