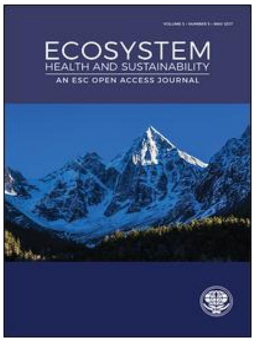

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Abstract. Economic valuation of ecological restoration most often encompasses only the most tangible ecosystem service benefits, thereby omitting many difficult-to-measure benefits, including those derived from enhanced reliability of ecosystem services. Because climate change is likely to impose novel ecosystem stressors, a typical approach to valuing benefits may fail to capture the contribution of ecosystem resilience to sustaining long-term benefits. Unfortunately, we generally lack predictive probabilistic models that would enable measurement and valuation of resilience. Therefore, alternative measures are needed to complement monetary values and broaden understanding of restoration benefits. We use a case study of Chesapeake Bay restoration (total maximum daily load) to show that ecosystem service benefits that are typically monetized leave critical information gaps. To address these gaps, we review evidence for ecosystem services that can be quantified or described, including changes in harmful algal bloom risks. We further propose two integrative indicators of estuarine resilience—the extent of submerged aquatic vegetation and spatial distribution of fish. Submerged aquatic vegetation extent is indicative of qualities of ecosystems that promote positive feedbacks to water quality. Broadly distributed fish populations reduce risk by promoting diverse responses to spatially heterogeneous stresses. Our synthesis and new analyses for the Chesapeake Bay suggest that resilience metrics improve understanding of restoration benefits by demonstrating how nutrient and sediment load reductions will alleviate multiple sources of stress, thereby enhancing the system's capacity to absorb or adapt to extreme events or novel stresses.

Key words: *benefit-relevant indicators; climate change; economic valuation; ecosystem services; non-monetary benefit indicators; resilience; total maximum daily load; water quality.*

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Introduction

Ecosystem service valuation has been promoted as a means to broaden the set of natural resource tradeoffs beyond what has traditionally been included in cost-benefit analysis or other types of performance assessment (Holl and Howarth 2000, Plummer 2009). Many federal agencies with missions to manage and protect natural resources have been expanding their use of ecosystem service analysis to communicate and track program benefits. Additional impetus for such efforts came from a multi-agency memo that directed federal agencies to consider ecosystem services in their planning and decision-making (OMB et al. 2015).

Despite the strengths of valuation to communicate benefits and demonstrate the economic efficiency of a proposed action (Hahn and Sunstein 2002, Sunstein 2014), the values that can be monetized will not capture the full suite of social concerns, due to multiple methodological issues including limits of data and understanding (Ackerman and Heinzerling 2002, Slovic 2000) and cultural norms that make monetization difficult or inappropriate (Winthrop 2014). Further, when considering how to mitigate risks from climate change or many other broad social concerns, information gaps can render monetized benefits ineffective for representing some primary goals of restoration. For example, it is rare for ecosystem service monetization to represent the contributions of system resilience to the reliability or longevity of benefits.

Another common limitation of valuation for representing restoration goals occurs when the enabling legislation or motivating policy for natural resource agencies is to protect *non-use* or *passive use values*, which are among the most difficult to measure. Non-use and passive use values represent the satisfaction that people derive from being good

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stewards of the environment and making it available to other users or future generations, distinct from any expected use of the resource (Krutilla 1967, Cross 1989). Use values, on the other hand, represent the direct or indirect tangible goods or services from ecosystems, such as outdoor recreation (e.g., recreational fishing, wildlife watching), resource extraction (e.g., timber harvest, commercial fishing), and hazard mitigation (e.g., property protection from flooding). Use values are most easily monetized and tend to be correlated with the presence of people. Therefore, measuring use values alone will create a systematic bias in favor of actions in densely populated areas, at the expense of actions in remote or lightly populated areas.

Here, we explore what valuation captures and omits for a restoration case study, the nutrient and sediment caps being implemented as part of the Chesapeake Bay total maximum daily load (TMDL; US EPA 2010). We examine what is represented well in valuation and the information gaps often associated with such analyses, particularly regarding long-term socio-ecological system resilience. To address the need for appropriate measures of ecosystem service value, given the uncertainties of climate change and other ongoing system changes, we explore how changes in system resilience might be quantified.

Resilience has been defined in many ways (Holling 1973, Carpenter et al. 2001). Perhaps an apt concept of resilience under climate change is for a system to "… continually change and adapt yet remain within critical thresholds" (Folke et al. 2010). This concept acknowledges that the system will change but that people value maintaining the system within certain bounds that provide desirable and consistent ecosystem goods and services.

Resilience is typically broken down into separate qualities of resistance, recovery, and adaptive capacity, although terms vary (Carpenter et al. 2001, Walker et al. 2004). Here, we use the term "resilience" as an umbrella term to represent multiple concepts. The most measurable or observable aspects of resilience may be resistance and recovery (Hodgson et al. 2015), where resistance is the ability to avoid change and recovery is the rapidity with which a system returns to its prior state following disturbance. The proximity to tipping points, which has been called *precariousness* (Walker et al. 2004), is what we desire to measure, as a function of water quality and other conditions. However, the science has not yet evolved to allow us to characterize this state with much accuracy, nor does it provide strong correlates of measurable system characteristics (e.g., biodiversity) with specific resilience outcomes (Hooper et al. 2005, Loreau and de Mazancourt 2013).

Resilience to perturbation is likely to increase in importance as the bay ecosystem resists, recovers, and adapts to climate change. Some threats are already evident; for example, some species of submerged aquatic vegetation (SAV) and fish, such as striped bass, have limited tolerance to high summer water temperatures and SAV growth can be limited by changing salinity. Other potential threats include increased susceptibility to disease (Roessig et al. 2004), invasive species (Sorte et al. 2013), and effects of eutrophication and hypoxia (Rabalais et al. 2009, Miller et al. 2016). Accordingly, increased capacity for resistance and recovery could shorten time in a degraded state (Dai et al. 2012) following the extremes of temperature, precipitation intensity, and salinity variability projected for the Chesapeake Bay under a changing climate (Najjar et al. 2010).

Valuing the Chesapeake Bay TMDL

For our case study, we examine benefits produced by plans to meet annual caps on nitrogen, phosphorus, and sediment that are currently being implemented in the Chesapeake Bay watershed TMDL (Fig. 1). The TMDL is a regulatory tool of the Clean Water Act (33 U.S.C. §§ 1251–1387, 2006) that directs states to establish pollution caps to achieve water quality goals. For the Chesapeake Bay, the TMDL was established to restore the aquatic habitat, which, in turn, will support and maintain human uses such as safe recreation and food harvesting. To achieve the TMDL, states develop *watershed implementation plans* that

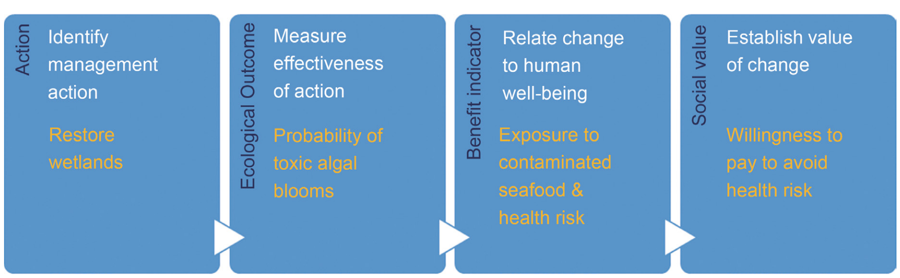


Fig. 1. Conceptual value diagram connecting actions to social benefits. Many cause-and-effect relationships are needed to measure social values of an ecological change and limits of current data and understanding can constrain which benefits can be valued. To demonstrate the challenges, the diagram includes a hypothetical example in which wetland restoration is first linked to a socially meaningful ecological change of the probability of toxic algal blooms. This link could be modeled by assessing the degree to which wetlands reduce nutrient loads to receiving water bodies and then assessing how those changes in loads reduce risk of blooms. Next, the risk of toxic algal blooms could be made relevant to people by estimating a change in the probability of people getting sick from eating fish. Finally, this change would be valued by summing up what all affected fish consumers might be willing to pay to avoid this risk. Each of these links presents analytic challenges that must be overcome if valuation is to succeed.

specify actions needed to meet the nutrient caps. These actions include non-point source controls placed on agricultural land, some urban stormwater practices, and upgrades to wastewater treatment plants.

We consider three approaches for valuing changes induced by the TMDL: (1) individual ecosystem service benefits that have been monetized and quantified, (2) enhanced reliability of ecosystem services, and (3) benefits representing existence, bequest, and altruistic values that arise from improvements in ecological condition and resilience (Madariaga and McConnell 1987). We briefly describe approaches to value ecosystem services using monetary units or benefit indicators to show why alternative analytic approaches may be needed to represent benefits.

Measurement of Ecosystem Service Benefits

Regardless of whether benefits are measured in monetary or non-monetary units, demonstrating ecosystem service benefits typically involves three main steps in a cause-and-effect sequence (Fig. 1). Three primary connections are needed to value an action: (1) trace a proposed management action to a change in a biophysical condition of the ecosystem, (2) quantify the relevance of that change to human well-being, and (3) value that ecological change by determining what people would be willing to trade off (i.e., pay) to achieve that change (Wainger and Mazzotta 2011). Often, many interacting biophysical connections must be quantified to generate a biophysical change that can be valued.

Consider a hypothetical example (Fig. 1), in which an analyst aims to connect wetland restoration to the benefit of lower health risks from toxic algal blooms. Conducting such an analysis first requires understanding how wetlands change nutrient loads and in-water concentrations, using measures that are relevant to harmful algal bloom (HAB) presence and toxicity. Similarly, social behavior reflecting how people fish, consume fish, or are otherwise exposed to fish with toxins would be assessed to calculate changes in HAB exposure risk. Finally, willingness to pay for the reduction in exposure risk would need to be estimated for the affected population.

When any of the steps to connect an action to a benefit cannot be quantified, monetary values of the final use (e.g., health risk) cannot usually be estimated. An alternative approach to measure potential benefits is to quantify, but not monetize, metrics that occur earlier in the cause-and-effect chain. The more that ecological outcome metrics can be tailored to beneficiary concerns, the more effective they become as *benefit indicators* or *benefit-relevant indicators*, as they are also known.

For our example, estimating HAB risk is extremely difficult, so decision makers are more likely to rely on simple metrics, such as changes in nutrient concentrations. However, an ecological change metric will only be effective in communicating and evaluating benefits if it resonates with stakeholders or decision makers. Probability of toxic algae blooms is more meaningful than nitrogen concentrations for a manager charged with allocating scarce resources because people are better able to judge their willingness to invest to avoid a specific health risk (toxin exposure) compared to an indicator of unspecified risks (nitrogen concentration). However, many resonant benefit metrics can be difficult to measure, requiring proxy measures to suggest benefits that cannot be directly measured. In some cases, it may be possible to value these proxy metrics; however, monetary values can be unreliable when benefits or risks are not specified.

Benefit indicators for non-use, or intangible, values are not simple to develop. What does it mean to tailor a metric to a beneficiary concern when value derives from some aspect of the system that people are not using but wish to maintain into the future? Some commonly expressed non-use values are *bequest values*, which is the satisfaction people derive from knowing that ecosystem goods and services will be available to future generations (Lazo et al. 1997). For example, we have heard people say they would like their grandchildren or great grandchildren to be able to fish for striped bass that are safe to eat. If that is the goal, a benefit metric should measure the probability of achieving it.

Maintaining bequest values for an estuary such as the Chesapeake Bay requires that we consider climate change as well as the full suite of changes affecting aquatic ecosystems (Rabalais et al. 2009, Breitburg et al. 2015*b*). Yet, the way in which multiple stressors interact to influence biota (in desirable or undesirable ways) is difficult to project with certainty (e.g., Najjar et al. 2010), suggesting that we may not be able to monetize some concerns that motivate ecological restoration.

The remainder of this paper is divided into three sections that cover the three types of valuation proposed for the Chesapeake Bay: (1) values of ecosystem service changes, (2) values of enhanced reliability of ecosystem services, and (3) value of ecological resilience.

Monetized, Quantified, and Described Values of Ecosystem Services

To understand information gaps in current practices of ecosystem service valuation, it is useful to examine the benefits that have been measured for the Chesapeake Bay restoration. The major ecosystem service benefits that have been valued (in monetary units) for the TMDL implementation include property value enhancements, recreation, seafood provision, aesthetics, and non-use values (Table 1). Although broad in scope, the ongoing evaluation represents only a subset of potentially monetizable benefits (Cropper and Isaac 2011).

Obvious omissions from Table 1 are recreational fishing and boating benefits, which prior research has suggested are likely to be generated by water quality

Table 1. Values measured for the Chesapeake Bay TMDL.

Ecosystem service	Spatial extent of beneficiaries	Monetary value	Authors
Property value enhancements due to enhanced water clarity	Waterfront and near-waterfront homes	$400–700 million (present value in perpetuity)	Klemick et al. (2016)
Property value enhancements due to expanded SAV extent	Waterfront and near-waterfront homes	$300–400 million	Guignet et al. (2016)
Multiple use and non-use benefits to watershed residents [from increased abundance of striped bass, crab, and oysters; improved water clarity in the bay (not related to home value); and reduced algae in lakes]	About 80% of the total benefits accrued to watershed residents who did not use the bay's tidal waters	$1.20–6.49 billion/yr	Moore et al. (2015)
Climate change damage costs avoided due to carbon sequestration†	Watershed	$12.6 million/yr	US EPA (2011: appendix, scenario 8a)
Health and aesthetic benefits due to air quality improvements†	Watershed	$2.5 million/yr	US EPA (2011: appendix, scenario 8a)
Hunting†	Watershed	$0.2 million/yr	US EPA (2011: appendix, scenario 8a)

Note: SAV, submerged aquatic vegetation; TMDL, total maximum daily load.
† Estimates based on optimized scenarios of TMDL implementation and not the implementation plans that were used in other values shown in table.

improvements (Lipton and Hicks 1999, Lipton 2004). Yet, modeled changes in fish abundance and diversity (referenced in Moore et al. 2015) projected either fisheries increases or decreases with the TMDL, depending on assumptions and model specifications. Although many assume that the TMDL will result in overall greater fish abundance (perhaps based on stories of the legendary abundance of sea life in Chesapeake Bay (Smith and Barbour 1986) under lower nutrient loads), empirical models developed for numerous fisheries, using assessment data and commercial landings, generally show a positive relationship between fish abundance (measured as mass) and nutrients (Breitburg et al. 2009*a*, *b*).

This example of the uncertain responses of fish populations to water quality is a telling example of why benefits of an ecosystem service analyzed with readily available data, and in isolation from other ecosystem changes, might fail to recognize resilience effects. Models built from historic data may not effectively represent how counterbalances in the socio-ecological system can mask signs of stress. For example, in the short term, negative effects of hypoxia on some species can be offset by increased productivity of fish that increases feeding rates when their forage is concentrated at the boundary of hypoxic zones (Pruell et al. 2003, Craig 2012). Further, fisheries catch data can reflect changes in human behavior that may mask declines in fish abundance (Pauly et al. 1998, Caddy 2000, Kemp et al. 2005, Essington et al. 2006). Fish abundance likely depends on a balance of these competing factors and feedbacks that can become unexpectedly unbalanced (Kemp et al. 2005, Breitburg et al. 2009*b*, Rose et al. 2009).

Quantified and described values for ecosystem services

Some ecosystem service co-benefits that have been identified but not monetized for the watershed implementation plans include (1) health protection of humans and domestic animals from pathogen reductions, (2) human and ecosystem health protection from reductions in HABs, (3) human health protection from West Nile Virus (WNV), and (4) regional economic benefits as a consequence of preventing stigmatizing water bodies.

Additional services, not detailed here, have been quantified for alternative implementation scenarios, including brook trout habitat quality (*Salvelinus fontinalis*) and flood risk reduction from additional wetlands (US EPA 2011). We did not conduct an exhaustive search for ecosystem service values for the Chesapeake Bay TMDL and have knowingly omitted studies that calculate economic impacts from spending, since these cannot be considered benefits in federal CBA. We also omitted valuation studies that used methods that are not derived from economic welfare theory, such as those that multiply a value per acre by all acres of an ecosystem (for further explanation, see Toman 1998).

We quantify and describe existing and novel analyses that support the idea that these ecosystem services are likely to change in response to the TMDL. Unless otherwise noted, all results in this section represent analyses comparing the implementation of all management practices specified by the watershed implementation plans for the TMDL relative to the 2009 baseline developed by US EPA to represent on-the-ground conditions in 2009, minus annual conservation practices (US EPA 2010).

Pathogens

Richkus et al. (2016) evaluated potential health protection benefits for humans and domestic animals and estimated a 27% reduction in fecal indicator bacterial loads to tidal waters of the bay from septic and agricultural management actions (sewage treatment improvements are not included). Current pathogen levels result in shellfish bed closures, beach closures, and cause a variety of human and domestic animal illnesses. Reductions in such outcomes would, therefore, be expected to create substantial benefits.

Harmful algal blooms

Reduced nutrient loads from the TMDL are projected by the Chesapeake Bay Program (CBP) model suite to reduce mean algal biomass by 40% (Table 2), and this finding is supported by field studies that show a correlation between nutrient loads and total algal biomass (Malone et al. 1996, Boynton et al. 2014). Reduction in total algal biomass has the potential to translate to a reduction in HAB occurrences (Heisler et al. 2008). Yet, effects are likely to vary by species because HAB species differ in their preferences for salinity, temperature, and nutrients, and therefore, a change in any one of these variables may reduce likelihood of occurrence or extent of one species, but increase likelihood or extent of another.

The Chesapeake Bay has many HAB species (Marshall et al. 2005) that can cause harm. Among the more common species are *Karlodinium veneficum*, which produces a toxin that can kill fish; it and *Prorocentrum minimum* can cause oyster recruitment failure (Wikfors 2005, Deeds et al. 2006, Glibert et al. 2007, Stoecker et al. 2008, Tango and Butler 2008). *Microcystis* sp., produces toxins that have the potential for substantial ecological and human health impacts (e.g., Codd et al. 1999, Brand et al. 2010). Because of the potential for harm, reduced HAB abundance would be expected to increase safety of activities involving water contact (boating, swimming, fishing), seafood handling, and consumption, and enhance security of businesses relying on water activities and seafood harvesting (Hoagland et al. 2002, Ramsdell et al. 2005).

Valuing a change in HABs is hindered by the inability of models to make specific predictions of how HAB species ranges will shift in response to nutrient changes (e.g., Kim et al. 2014) and whether those shifts will affect people. Yet, since risk of HAB occurrence can be correlated with water quality, water quality variables can suggest a directional change in risk. We developed an indicator of *HAB habitat suitability* for the three common species noted above, using their documented preferences for salinity and temperature and evidence that their occurrences or harmful properties are favored when ratios of dissolved inorganic nitrogen to dissolved inorganic phosphorus (DIN : DIP) exceed Redfield proportions (>16:1) (Redfield 1934, Glibert and Burkholder 2011, Glibert et al. 2014, Li et al. 2015). Applying these factors provides reasonable approximations of historical occurrences (Fig. 2, panels B vs. A, E vs. D, and H vs. G). Our index is a simple representation of percent of time when all assumed growth conditions were met, but is not intended to represent total biomass or level of toxins.

We evaluated potential changes in this index with the TMDL using the CBP estuarine model. Results suggest that, although overall magnitude of blooms is likely to be lower (by reduction in all algae), the TMDL would only modestly reduce the extent of habitat suitability for *Prorocentrum* and would likely have negligible effects on *Microcystis* and *Karlodinium* (Table 3). Small changes in areas of occurrence were also projected (Fig. 2, panels C vs. B, F vs. D, and I vs. H). These results are only intended to indicate general habitat suitability for these specific HABs.

Table 2. Projected water quality conditions with and without the TMDL.

Water quality variable	Without TMDL	With TMDL	Difference	Percentage of change
Average DIN : DIP (by weight)	10.8	6.5	−4.3	−40
Avg chl-*a* (spring; μg/L)	11.8	6.8	−5.0	−42
Avg chl-*a* (summer; μg/L)	11.5	6.8	−4.7	−41

Notes: TMDL, total maximum daily load. A negative change represents a projected decline in nutrients or chl-*a* with the TMDL. Model output data were provided by P. Wang, Chesapeake Bay Program Office. 10 December 2016.

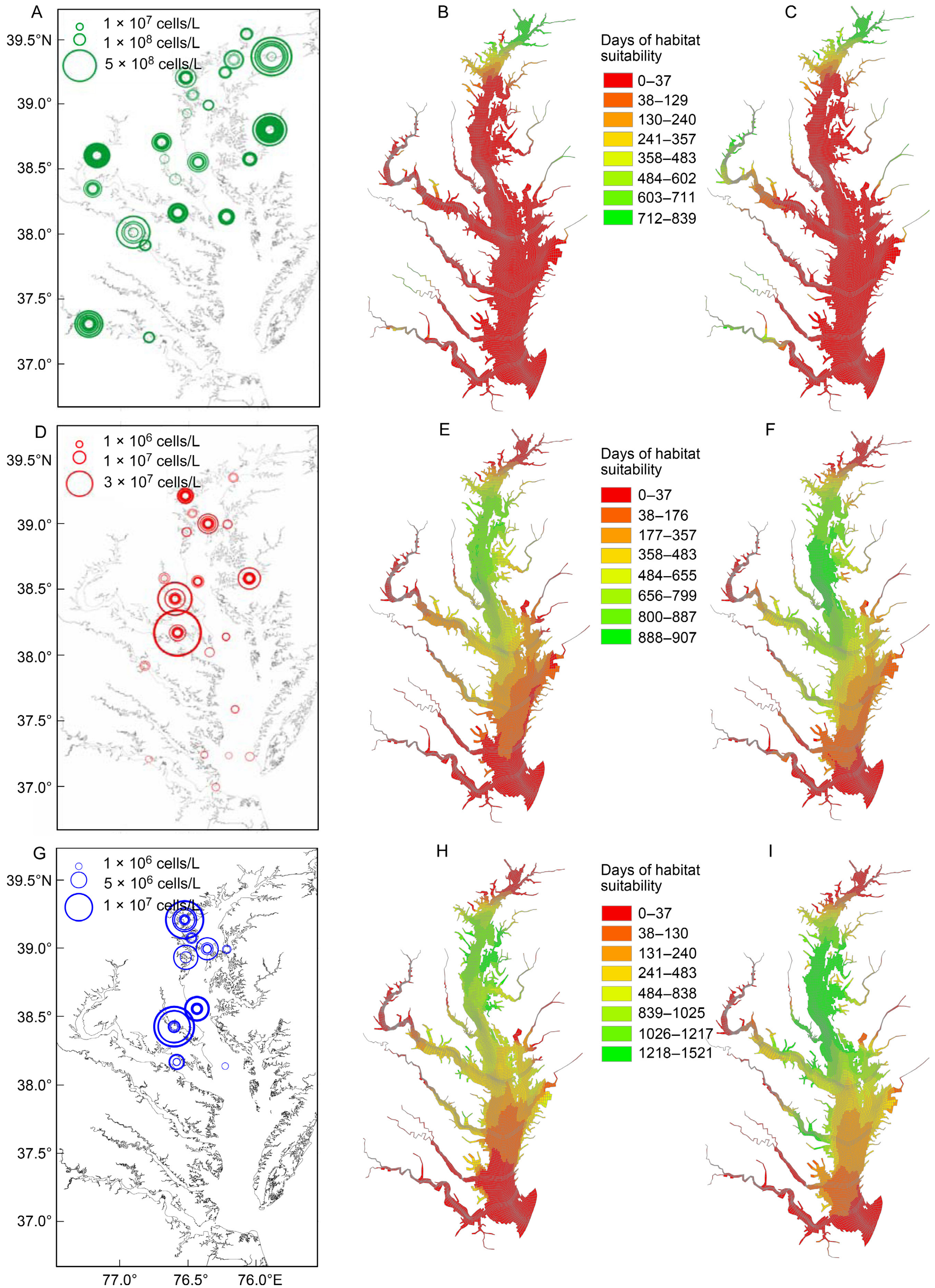
West Nile Virus

In 2015, 70 cases of WNV and six WNV-related deaths were reported in Virginia, Maryland, and D.C. (CDC 2016). Although counterintuitive, increases in wetland area and other vegetated areas suitable for bird habitat are negatively correlated with WNV transmission to humans (Ezenwa et al. 2007, Allan et al. 2008, Swaddle and Calos 2008). The TMDL implementation plans propose 600,000 acres of riparian buffers and wetlands that could reduce human health risks from WNV transmission due to biodilution. Bird diversity is positively correlated with area of forested land and wetlands. When bird diversity is high, WNV reservoirs are diluted (Melles et al. 2003, Ezenwa et al. 2007) and potential incidence of WNV in humans is reduced.

Regional economic support from preventing stigma of bay waters

Total maximum daily load implementation potentially will create an economic benefit more valuable than specific risks to users, by reducing perceived risks from using the bay. While the public safely uses the majority of the bay, fish kills, closed beaches, and rare occurrences of severe illnesses and deaths resulting from *Vibrio* and other bacteria in the bay heighten awareness of potential risks (Hogan 2014, Haendiges et al. 2016). Perceptions of the bay as a risky place to recreate can have broad economic implications because such stigma can generate behavioral shifts disproportionate to the risks (Kasperson et al. 1988, Slovic 2000), which can, in turn, generate disproportionate economic harms.

The process whereby small risks generate large changes in behavior has been called the *social amplification of risk* (Kasperson et al. 1988). It occurred in the Chesapeake Bay region in 1997 when people responded to reports of a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358–483
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656–799
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Fig. 2. Historical distributions of several common harmful algal blooms in Chesapeake Bay and projections of their habitat suitability based on their reported temperature, salinity, and N:P preferences. Panels A–C represent *Microcystis aeruginosa*, D–F represent *Prorocentrum minimum*, and G–I represent *Karlodinium veneficum*. Panels A, D, and G are the historical spatial distribution and abundance based on monitoring data (reproduced from Li et al. 2015 with permission of the publisher). Panels B, E, and H are modeled scenarios of days of suitable habitat (see text for definition), using weather from the period of 1991–2000, assuming baseline nutrients. Panels C, F, and I represent days of suitable habitat with nutrient reductions by the total maximum daily load.

HAB in one isolated area by avoiding seafood and canceling fishing trips throughout the bay. This event is referred to as the *Pfiesteria* outbreak and causes of reported lesions in fish and health impacts on watermen have since been debated (Blazer et al. 2000, Burkholder and Glasgow 2001, Kiryu et al. 2005). Even though effects were confined to a few areas or seafood types (Magnien 2001), the public response had a substantial, though temporary, effect on restaurants, charter boat businesses, seafood wholesalers, and others (Meyer 1997). In a survey, Whitehead et al. (2003) confirmed the high-risk aversion among the public to such threats and estimated that lost benefits to seafood consumers in the Mid-Atlantic would be $37–72 million in the month following a fish kill caused by a HABs.

The TMDL can lower the threat of minor risks generating major economic ripples by reducing the number of adverse incidents (fish kills, HAB events). A reduction in the number of adverse events would be expected to attenuate economic risk by changing peoples' perceptions about the seriousness of minor events. When people are confident that a place is safe and have few personal experiences with harms, they are less likely to amplify risk (Kasperson et al. 2003).

Value of Ecosystem Service Reliability

Measuring changes in selected ecosystem services that are enhanced by the TMDL provides important information about benefits, but does not capture resilience to climate change or other stressors. One way that resilience could enter into valuation would be to calculate the enhanced value that accrues from making a good or service more reliable. In many markets, people are willing to pay a premium for goods and services that are more reliable or resistant to risk. For example, homes in flood zones that are threatened by sea level rise had 21% higher sales prices (all else equal), if they had structural elements to allow them to withstand flooding (Walsh et al. 2015).

Table 3. Illustration of HAB suitability with and without the TMDL for three common HAB species.

HAB species	Without TMDL (ha-d)	With TMDL (ha-d)	Difference (ha-d)	Percentage of change
Microcystis	1.19×10^{12}	1.19×10^{12}	3.37×10^{8}	0.03
Karlodinium	8.24×10^{12}	8.58×10^{12}	3.42×10^{11}	4.14
Prorocentrum	5.94×10^{12}	5.18×10^{12}	-7.62×10^{11}	−12.83

Notes: HAB, harmful algal bloom; TMDL, total maximum daily load. Habitat suitability is quantified as hectare-days which are calculated as (sum of days during which all habitat conditions are suitable) × (hectares/cell) × (cells) for a 10-yr period. A negative change represents a projected decrease in the HAB extent and/or duration with the TMDL, but it does not reflect change in cell concentration. Model output data were provided by P. Wang, Chesapeake Bay Program Office. 10 December 2016.

Considering the idea that reliability adds value to ecosystem services, we expect that people would receive greater benefits when ecosystem services were temporally consistent in the places where they use or enjoy the service. However, reliability values could be difficult to disentangle from other components of value. Nonetheless, further development of data and models could present opportunities to measure contributions of reliability to value by quantifying price premiums paid for reliable goods and by developing ecological models to project the degree to which restoration actions reduce variability of desirable outcomes.

Value of Ecological Resilience

If we were successful at valuing reliability, we still would not be measuring resilience in the sense of preventing a system from reaching a tipping point that degrades or eliminates multiple ecosystem services. Case study examples of estuaries or semi-enclosed seas support the idea that eutrophication (excess nutrients) sets the stage for shifts to undesirable states. Evidence from lakes and semi-enclosed seas (Great Lakes, Baltic Sea, Black Sea, Sea of Azov, and Mediterranean Sea) suggests that, in concert with fishing pressure and other stressors, eutrophication-induced hypoxia has played a role in reducing the abundance of some economically important fish and invertebrates and in enhancing the dominance of undesirable invasive species, including jellyfish (Caddy 1993, 2000, Diaz 2001, Breitburg 2002, Daskalov 2002, 2003, Oguz 2005). The documented deterioration of fisheries in the Black Sea (Daskalov et al. 2007, Oguz and Gilbert 2007) indicates that detrimental effects of eutrophication may not fully manifest until stressors combine. For the Black Sea the combination of nutrient loading, excessive fishing, unusual climate regimes, and introductions of alien species overwhelmed the ecosystem's resilience.

Because of the uncertain ways in which stresses can combine, particularly under climate change, incorporating resilience in values of the changes induced by the TMDL requires viewing the aquatic ecosystem as an orchestrated system, rather than a bundle of separate ecosystem services. Although many ecologists often study just one aspect of ecosystems (e.g., fish, algae, microbes), they recognize that complex biophysical interrelationships maintain characteristic system functioning (Gunderson 2000, Borja 2014, Sheaves et al. 2015) and create resistance to multiple stressors (Russell and Connell 2012, Breitburg et al. 2015*b*). Thus, a piecemeal approach to valuing ecosystem services, while practical and persuasive, is incomplete if we are to comprehend the value of restoration actions in preventing tipping points.

We have evidence that some ecosystems have responded to stress by shifting, often abruptly, from one stable state to another (May 1977, Gunderson 2000), but the threshold level of stressors required to drive the shift is virtually impossible to quantitatively predict (Rietkerk et al. 2004). This gap in understanding limits our ability to project probability of a shift—and through a risk assessment—attach a monetary value to avoid such a shift. If we could generate risk probabilities and consequences, we would have a means to estimate benefits in terms of a reduction in the probability of negative outcomes (Keeney and Raiffa 1993). But traditional risk assessment is likely to fail to adequately manage risk of low probability but high consequence events (Camerer and Kunreuther 1989) and people may not be willing to pay to "insure" against low probability events (Slovic et al. 1977).

The analytic limits to risk assessment suggest that traditional economic valuation that relies on individuals to judge what they are willing to spend to avoid risk may not be able to represent the values for protecting long-term public welfare that emerge through institutional deliberations (Vatn 2009). Although people value the existence of species and ecosystems (e.g., Richardson and Loomis 2009, Rudd et al. 2016), ecologists may not be able to quantify how specific actions reduce risk to species, which is needed to enable options to be valued. Therefore, non-monetary metrics that measure the degree to which actions promote resilience and lessen risk of system collapse may be the only quantitative approach to representing collective desires that cannot easily be represented through monetary valuation.

To generate non-monetary benefit indicators of resilience value, we synthesized available ecological evidence that indicated water quality improvements would lead to socially desirable outcomes. The ecological mechanisms by which TMDL implementation may enhance the Chesapeake Bay ecosystem's ability to resist and adapt to disturbance fall under a common theme; that is, reducing nutrients and sediments alleviates multiple sources of stress and enhances the system's capacity to respond to extreme events or novel stresses (Boesch 2000, Carpenter et al. 2012).

Specifically, research suggests that estuaries are more resilient when two conditions are met, as detailed below: (1) SAV is extensive and (2) fish, particularly juveniles, are broadly distributed across suitable habitat. Because SAV and fish habitat extent depend on diverse biological, chemical, and physical processes, they are indicative of how the component ecosystem parts are functioning and interacting. Further, these conditions directly support ecological functions that may help a system maintain desirable structures and functions and promote rapid recovery from disturbance. The following sections summarize evidence supporting these metrics and how they may change with the TMDL in the Chesapeake Bay.

Why SAV extent is a measure of resilience

Increasing SAV extent and abundance is a frequent goal of estuarine restoration because both are indicators of overall system condition. Submerged aquatic vegetation extent reflects water quality because SAV only grows when water clarity, and associated light availability, is sufficient for photosynthesis (Dennison et al. 1993). The beds of SAV that can carpet the shallows of estuarine systems provide ecological structure that supports diverse aquatic life (Heck et al. 2003). Moreover, SAV promotes ecosystem resilience through its ability to engineer the ecosystem by stabilizing sediments and creating positive feedbacks that improve water quality (De Boer 2007).

Submerged aquatic vegetation is thought to support aquatic life (fish and mobile invertebrates) because it harbors abundant food resources relative to unvegetated areas (Heck et al. 1989, Thorp et al. 1997) and provides conditions that promote survival of economically important species (e.g., striped bass or *Morone saxatilis*, spotted sea trout or *Cynoscion nebulosus,* and blue crabs or *Callinectes sapidus*) and other juvenile fish (Orth et al. 1984, Perkins-Visser et al. 1996, Heck et al. 2008). Some fish congregate in greater densities within SAV beds relative to unvegetated areas (Lubbers et al. 1990, Duffy and Baltz 1998, Heck et al. 2003), and loss of such habitats is thought to increase stresses on those fish populations (Beck et al. 2001). Research in Chesapeake Bay found that juvenile crab density increased as a function of SAV density (Ralph et al. 2013). However, strong evidence that SAV is required to support fish populations remains elusive (Blackmon et al. 2006).

Submerged aquatic vegetation also serves as gathering grounds and sources of food for diverse migrating or overwintering waterfowl (see Kemp et al. 1984). Declines in waterfowl populations are correlated with SAV declines (which does not demonstrate causality). But, at a minimum, reduced SAV extent and diversity have reduced diversity of waterfowl diets, which is potentially a risk to some populations since diet diversity promotes stability (Perry and Deller 1996).

Once established, SAV beds can improve their own growing conditions through positive feedback processes that enhance water clarity and increase light availability to plants. The physical structure of a plant bed slows moving water (Koch 2001, De Boer 2007), which allows suspended particles to sink to the bottom and reduces the likelihood of particle resuspension (Ward 1985, Gacia and Duarte 2001, De Boer 2007). Submerged aquatic vegetation also takes up nutrients and, in some cases, enhances denitrification (McGlathery et al. 2007), thus removing nitrogen from the aquatic system. When nutrient concentrations decrease, phytoplankton and epiphytic algae also decrease, which increases light availability to SAV. These positive feedback loops are self-stabilizing, enabling SAV beds to withstand fluctuating environmental conditions, and generally increase in strength as a function of bed density and size (Luhar et al. 2008, Gruber et al. 2011). Larger, denser beds, therefore, have a greater buffering capacity and are considered to be more resilient (Gruber et al. 2011, Suykerbuyk et al. 2015, van Katwijk et al. 2016).

An expansive SAV population, consisting of dense continuous beds, may translate to greater resilience for the broader ecosystem, as the same feedback processes that directly benefit SAV growth can also affect regional ecological processes. For example, nutrients bound in plant tissue or attached to trapped particles are sequestered during the algal growing season (Havens et al. 2001, McGlathery et al. 2007). Wave attenuation and sediment trapping may also reduce storm surges, shoreline erosion, and property damage (Koch et al. 2009, Feagin et al. 2010, Barbier et al. 2011, Barbier 2013).

Kemp et al. (2005) estimated that SAV restoration to historic coverage in the main-stem upper Chesapeake Bay alone would remove, on average, 45% of nitrogen inputs from the Susquehanna River. Because nutrients fuel algal production which, in turn, is linked to the volume of hypoxic water and related trophic imbalances in Chesapeake Bay, SAV-mediated nutrient removal could help buffer systemic responses to disturbances, such as nutrient and sediment pulses associated with floods.

Why broadly distributed fish populations are a measure of resilience

Climate change will influence the temporal and spatial incidence of stressors on fish populations and communities, particularly the frequency of hypoxia (low dissolved oxygen levels created by excess nutrients in combination with physical conditions), which directly and indirectly impairs fish and fisheries production. Hypoxia creates well-documented stresses on fish by degrading and reducing the spatial extent of foraging and nursery habitats. Reduced habitat availability can limit reproduction, curtail migration, and concentrate fish into smaller volumes, increasing predation rates, fishing mortality, disease incidence, and growth (Coutant and Benson 1990, Brandt et al. 2009, Rose et al. 2009, Craig 2012, Campbell and Rice 2014, Kraus et al. 2015). On the other hand, hypoxia can induce high concentrations of prey, resulting in efficient commercial fishing (Craig 2012) and efficient feeding opportunities for piscivores (fish that eat other fish), which include many sportfish (Pihl et al. 1992, Kraus et al. 2015).

Yet, we cannot rely solely on current measures of fish abundance to indicate resilience because abundance of some indicator species is relatively insensitive to systemic stress. Rather, what is needed is a metric that captures the capacity for behavioral and physiological adaptations to climatic and other stresses (Rose et al. 2009).

We propose that indices related to the spatial distribution of fish populations and communities can serve as metrics of resilience, just as SAV extent serves that purpose. The logic in this proposal is that evenly spread and redundant distributions of fishes among bay regions promote *response diversity*. Response diversity (Elmqvist et al. 2003) represents the capacity of fish populations and communities to generate uncorrelated responses to stresses, much as financial portfolios are invested in bonds and stocks to diversify responses to economic forces (Secor et al. 2009, Schindler et al. 2010). Spatial dispersal is part of a bet-hedging strategy to promote long-term persistence and range extensions in response to altered habitats (McPeek and Holt 1992).

Two types of response diversity are particularly important to create resilient populations: (1) ability to mitigate effects of spatially heterogeneous stresses and (2) capacity to colonize peripheral habitats under changing conditions. The occurrence of redundant fish populations in multiple locations is valuable because acute stressors may be tributary-specific (Paerl et al. 1998, Bailey and Secor 2016) and, therefore, catastrophic changes in a subset of habitats (from hypoxia, storm events, toxic spills, and other stressors) would be less likely to impact most individuals (Schaaf et al. 1993, McGilliard et al. 2011). The unaffected, but populated areas provide refuge for a portion of the fish, allowing those individuals to rebuild the population. Further, maintaining populations in habitat that is considered peripheral today, may allow climate change adaptation through future range expansion (Nye et al. 2009, Petitgas et al. 2013).

Evidence from many taxa supports the idea that response diversity, represented by broad spatial distribution, enhances a community's ability to resist stress. The conservation of population segments of American eel and white perch that reside in different regions of Chesapeake Bay has succeeded in maintaining stable populations in the face of stresses that differentially affect those regions (Secor 2015). Similarly, stable abundance of sockeye salmon on the west coast of North America that are subjected to high and variable levels of fishing mortality was attributed to distributing that stress across geographically dispersed populations (Schindler et al. 2010). More generally, the role of spatial buffering, for example, occupancy of both central and peripheral habitats in a species range, in reducing fishing and other anthropogenic stresses has been recognized across diverse settings and species (Duplisea and Blanchard 2005, Secor et al. 2009, Kerr et al. 2010, Schindler et al. 2010, Yates et al. 2012, Secor 2015).

Evidence that peripheral habitats can serve as refugia in times of stress and contribute to resilience comes from research external to the Chesapeake Bay. For example, white sturgeon in the lower Columbia River estuary retreated to coastal regions in response to ash and sediment loads from the Mount. St. Helens volcanic eruption (DeVore et al. 1999). Similarly, Bailey and Secor (2016) observed striped bass evacuating the Hudson River in the aftermath of tropical storms. In both case studies, fish were forced to move into less ideal, peripheral habitat that sustained the populations in times of stress.

Conservation of multiple habitat types or areas countervails a common management tactic in spatial planning, which is to conserve only the most productive fish habitats. Applying this concept, only those nursery habitats producing the majority of adults receive high priority for protection (Beck et al. 2001). This tactic risks loss of persistence resilience in stressed and changing environments by ignoring habitats that may respond differently to stress (Kraus et al. 2015) or by concentrating individuals in protected habitats, thus increasing their risk to catastrophic events (McGilliard et al. 2011).

Developing a fish distribution metric is more difficult than simply observing spatial extent. Fish distributions are influenced by species- and population-specific reproduction, feeding, and migration, and habitat is often patchily distributed. As a result, resilience metrics derived from fish distributions must be sensitive to these factors. Survey sampling can support metrics and define trends in incidence of occurrence and diversity of habitat use by fishes in nursery habitats and indicate how fishes respond to habitat-limiting conditions such as hypoxia (Campbell and Rice 2014, Kraus et al. 2015).

We used response diversity rather than *species diversity*, which has also been suggested as an indicator of resilience (Yachi and Loreau 1999, Loreau et al. 2001, Hooper et al. 2005, Hector and Bagchi 2007). However, evidence is mixed as to whether species diversity is truly representative of resilience (May 1977, McCann 2000). More refined biodiversity metrics that are closely linked to functional differences between species may eventually lead to metrics that are more tightly correlated with resilience (Petchey and Gaston 2006, Ives and Carpenter 2007).

How is the TMDL Likely to Change these Measures of Resilience?

The SAV resilience story

Although positive feedbacks enable SAV beds to withstand disturbance, high nutrient loads can push beds over a stress threshold, beyond which they collapse. Recent observations of seagrass beds suggest that the TMDL might prevent systems from reaching such tipping points. High nutrient loads limit the light available for photosynthesis, which decreases available energy stores and makes plants more susceptible to dying during light-limiting disturbances such as turbidity pulses during storm events (Moore et al. 1997, Longstaff and Dennison 1999, Yaakub et al. 2014). Under these conditions, SAV cover becomes increasingly fragmented, reducing the capacity for feedbacks that enhance habitat conditions (Montefalcone et al. 2010, Santos et al. 2016). Therefore, nutrient and sediment loading rates, to a great extent, determine bed resilience to multiple stressors.

Evidence of threshold responses in SAV (primarily *Vallisneria americana, Heteranthera dubia,* and *Hydrilla verticillata*) beds comes from recent events. The Susquehanna Flats SAV bed was a vibrant lush ecosystem that served as valuable habitat for fish and waterfowl and offered popular sites for anglers and hunters until SAV abundance began to decline in the 1960s (Bayley et al. 1978). Through the 1960s and early 1970s, SAV cover gradually declined to about 30% of historic peak levels, coinciding with deteriorating water quality. The final straw was Tropical Storm Agnes in 1972, which delivered a massive influx of water, nutrients, and sediments that tipped the system into a new regime of low SAV cover (Kemp et al. 2005). Although the storm itself triggered rapid plant loss, increased nutrient loading rates appear to have decreased the system's resilience, preventing SAV regrowth.

Remarkably, the Susquehanna Flats SAV recently underwent a rapid (2005–2010) and complete recovery and was able to withstand two subsequent extreme weather events (Hurricane Irene and flooding associated with the remnants of Tropical Storm Lee in 2011). Research by Gurbisz and Kemp (2014) suggested that recovery was initiated by gradual long-term reductions in nutrient loading coupled with calm conditions and exceptional water clarity during an extended dry period. Although the decreasing nutrient inputs led to gradual improvement in water quality (Orth et al. 2010), the drought provided the extra push necessary to tip the system into a new normal of abundant SAV cover. Once a critical mass of grasses was established, feedbacks that reduced wave energy and promoted propagule establishment likely accelerated bed expansion and enhanced its buffering capacity. For example, when the 2011 storm events battered the region with high waves and massive sediment loads, the large, dense SAV bed attenuated water velocity and decreased suspended particle concentrations, facilitating persistence despite deleterious conditions (Gurbisz et al. 2016). Two years after the storm, clear water from within the SAV bed regularly "spilled over" into adjacent regions during ebb tides. This facilitated recovery of damaged portions of the bed by increasing light availability for new plant growth. The system, therefore, demonstrated a high degree of resilience in its recovered state.

Does this story apply to other SAV beds and species?

As the Susquehanna Flats SAV recovery story suggests, changes in a single source of chronic stress (nitrogen and phosphorous) can alter SAV resilience to other pressures (storm events), which can lead to bed loss or degradation. Similarly, the TMDL may help multiple SAV species resist anticipated stresses from additional climate change impacts. For example, the combined stress of high summertime temperatures expected with climate change (Preston 2004, Najjar et al. 2010) and low light as a result of eutrophic water can lead to severe eelgrass bed loss. High temperatures create stress on eelgrass (*Zostera marina*) by increasing its respiration rate, thereby requiring more light to maintain plant conditions (Wetzel and Penhale 1983, Evans et al. 1986, Moore et al. 1997). Beds lose the ability to recover from stress when combined stresses occur in consecutive years because plants can die before producing seed, thereby depleting the seed bank (Moore and Jarvis 2008, Jarvis and Moore 2010, Moore et al. 2012). Similarly, the higher variability of salinity in the bay that is expected with climate change (Neff et al. 2000, Najjar et al. 2010) is likely to stress a second SAV species, wild celery *V. americana*. French and Moore (2003) found that light requirements may be 50% higher when this plant is growing in higher salinity, suggesting that periods of both high salinity and low water clarity will be difficult for the species to tolerate.

Will the TMDL improve SAV extent and resilience?

In 2015, bay grasses were just under half the 2020 areal extent goal set by the CBP. The roughly 75,000 ha goal incorporates all bottom area historically known to have contained SAV based on aerial photographs from the 1930s to recent decades (Batiuk et al. 2000). The TMDL caps were set, in part, based on the water clarity needed to enable SAV to again occupy those areas (Kemp et al. 2004).

Evidence is strong that reduced nutrient loads can generate SAV recoveries. Submerged aquatic vegetation goals have been met in some tributaries of the Chesapeake Bay, including the Potomac River, the adjoining Mattawoman Creek, and the upper Patuxent River, which were linked to reductions in nitrogen and phosphorous loading associated primarily with wastewater treatment plant upgrades (Boynton et al. 2008, 2014, Orth et al. 2010, Ruhl and Rybicki 2010). Additionally, upper Chesapeake Bay SAV, which includes the Susquehanna Flats bed, surpassed the goal for that region in the early 2000s, likely due to decreased nutrient loading from the Susquehanna River (Orth et al. 2010, Gurbisz and Kemp 2014, Gurbisz et al. 2016). Submerged aquatic vegetation recoveries following reductions in nutrient loads have occurred in other systems including Tampa Bay, Florida, USA (Greening et al. 2014), and Mondego estuary, Portugal (Cardoso et al. 2010). These case studies strongly suggest that load reductions and associated water clarity improvements that accompany the TMDL can facilitate SAV restoration.

Although SAV is expected to expand its range in response to improved water quality attributable to the TMDL, the future of the dominant SAV eelgrass, in particular, remains uncertain due to its intolerance to heat. The extent to which Chesapeake Bay SAV species will acclimate or adapt to changing temperature and salinity (Maxwell et al. 2014), or if new species will gradually colonize as conditions change, is largely unknown. Evidence for a bumpy transition to new species comes from a recent review of aquatic plant responses to climate change, which concluded that aquatic plants do not have the capacity to migrate quickly enough to escape climate change impacts (Bornette and Puijalon 2011). Given that circumstance, the TMDL may elevate chances for a smooth transition by extending the duration over which resident species can persist or adapt rather than succumbing to climate change stresses.

The fish resilience story

Case studies provide evidence that the TMDL could encourage broader fish distributions and promote other aspects of resilience. A well-known example that demonstrates how managing stress can promote recovery, despite multiple co-occurring stresses, is the decline and recovery of the striped bass population in Chesapeake Bay. Catches of this valued recreational and commercial species peaked in 1973 and then declined 80–90% in the years before a fishing moratorium was instituted in 1984 (Houde 2011: fig. 6 therein). The moratorium led to a sustained recovery of the fishery. Depletion was largely attributed to overharvesting; however, a period of sustained low juvenile production during the collapse (1970s–1980s; Fig. 3) suggested that water quality contributed to the decline (Goodyear 1985, Hall et al. 1993). Evidence from several smaller East Coast estuaries further indicated that water quality contributed to the decline of striped bass nurseries. The relationship between water quality improvements and nursery recovery was more apparent in these smaller systems than in Chesapeake Bay (Boyle 1979, Chittenden 1971, Daniels et al. 2005, Woodland et al. 2009).

The rebound of the striped bass fishery in the Chesapeake Bay suggests that conservation of water quality in diverse functional nurseries has likely contributed to resilience by promoting capacity for striped bass to recover from stresses. The recovery was facilitated by the response diversity offered by nursery grounds in multiple locations (Kraus and Secor 2005, Schindler et al. 2010). In Maryland, data from four principal nurseries (D. H. Secor, *unpublished data*) suggest that these nurseries provide a level of buffering to maintain overall juvenile production under heterogeneous stresses. Significant correlation in recruitment (a measure of reproductive success) was found within two pairs of the four nursery systems: the Choptank and Nanticoke Rivers (Group 1) and the Potomac River and Upper Bay (Group 2), but recruitments in the two groups responded independently (Fig. 3). Prior to the striped bass population crash in the 1970s, Group 2 provided most of the recruits. During the sustained period of depressed abundance, Groups 1 and 2 were similarly diminished, but during the recovery, Group 1 provided the majority of recruits, suggesting that faster recovery in some areas of

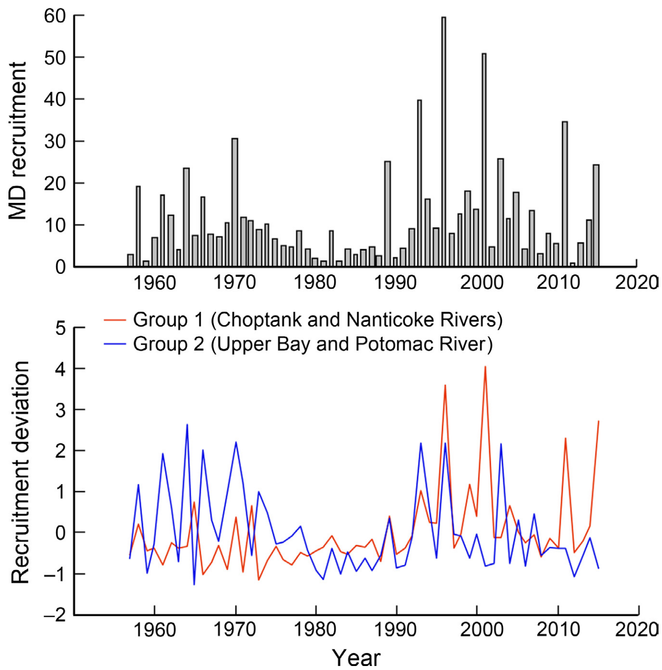

Fig. 3. Overall recruitment levels (top panel) and recruitment deviations for striped bass in two groups of spawning tributaries in Chesapeake Bay. Inversely correlated recruitment levels across the two geographic areas suggest that both areas are needed for fish resilience. Groups were identified through principal components analysis for 1957–2015 Maryland juvenile seine survey data (Maryland Department of Natural Resources Fisheries, *public communication*). Recruitment deviations on the *y*-axis are factor scores from the eigenvalues associated with Group 1 and Group 2.

the bay jump-started the recovery of the bay-wide population. Importantly, years with the highest recruitments typically received substantial contributions from only one of the two nursery groups.

Does this story apply to other fishes?

Many fish species are sensitive to hypoxia and their spatial distribution would be expected to expand with the Chesapeake Bay TMDL due to increases in types and sizes of available habitats. Because the magnitude and timing of hypoxia events vary among locations, mosaics of habitats can promote resilience to hypoxia and other recurring stresses through response diversity (Kraus and Secor 2005). Examples of this response diversity include resident and migratory ecotypes of white perch and American eels (dominant fish species in the Chesapeake Bay), which have varying migrations into brackish and freshwater that buffer responses to stress at the population level (Kraus and Secor 2004, Secor 2015).

Will the TMDL improve spatial distribution of fish?

As the example above suggests, excess nutrients limit spatial distribution of fish by forcing them out of hypoxic areas and other marginal habitats and concentrating them in fewer habitats. Because the TMDL was designed to improve dissolved oxygen conditions, many fish species and communities are expected to benefit through expanded spatial distributions. That expanded spatial distribution promotes resilience to the substantial interannual variation in the volume and spatial distribution of hypoxia within both shallow and deep waters (Bell and Eggleston 2005, Tyler et al. 2009). For example, Buchheister et al. (2013) observed that numerous demersal fish species (fish living on or near the bottom of the water column) strongly avoided dissolved oxygen below 4 mg/L in Chesapeake Bay and in years with the most extreme hypoxia, their distributions shifted downbay to areas with better dissolved oxygen conditions.

The effect of the TMDL on habitat distribution may be particularly important in nursery habitats in tributaries, which are critical to promoting recovery potential of fish populations. A well-known example of lost nursery habitat attributable to excess nutrients was a region of the upper Hudson River estuary known as the "Albany Pool" (Boyle 1979). During most of the 20th century, sewage and industrial waste in the Albany Pool led to summertime anoxia in a region that had historically supported nurseries for diadromous species such as striped bass, American shad, and sturgeons. This region of the Hudson River did not support juveniles and young fish again until new sewage treatment plants reduced nutrients and restored normal dissolved oxygen levels. Similar recoveries after nutrient reductions have been documented in New York Harbor and the Penobscot, Delaware, and Potomac River estuaries (Chittenden 1971, Daniels et al. 2005, Woodland et al. 2009).

Improved spatial distribution of fish may also play a role in resisting disease. Hypoxia, combined with temperature stress, is projected to increase risk of disease under climate change (Roessig et al. 2004). In recent years, a potentially lethal bacterial disease (mycobacteriosis) has become common in striped bass, causing lesions on more than 50% of four- to five-year-old bass in the Chesapeake Bay (Overton et al. 2003, Kaattari et al. 2005, Gauthier et al. 2008, Houde 2011). As with other stressors, the infection rate differs among areas of the bay, emphasizing the benefit of maintaining multiple productive habitats.

In addition to improving resilience through increasing fish distribution, the TMDL is likely to promote resilience through improved health and reproductive potential for some species. Even though fish are quite abundant in the bay, the proportion of large fish has declined in recent decades for some species and certain species have shown dramatic declines (CBFEAP 2006). For some species that have declined, water quality has played a role. Hypoxia of long duration can kill sessile organisms such as oysters and its sub-lethal effects may reduce resilience to other stressors. Recent research suggests that hypoxia reduces the immune response of oysters, rendering them more susceptible to disease (Breitburg et al. 2015*a*), a factor that may have played a role in the near-complete collapse of oysters in the bay (Wilberg et al. 2011). In another example, hypoxia influences egg characteristics of Atlantic croaker and acts as an endocrine disruptor that affects survival of young (Tuckey and Fabrizio 2016), with the potential to cause widespread failure of reproduction (Wu et al. 2003, Thomas et al. 2006, 2007, Landry et al. 2007).

Summary and Discussion

The accumulated scientific knowledge strongly indicates that the ability to adapt to stressors such as higher water temperatures, salinity variability, storm damage, or the introduction of invasive species and diseases will be enhanced once the bay ecosystem is released from high eutrophication stress. Yet, the complexities of estuarine responses to nutrient and sediment loading suggest that the benefits of implementing the TMDL will not simply lead to a gradual improvement in all conditions as these inputs decline (Boynton et al. 1983, Cloern 2001). Rather, the benefits of having lessened the stress of eutrophication may not fully manifest until the system is challenged to withstand or recover from new or unusual levels of stress associated with climate change or other novel pressures.

The TMDL will reduce controllable stressors, thus creating capacity to absorb uncontrollable or novel stressors and stay within critical thresholds of ecosystem condition. This result is especially important to maintain or improve fish health and conserve abundance, because species or ecosystem resilience is an emergent property of multiple biophysical processes and their interactions. We have suggested two measures of resilience, SAV extent and fish distribution, that could serve as integrative indicators to represent long-term public benefits. We chose SAV extent because it is a marker of the level of ecosystem stress and the ecosystem's capacity to absorb some stressors. We chose fish spatial distribution because it demonstrates capacity for fish to manage risk when stresses are heterogeneous in space. We have summarized literature that supports these choices to document why they may serve to represent long-term resilience and, through that mechanism, support public values associated with maintaining fish production and overall health of the bay aquatic ecosystem far into the future.

Overcoming the limits of valuing ecosystem services

Ecosystem service valuation has great potential to guide ecological restoration investments to balance multiple goals. Yet, it is also clear that monetary valuation alone will not fully represent many public goals and will likely omit benefits associated with avoiding low probability, but high consequence, tipping points. Adopting ecosystem services in decision-making, therefore, requires thoughtful application of multiple benefit assessment tools to reveal which actions are in the public interest.

The Chesapeake Bay TMDL case study demonstrated that estuarine restoration can generate many ecosystem service benefits, only a small portion of which can be monetized due to data and knowledge constraints. The monetary values that have been estimated for benefits of water quality improvement throughout the watershed are substantial and include property value enhancements, fishing and aesthetic benefits, and non-use (intangible) values for the aquatic ecosystem. The monetized co-benefits (supplemental benefits not related to estuarine water quality) of the conservation practices used to achieve goals include health, safety, other effects of climate risk reduction, and hunting opportunities. Our summary of studies and new analysis of HAB changes quantified and described the potential for additional ecosystem service benefits from health and commercial business effects.

While additional benefits could be monetized with further effort, they are unlikely to address a major motivation of restoration, which is to capture the intangible value people derive from reducing the probability of the system reaching a degradation tipping point. It may be tractable to estimate the extra value that people would pay for more *reliable* aesthetics, hunting, fishing, or other benefits from the Chesapeake Bay. However, it is not possible to robustly estimate the change in probability of a system reaching a tipping point with implementation of the TMDL. As a result, any monetary valuation of the TMDL is likely to fall well short of representing total benefits.

Rather than ignoring the intangible benefits of improved system resilience, we have proposed two types of quantitative, non-monetary metrics to indicate level of benefits. Although these benefit indicators cannot be directly compared to costs in a CBA, they can, nonetheless, guide appropriate investments and enhance cost-effectiveness by helping decision makers optimize relevant outcomes. For example, using dissolved oxygen levels to judge appropriate investment in water quality improvement potentially ignores the ability of some fish to adapt to hypoxia. In contrast, fish distribution metrics more directly indicate when stressors have exceeded adaptive capacity. Thus, using non-monetary benefit metrics to reflect the degree to which restoration actions promote resilience and lessen risk of system collapse can promote restoration choices that support a full range of social values.

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