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ADDITIONAL BENEFICIAL OUTCOMES OF IMPLEMENTING THE CHESAPEAKE BAY TMDL:

Quantification and description of ecosystem services not monetized

Prepared for
Naomi Detenbeck, Brenda Rashleigh
US Environmental Protection Agency
ORD / NHEERL
Atlantic Ecology Division
Narragansett, RI

Prepared by
Lisa Wainger[1], Jennifer Richkus[2], Mary Barber[2]
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[1] Oneida Total Integrated Enterprises, LLC, Oak Ridge, TN and
University of Maryland Center for Environmental Science, Solomons, MD
[2] RTI International, Washington, DC

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Developing and Refining Ecosystem Services Approaches Relevant to Restoring the Chesapeake Bay

Notice

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Abstract

Over the last 60 years, the Chesapeake Bay water quality and seagrass beds have diminished to the point that the system is less able to support abundant crabs and diverse fish, feed waterfowl, and produce safe recreational opportunities. Further, the long-term resilience of the bay is in question as climate change, invasive species, and emerging diseases create novel stressors on this already struggling system. As a result of improved pollution management, the bay has improved in some respects in the most recent decades, and the implementation of Chesapeake Bay Total Maximum Daily Load (TMDL), which sets pollution caps, has the potential to further ameliorate problems and provide a wide variety of benefits to society.

This effort supplements other efforts to monetize a range of benefits of the TMDL and explains why some key benefits that motivated the TMDL will not be included in the final dollar estimate. The purpose of this report is to provide quantification and description of the magnitude of improvements to conditions in the bay that cannot be monetized but can be linked to human welfare. We evaluate benefit indicators (e.g., reductions in disease-causing organisms), but we are not demonstrating benefits in the strict sense because we have not evaluated what people would have been willing to pay to achieve these benefits. Yet, non-monetary benefit indices are used routinely to establish cost-effectiveness of management actions and can enrich the context in which the benefit-cost results are considered.

We analyze and synthesize existing scientific literature and data to quantify and describe how the practices that the bay states have proposed to meet the TMDL could positively affect selected ecosystem services produced by the Chesapeake Bay system. In support of public health, food supply, and recreation, we estimate that the TMDL practices collectively have the potential to decrease disease-causing pathogen loads to the bay by at least 19-27%, reduce human exposure to West Nile Virus, and reduce incidence of harmful algal blooms. Perhaps most significantly, implementing the practices to meet the TMDL would also promote benefits derived from enhancing or maintaining bay ecosystem resilience. We describe how resilience to multiple stresses, including climate change effects, is fostered by the regrowth of submerged aquatic vegetation, increased fish diversity, and reduced hypoxia. These changes would be expected to promote a system that recovers more readily from disturbance and avoids tipping points that could shift the system to an undesirable state.

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Introduction

The US Environmental Protection Agency (EPA) is conducting a benefit-cost analysis (BCA) of the Chesapeake Bay Total Maximum Daily Load (TMDL), even though the enabling legislation does not require that TMDLs meet a benefit-cost criterion and, more commonly, TMDLs are set using other criteria (Keplinger 2003). What is driving this unusual effort is a desire to determine whether the money that is, and will be, invested in reducing nutrients and sediments to the bay generates net benefits, and therefore, is in the public interest. BCA is a powerful tool for promoting the sound use of our public resources because it allows us to evaluate economic efficiency by evaluating whether the costs or sacrifices of a proposed action are more than offset by the improvements in welfare that it creates.

However, the monetized benefits of a BCA never provide a complete picture of benefits of any action and this will be true for the BCA of the Chesapeake Bay TMDL. Despite its strengths for establishing the economic efficiency of an action, BCA cannot incorporate everything society values, due to information gaps and the inability of economic analysis to fully incorporate the full suite of public concerns. For example, a well-known drawback of BCA is that it does not address how fairly benefits and costs are distributed among different groups and may not fully consider effects on future generations. People are concerned about equity among current and future generations and are averse to some types of risk in ways that are not captured by the expert assessments of the expected incidence of morbidity and mortality (Sunstein 1996; Slovic 2000a). Thus, BCA is intended to *support* decisions, rather than replace the judgment of decision makers who may need to consider many other factors (Freeman et al. 2014 p. 10).

Of particular relevance to environmental decision-making are subtle issues related to what can be valued in BCA. Namely, some types of environmental benefits are more readily monetized than others, and the benefits that are monetizable may not be the primary benefits that are motivating the regulatory or other action. The "use" services of the environment represent things people use (directly or indirectly) from ecosystems, such as resource extraction (e.g., timber harvest, commercial fishing), outdoor recreation (e.g., recreational fishing, wildlife watching) and hazard mitigation (e.g., property protection from flooding). Use services of the environment, such a fishing, hunting, and food production, have been monetized numerous times in multiple settings. However, the "nonuse values" or "passive uses" of ecosystems represent things people value just because they like it or think it should be protected, such as values people hold for protecting natural elements of their heritage (National Research Council [NRC] 1999). Nonuse values tend to be more time consuming and controversial to monetize and thus are measured less frequently than use values.

What the Chesapeake Bay BCA is likely to reveal is that the TMDL's caps on nutrients and sediments are expected to generate many types of benefits induced by the expected water quality improvements. In addition to direct water quality effects, the management actions used to prevent nutrients from reaching water bodies (e.g., planting forested riparian buffers) are anticipated to generate many co-benefits such as improved recreational opportunities, aesthetics and air quality (Van Houtven 2011; Wainger et al. 2013). Yet, when it comes to complex ecosystems, such as the Chesapeake Bay, only a fraction of benefits from water quality improvements are captured as part of monetary valuation due to limitations on data, scientific understanding, and analytic resources.

The process of quantifying and describing benefits that cannot be monetized is an inherent part of conducting BCA and is the recommended practice for federal agencies preparing materials for regulatory review (U.S. Office of Management and Budget 2003). Yet, effects that have not been monetized cannot be included in a benefit-cost ratio, which can serve as a litmus test for taking an action. For this reason, some consider the non-monetized potential benefits to be uncounted in policy decisions.

Given that the TMDL decision context is not required to make decisions based on a benefit-cost ratio, the purpose of this report is to provide quantification and description of the magnitude of improvements to conditions in the bay that cannot be monetized but can be linked to human welfare. We evaluate benefit indicators (e.g., reductions in disease-causing organisms) but we are not demonstrating benefits in the strict sense because we have not evaluated what people would have been willing to pay to achieve these benefits. Yet, non-monetary benefit indices are used routinely to establish cost-effectiveness of management actions and can enrich the context in which the benefit-cost results are considered.

Management actions

It is difficult for many to imagine today, but shallow regions of the bay were, in relatively recent history (1950s and prior), covered by extensive submerged aquatic vegetation (SAV) beds that were rich with plant and animal life (Kemp et al. 1984; Borum et al. 2012). Throughout the bay, diverse finfish species were abundant and oyster beds were consistently producing healthy oysters. The bay has shown some improvements in water quality in recent years (Kemp et al. 2005), but is still under stresses that limit some benefits and make it vulnerable to climate change and other emerging threats.

To restore the Chesapeake Bay, jurisdictions in the watershed are taking actions to meet the Chesapeake Bay Total Maximum Daily Load (TMDL). A TMDL is a quantitative pollutant cap that is established by a state to enable a water body to attain its designated use, as required under Section 303(d) of the Clean Water Act, when a water body is in non-attainment. The legislation requires the states to develop implementation plans, known as watershed implementation plans (WIPs) to achieve the caps. In this study, we quantify and describe a suite of benefits that are likely to be attributable to these actions but that could not be readily monetized.

The Chesapeake Bay TMDL sets yearly caps on nitrogen, phosphorus and sediment entering the Chesapeake Bay. We will refer to these caps as a single TMDL. The caps are intended to restore the aquatic habitat of the Chesapeake Bay, which, in turn, will support human uses such as safe recreation and food harvesting and promote the future use and maintain the heritage of this valued resource. The work presented here to quantify benefits complements other on-going work to monetize many benefits of the program (including commercial and recreational fisheries, property value enhancements, avoided costs of dredging, water treatment and trout stocking, and value of water quality improvements in the bay and freshwater tributaries) but reveals why some key benefits that motivated the TMDL will not be included in the final dollar estimate.

Measuring Potential Benefits without Economic Valuation

Monetizing or valuing the benefits of an action, like the TMDL, to restore environmental quality involves 1) establishing the effectiveness of the action for changing ecological structures and processes in desirable ways; 2) relating those ecological changes to outcomes that affect well-being; and 3) establishing how much people value those changes, after considering their ability and willingness to adapt, substitute, or trade off goods and services. All of these steps must be possible in order to estimate values of a change.

Conducting these valuation steps involves a great deal of effort and can easily break down due to lack of appropriate data or understanding. Ecological effectiveness is measured as the change in a biophysical condition, relative to a no-action baseline, based on available field studies and models. Socially-relevant outcomes that reflect well-being are usually developed with models that accept the changes in basic ecological structures and process (e.g., number and types of trees, water quality) and then convert those inputs to metrics that are easier for people to value (e.g., abundance of game fish, water clarity, or bird abundance and diversity).

The value of these changes is analyzed in terms of the quantity of goods and services (i.e., dollars) that people would be willing to give up in order to achieve one or more changes. Those values are measured by evaluating markets for commercial goods, analyzing behavior of recreators, homebuyers and others, and by implementing survey instruments. Values depend on the number of affected beneficiaries and the importance of the change to beneficiaries.

When any step in this process cannot be rigorously quantified, we quantify or describe the evidence for benefits in terms of socially relevant outcomes or ecological effectiveness, as appropriate to the data conditions. To ensure that such changes are connected to the action, we seek to demonstrate changes in these variables from a no-action baseline.

Steps used in analysis

The detailed methods used to quantify or describe changes in ecosystem services depend on the ecosystem service being analyzed. For each service, we do not follow the three valuation steps mentioned above exactly, but rather, adjust that approach to quantify or describe changes in terms most relevant for understanding the magnitude of potential benefits for each service. The steps, which are implemented only when feasible, are:

1. Relate ecological changes to potential well-being by identifying the specific benefits that may be created by implementing the TMDL in terms of ecosystem services.
2. Establish effectiveness of the action by estimating the magnitude of change in a beneficial outcome (basic biophysical or benefit-relevant metric) as a result of all management actions expected to be implemented specifically as part of the TMDL and omitting practices that were otherwise required.
3. Suggest an order of magnitude of benefits by estimating the number of potential beneficiaries.
4. Suggest an order of magnitude of benefits by estimating a range of value change per beneficiary.

If all four steps were possible to conduct, the benefit could be monetized through benefit transfer methods. However, in all cases presented here, one or more of the steps could not be completed to generate quantitative outputs. Some of these effects might be monetized with additional resources.

Defining socially-relevant outcomes

We express socially-relevant outcomes in terms of ecosystem services, which have been defined in multiple ways (Heal 2000; Brown et al. 2007; Costanza 2008; Fisher et al. 2009). Here, we use the term to mean the ecological outcomes that matter to people and for which they can express meaningful preferences. For example, ecosystem services are expressed as "increases in fish catches" or "fewer days of beach closures" rather than in terms of basic biophysical processes, such as "changes in nutrient cycling." The general principle is that the closer an ecosystem service outcome metric is to a familiar good or service, the easier it is to understand likely benefits of that change. However, this definition does not preclude using basic ecological metrics to reflect the nonuse values that people hold, including restoring the health and integrity of an ecosystem or preserving it in good condition for future generations.

We developed a list of potential benefits of the TMDL through literature reviews and conversations with scientists, government officials, representatives of non-governmental organizations and other stakeholders. The list included potential benefits in terms of public health, waterfront or water-associated businesses including commercial fishing, agricultural producers, recreation (fishing, boating, swimming, wildlife watching), homeowners, heritage tourism, and quality of life effects throughout the watershed including social relationships and sense of place. These conversations informed this analysis and the separate (monetary) valuation analyses.

Ecosystem Service 1. Public Health Protection

Under the category of public health protection, we evaluate the ecological effects of reduced pathogen loads to surface waters, reduced incidence of harmful algal blooms (HABs), and increased area of wetlands and forests, associated with increases in bird diversity. Reduced toxin exposures (other than HABs) are another outcome with the potential to affect human health, but the risks to human health are not well understood, so we only relate this outcome to aquatic organism health in the next ecosystem service of Ecosystem Resilience, because recent research has synthesized what is known about these risks.

Pathogens

The WIPs have the potential to reduce pathogens that can cause disease through water contact, seafood consumption, or mosquito bites. Water contact and seafood consumption carry risks of gastrointestinal illnesses and ear and skin infections, among other illnesses, when they are contaminated by bacteria and viruses. Mosquitoes can carry multiple illnesses, but here we focus on the potential for WIP practices to reduce incidence of West Nile Virus by lowering transmission rates as a result of increases in natural land cover and bird diversity.

Current conditions and effects on ecosystem services

A variety of potentially dangerous pathogens enter water bodies when runoff entrains and accumulates fecal matter from humans and domestic and wild animals along its flow path. Agricultural and urban settings are the primary areas where delivery of pathogens via runoff has been observed (Howell et al. 1995; Kistemann et al. 2002; Shehane et al. 2005; Coulliette & Noble 2008). The amount of pathogen delivery depends on many factors including precipitation patterns, concentration of fecal material on land, soil and aquifer properties, location of fecal material relative to water bodies of different sizes, and patterns of land use that affect delivery of pathogens to water bodies (US EPA 2001; Vann et al. 2002; Thurston-Enriquez et al. 2005).

Water that interacts with fecal matter can contain diverse pathogens such as *Vibrio, E. coli* (pathogenic), *Shigella, Rotavirus, Yersinia, Cryptosporidium* and *Giardia* (Savichtcheva & Okabe 2006) that have been linked to gastrointestinal illnesses, skin infections, fevers and other human health concerns (Vann et al. 2002). Humans become exposed to these pathogens through direct contact with water, consumption of shellfish, or handling of animals that are contaminated with the waterborne pathogens. Because of the potential for creating illness, government officials respond to potential water contamination by closing beaches and waterways to recreators and closing shellfish beds to commercial and recreational harvest. Such decisions can be based on whether pathogens are detected at all, for the most dangerous pathogens, or whether they reach threshold concentrations associated with illness.

The level of concern for pathogens in the bay and watershed is evident from the actions that officials have taken to address them. A variety of streams have local TMDLs (distinct from the bay-wide nutrient and sediment TMDL) to alleviate chronic pathogen impairment in the three states that make up the majority of the watershed.[1] Over 9000 stream miles in Virginia (US EPA 2013b), over 4000 miles in Maryland (Maryland Department of the Environment [MDE], 2012), and 190 miles in Pennsylvania (US EPA 2013a) are impaired for *E. coli* and fecal coliform. In addition to streams not meeting designated uses, 176 Virginia shellfishing areas are indefinitely

[1] The watershed also includes the District of Columbia and portions of West Virginia, Delaware, and New York.

closed due to elevated fecal coliform (VDH 2012; US EPA 2013b) and 77 shellfish beds are occasionally or permanently closed in MD (MDE 2014). Virginia also had 29 days of beach actions (notifications and closure days) out of a total of 6,900 beach days (open days multiplied by number of beaches) in 2012 (VDH 2011a; US EPA 2012b) and Maryland had 139 days of beach actions out of a total 6,501 beach days in 2012 (US EPA 2013c, 2013d) (**Table 1**).

The strongest evidence of harm from pathogens may come from cases of reported illness that can be linked to waterborne pathogens. We evaluated the reported data for illnesses that have the potential to come from untreated water bodies but report total cases of these illnesses from treated and untreated sources because we do not have data to separate illnesses by source water (**Table 2**). Virginia had an annual average of 902 reported cases of gastrointestinal illnesses over the period 2004-2013 that were associated with waterborne pathogens (VDH 2014). Maryland had an annual average of 621 cases of reported gastrointestinal illnesses related to waterborne pathogens from 2005-2012 (Maryland Department of Health and Mental Hygiene 2013). We analyzed MD and VA because they are the states with bay shoreline, but other swimmable water bodies will also be affected by pathogens.

These data on illnesses must be interpreted cautiously since only a fraction of cases are likely to be associated with swimming in the bay or its tributaries. Data collected by the CDC using different reporting criteria found that 70% of reported illnesses due to waterborne pathogens were from pools or other treated water, and 30% were from open (untreated) water, such as lakes and oceans (Hlavsa et al. 2014). On the other hand, many more cases of gastrointestinal illnesses are likely to occur than are reported (Hlavsa et al. 2014). In addition to gastrointestinal illness, anecdotal information suggests that skin rashes and infections due to water contact are not an uncommon ailment in the Chesapeake Bay (Kobell 2011, 2013) particularly in the warmest months. These are not usually reported, although they have been documented elsewhere (Wade et al. 2010). Thus, the numbers in Tables 1 and 2 are provided to suggest a potential order of magnitude of illnesses caused by pathogens but are not an accurate accounting due to data limitations.

Table 1. Days of Beach Closure in Maryland and Virginia (2007-2012)

No. of days under a beach action (monitored beaches)	2007	2008	2009	2010	2011	2012	Total	Average Annual
VA beach	63	41	51	81	69	29	334	**56**
MD beach	248	61	133	351	244	139	1176	**196**

Source data: VDH (2011a) and US EPA (2012b).

Table 2. Reported Diseases due to Pathogens in Water Bodies in Maryland and Virginia (2004-2013)*

Maryland	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	Avg
Cryptosporidiosis	na	33	20	36	54	43	42	70	86	na	**48**
Giardiasis	na	210	256	269	284	277	262	291	239	na	**261**
Listeriosis	na	19	28	15	17	14	11	19	16	na	**17**
Shiga - toxin producing *E. coli* (STEC)	na	75	131	85	129	91	107	71	75	na	**96**
Shigellosis	na	103	139	117	137	370	131	94	222	na	**164**
Vibriosis	na	25	31	25	33	34	45	35	53	na	**35**
Total		**465**	**605**	**547**	**654**	**829**	**598**	**580**	**691**	na	**621**

Virginia	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	Avg
Cryptosporidiosis	66	77	71	90	81	86	109	140	144	144	**101**
Pathogenic *E. coli*	62	111	168	165	241	156	149	123	81	109	**137**
Giardiasis	563	602	514	582	432	503	512	290	272	278	**455**
Listeriosis	27	17	20	16	17	16	13	15	18	29	**19**
Shigellosis	167	134	120	200	310	198	145	107	91	115	**159**
Vibriosis	20	25	32	33	29	29	40	30	41	42	**32**
Total	**905**	**966**	**925**	**1086**	**1110**	**988**	**968**	**705**	**647**	**717**	**902**

* Totals include illnesses due to treated (e.g., pools) and untreated (e.g., estuaries) water bodies, and the majority of these illnesses are likely from treated water bodies. One study suggested 70% of cases are from treated waters, however, that study was not conducted on these data so they are unadjusted (Hlavsa et al. 2014).
Source data: VDH (2014) and Maryland Department of Health and Mental Hygiene (2013).

Potential improvements from implementing the WIPs

Pathogens are associated with wildlife, livestock, manure handling, pets, failing septic systems, and wastewater treatment plants. The WIP efforts include the following types of actions that have the potential to affect the delivery and concentration of these waterborne agents of disease (categorized by source):

1. **Agricultural**: pasture and grazing management, nutrient management on crop fields, livestock waste management, restricted stream access, plantings and other structural practices to reduce nutrient and sediment runoff.
2. **Urban**: detention and retention ponds, impervious surface reduction, street sweeping, forested riparian buffers, bioswales, and afforestation.
3. **Septic**: connecting septics to sewers, septic pumping, and on-site septic upgrades.
4. **Wastewater Treatment Plants**: new and enhanced treatment of municipal waste.

The actions prescribed by the WIPs have the potential to reduce pathogens either by directly preventing pathogens or fecal matter from being washed into waterways or by reducing the amount of fecal matter available to runoff (Mallin et al. 2000). Efforts to enhance the water-holding capacity of WWTPs will prevent some sewer overflows, which are another major source of pathogen inputs to water bodies, but we do not evaluate those effects here, particularly since many systems will be required to make upgrades to prevent overflows to comply with state and federal regulations that are required external to the TMDL. Here we focus on the effect of agricultural best management practices (BMPs) and selected urban stormwater (SW) practices.

In order to compare pathogens with and without the TMDL, we compared conditions for two scenarios that have been developed by the US EPA Chesapeake Bay Program (CBP) to represent a "without TMDL" baseline and a "with TMDL" condition. The baseline "without TMDL" scenario uses population; land use; point sources; and the number, location, and size of point and nonpoint source nutrient and sediment control practices in the Chesapeake Bay Watershed that were present in 2009. It also includes additional nutrient and sediment control measures that were not in place in 2009, but that would be expected to be implemented by 2025 to meet non-TMDL requirements (e.g., requirements associated with the Clean Air Act and new stormwater requirements). The "with TMDL" scenario captures the actions expected to occur as part of the WIP implementation targets for 2025 (as specified in the Phase II WIPs). These actions include non-point source controls placed on agricultural land, some urban stormwater practices, and upgrades to wastewater treatment plants. To allow comparison with the "without TMDL" scenario, the "with TMDL" scenario assumes that population, land use, and the point and nonpoint sources in the bay watershed are the same as in 2009. The WIPs provide types and quantities of BMPs that are expected to be implemented to meet the nutrient caps, and the CBP modelers have estimated their placement in the watershed.

We conducted four steps to estimate pathogens for these two scenarios for the Potomac River watershed and the Chesapeake Bay. First, we estimated the pathogen reductions (as percent removal) that were likely to be associated with the BMPs specified in the Phase II WIPs. Second, we estimated the total baseline pathogen load by multiplying an average per acre pathogen load for major land uses (derived from Vann et al. 2002) by the land use acreages of the CBP baseline scenario to generate a total baseline pathogen load at the edge of stream. Third, we applied an average downstream delivery factor derived from Vann et al. (2002) to capture the attenuation of pathogens that occurs in stream prior to delivery in the main channel. Fourth, using the BMP reduction efficiencies and downstream delivery factors, we estimated the total reduction of pathogen loadings (*E. coli* and fecal coliform) to edge of stream and main channel due to TMDL implementation.

Pathogen Reduction Efficiencies for BMPs

We conducted a literature review to investigate the potential efficiency of agricultural, urban and septic BMPs in reducing pathogens at edge of field or edge of (small) streams. The literature review was conducted by employing each of the Chesapeake Bay Best Management Practices (BMP) within the state Watershed Implementation Plans (WIPs) and pathogen-related terms as keywords (e.g., "restricted stream access" or "riparian buffer" and "fecal coliform" or "*E. coli*" or "bacteria") using Google Scholar, EBSCO and Google as search engines. Pathogen reduction estimates from peer-reviewed journal articles, documentation prepared by state agencies for compliance with TMDLs, and best practice guidance reports from state agencies and universities were all considered for the purposes of this paper. Data were only included in the analysis when they matched the BMP and pathogen being evaluated.

Table A.1 (Appendix A, page 43) summarizes the removal efficiency data found. Fecal Indicator Bacteria (FIB), which includes fecal coliform and *E. coli,* were most often evaluated as a surrogate for a variety of pathogens (Marion et al. 2010; US EPA 2012a). Therefore, we averaged the efficiencies found in the literature by practice for FIB to use in the analysis.

Agricultural practices showed a range of efficiencies at removing fecal coliform and *E. coli* (28-100%), but the average performance per practice was above 50% for all practices except wetland and stream restoration. Studies also showed high efficiency of grassed buffers at removing cryptosporidium (93–99%). Stormwater practices showed a wider range of removal efficiencies (-6%–99%) than agricultural practices when looking across the range of practices. However, a few practices were responsible for the cases of low performance (bioswales, street sweeping and septic pump outs). The majority of practices had average efficiencies of 48% or greater.

Baseline Pathogen Loads

To estimate the baseline pathogen load, we required an understanding of pathogen sources and deliveries to water bodies for the given level of management practices implemented in the baseline scenario. A study of the portion of the Potomac River basin that lies above the fall line[2] provided the best available information about how pathogens were being produced, transformed, intercepted and, finally, delivered downstream (Vann et al. 2002). That study estimated average annual edge of stream pathogen loadings for a period that roughly corresponded to 2000-2010. The 2010 scenario was a projection of land use and population changes expected to occur by 2010 combined with 2000 estimates of non-point source BMPs and wastewater loads, and 2010 estimates for septic conditions. We use the 2010 model results as if they occurred in conditions equivalent to the CBP 2009 baseline scenario.

Vann et al. (2002) estimated loadings by land use type, using models similar to those of the CBP but modified to include pathogen movement and transformation and a wide variety of data sources on fecal sources. Data on livestock, geese, deer and human populations; NPDES and wastewater emissions; and other sources were used to inform modeling of pathogen loads by

[2] The fall line in the Chesapeake Bay watershed is a geomorphic feature marked by a steep drop in elevation that occurs where the Piedmont and Coastal Plain geophysical provinces meet. It roughly corresponds to the division between non-tidal waters (above) and tidal waters (below).

land use type. The CBP watershed model was adapted to include bacterial fate and transport, and the loads by land use were calibrated using pathogen concentrations measured at monitoring stations, primarily within the main channel of this major Chesapeake tributary (**Table 3**). The model was also combined with data from surface water intakes to estimate the downstream delivery factors for the Potomac.

To use the Vann et al. (2002) results to estimate baseline loads for the land uses in the entire Potomac watershed and the bay watershed, we converted the edge of stream loads to per acre loadings per land use type (**Table 3**). We only evaluated the three land uses being modified by the BMPs used in the analysis, as described below. We then multiplied the per acre loads for acreages of pasture, cropland and urban for the baseline scenario to estimate baseline loads.

This method relies on transferring results of sophisticated models for the upper Potomac to two different scales of analysis (Potomac and entire bay watershed) to provide a rough estimate of TMDL effects at these scales. Clearly, using data from a portion of the Potomac to represent either the whole Potomac or the entire bay requires making considerable assumptions about the similarity of patterns and processes at these two scales. We have greater confidence in the Potomac results since the whole Potomac would be expected to be more similar to the modeled area than the bay as a whole. The Potomac may be a reasonable model for the entire bay since it makes up over one-fifth of the bay watershed and has proportions and distribution of land use types that are similar to the entire bay watershed. However, the Potomac differs from the bay in that it has slightly more urban land and pasture and less forest (**Table 4**), and BMPs were applied in different proportions to the whole bay, as shown in the results.

Table 3. Modeled Loadings per Land Use Source in the Upper Potomac River Basin (above the Fall Line)

Loading Type/ Land Use	Edge of Stream Delivery of Fecal Coliform (cfu/yr)‡	Edge of Stream Delivery per Acre (cfu/ac/yr)	% of Edge of Stream Loading Delivered Downstream (%) *
Forest	3.9E+14		18%
Crop	6.0E+16	5.18E+10	25%
Geese[†]	8.6E+13		33%
Pasture	3.2E+17	3.88E+11[∫]	28%
Feedlots	6.3E+16		24%
Cattle[†]	1.0E+16		21%
Point	3.1E+13		23%
Septic	3.2E+14		28%
Urban	2.2E+16	1.82E+10	27%
Total	4.7E+17		28%

All data derived from Vann et al. (2002)

† *Geese* and *Cattle* land uses are an estimate of deposition of feces directly into water bodies.

‡ Pathogens were measured as fecal coliform in colony forming units per year (cfu/year).

* Proportion delivered downstream was calculated with mass balance equations, based on data provided by Vann et al. (2002).

∫ Land uses were combined for the delivery estimates per acre because acreages were not reported separately for these land uses.

Table 4. Land Use Composition of Potomac River Basin and the Chesapeake Bay Watershed

Land Use	Potomac River Basin (acres)	Potomac River Basin Land Use	Chesapeake Bay Land Use (acres)	Chesapeake Bay Basin Land Use
Forest	5,189,905	59%	26,512,720	65%
Cropland	1,405,191	16%	6,640,633	16%
Pasture	920,935	10%	2,438,478	6%
Urban	1,245,535	14%	4,853,216	12%
Other	99,827	1%	653,219	2%
Total	8,861,392	22% of bay watershed	41,098,267	100%

Data provided by Jeff Sweeney of the US EPA CBP; 2009 baseline scenario data

Change in Pathogens Due to the TMDL

The acreage of BMPs implemented due to the TMDL was derived by subtracting the baseline BMP implementation (on the ground in 2009) from the "with TMDL" scenario. Each BMP was associated with a particular land use and quantified in terms of the acres of that land use that were affected by implementation of the BMP. For example, prescribed grazing was associated with pasture and the percentage of total pasture under prescribed grazing was used to estimate changes in pathogen loads. These acreages were used to estimate change in loads based on baseline loads from the associated land use.

Because of data limitations, only a subset of BMPs that are capable of reducing pathogen loads were used in our analysis. BMPs were omitted from analysis if they were not measured in terms of acreage in the state WIPs or if efficiencies were specific to baseline conditions that could not be accurately measured. For example, BMPs measured as pounds of manure transported outside of the watershed and miles of stream restored were omitted (**Table A.2**, page 47). Also, some cropland practices in widespread use, such as continuous no-till, can be effective at reducing pathogens, but only when applied to cropland receiving manure; lack of sufficient data on manure handling prevented their inclusion. Omitting these practices, as well as point source practices, tends to make our study more conservative in terms of the TMDL effectiveness for reducing pathogens, since practices that are expected to be implemented as part of the WIPs were not counted, and some of these practices have been demonstrated to be highly effective at reducing pathogen loads (Table A.1).

To estimate the change in pathogen loads (measured as FIB) delivered to the main channel as a result of applying a subset of BMPs from the WIPs, we applied Equation 1:

$$\Delta FIB_{DS} = \sum_{l}\left(\sum_{b}\frac{(BMP\ Acres)_{b,l}}{(Total\ land\ area)_{l}}(\%FIB\ reduction)_{b}\right)(EOS\ load)_{l}(\%DS\ Delivery)_{l}$$

Equation 1

where

b is the BMP applied and

l is the land use type.

The equation shows that delivery of pathogens to the main channel depends on edge of stream (EOS) loads and downstream (DS) attenuation of pathogen loads. However, we report changes in EOS stream loads in addition to attenuated downstream loads, because EOS loads represent delivery to small channels, which can be relevant for projecting human health if people have contact with water in these small channels or adjacent receiving water bodies prior to substantial attenuation.

BMP Acres represents the acres of a given land use treated with a given BMP. The *Total land area* per land use (*l*) was derived from the baseline scenario. The *%FIB reduction* was the average removal efficiency for fecal coliform and *E. coli* for a given BMP. The proportion of treated acres to total acres in a given land use was multiplied by the percentage reduction for a given practice, and then these values were summed for all BMPs affecting a land use in order to generate a weighted sum representing the percentage reduction in pathogen loads expected for a given land use. The expected percent reduction for a given land use was multiplied by the baseline load for that land use to generate the edge-of-stream (*EOS*) load (cfu/yr). Finally, the downstream (*DS*) load was estimated by multiplying the edge of stream load by the average delivery ratio for all Potomac River segments modeled in the Vann et al. (2002) study, which was 21%.

Results of Pathogen Reduction Analysis

The analysis suggested that even with the limited set of BMPs that we were able to include, the pathogen reductions in the Potomac Basin due to the TMDL would be on the order of 23% of loads from pasture, 6% of loads from cropland, and 7% of urban loads (excluding point source loads) (**Table 5**). These load reductions sum to 19% of total estimated pathogen loads from all sources to the mainstem Potomac, including domestic and wild animal sources.

Percentage reductions are higher for the entire Chesapeake Bay Watershed. We estimate pathogen reductions on the order of 36% of loads from pasture, 8% of loads from cropland, and 17% of urban loads (excluding point source loads) (**Table 6**). These load reductions sum to 27% of total estimated pathogen loads from all sources to the tidal waters of the bay.

We expect these numbers to be underestimates of the mainstem effects because the analysis does not include effects of septic upgrades, CSO eliminations and some BMPs that are known to have high efficiency at removing pathogens. Also, the urban load reduction was lower for the Potomac compared to the bay watershed because some urban BMPs that were estimated to be on the ground in 2009 are not expected to be present in 2025 (i.e., negative acreages in Table 5). Urban load reduction results are sensitive to assumptions that practices will not be maintained.

The exclusion of BMPs, such as the waste management systems and septic connections, are a source of underestimation. For example, analyses developed for *The Bacteria TMDL Development for Three Tributaries to the Potomac River* (2011) estimated that the elimination of emissions from 46 failing septic systems in Sugarland Run would reduce *E. coli* loadings by 8.89 x 10^{11} cfu/yr – which is an estimated per unit loading of 1.93 x 10^{10} cfu/yr (VDEQ 2011). If, based on the literature review, we assume 1.93 x 10^{10} cfu/yr loadings per failing septic[3], and if the number of septic system connections identified in the TMDL were implemented, loadings could

[3] Several estimates of fecal coliform loadings per failing septic units were identified within the Chesapeake Bay Watershed during the literature review. The range per unit was 4.47 x10^9 to 6.39x10^{12} cfu/yr. The median range was selected for this estimate because it was based on HSPF modeling of instream loadings rather than per capita fecal coliform production rate (VDEQ 2003, 2011; WVDEP 2012).

be reduced by 4.22×10^{15}, which is 19% of the fecal coliform loadings from other urban non-point sources in the Potomac River Watershed but only 1% of total loadings from all natural and anthropogenic sources.

These reductions in loads are a substantial fraction of total loads to either the Potomac or bay watersheds. However, percentage reductions could be much higher in small water bodies. Because pathogen loads tend to become concentrated in localized areas, these reductions could be significant in terms of improving local water safety and preventing beach or shellfish closures, if practices were implemented at sufficient levels within small basins.

Table 5. Total Loading Reduction Estimates for the Potomac River Basin

BMPS	Land Use	BMP acres Phase II - Potomac	% Land use category covered by BMP	Average Fecal Indicator Bacteria reduction	Weighted sum (efficiency x % BMP cover)	Potential reduction at edge of stream (cfu/yr)	Potential reduction main channel (cfu/yr)	% Loadings Reduced
Pasture Practices								
Barnyard Runoff Control	Pasture	3,028	0.33%	81%	0.0027			
Loafing Lot Management	Pasture	55	0.01%	75%	0.0000			
Pasture Alternative Watering	Pasture	20,702	2.25%	90%	0.0202			
Prescribed Grazing	Pasture	165,042	17.92%	80%	0.1425			
Precision Intensive Rotational Grazing	Pasture	31,017	3.37%	90%	0.0303			
Horse Pasture Management	Pasture	15,074	1.64%	72%	0.0118			
Forest Buffers on Fenced Pasture Corridor	Pasture	3,845	0.42%	50%	0.0021			
Grass Buffers on Fenced Pasture Corridor	Pasture	6,741	0.73%	77%	0.0056			
Stream Access Control with Fencing	Pasture	27,919	3%	36%	0.0108			
Total Pasture Reduction (pasture + feedlots)		273,423	**30%**		0.2261	**8.07E+16**	**1.73E+16**	**23%**
Agriculture Practices								
Forest Buffers	Crop	41,934	2.98%	43%	0.0128			
Wetland Restoration	Crop	13,156	0.94%	35%	0.0033			
Land Retirement	Crop	39,312	2.80%	93%	0.0260			
Grass Buffers	Crop	41,700	2.97%	69%	0.0205			
Water Control Structures	Crop	238	0%	67%	0.0001			
Total Crop Reduction		136,341	**10%**		0.0627	**4.57E+15**	**9.80E+14**	**6%**
Urban/Suburban Practices								
Wet Ponds & Wetlands	Urban	-9,098	-0.7%	48%	-0.0035			
Dry Ponds	Urban	-78,767	-6.3%	80%	-0.0506			
Extended Dry Ponds	Urban	-6,324	-0.5%	80%	-0.0041			
Infiltration Practices	Urban	69,533	5.6%	93%	0.0519			
Filtering Practices	Urban	112,630	9.0%	75%	0.0678			

Table 5 (continued)

BMPS	Land Use	BMP acres Phase II - Potomac	% Land use category covered by BMP	Average Fecal Indicator Bacteria reduction	Weighted sum (efficiency x % BMP cover)	Potential reduction at edge of stream (cfu/yr)	Potential reduction main channel (cfu/yr)	% Loadings Reduced
BioRetention	Urban	15,321	1.2%	71%	0.0087			
BioSwale	Urban	6,685	0.5%	-6%	-0.0003			
Retrofit Stormwater Management	Urban	354	0.0%	57%	0.0002			
Erosion and Sediment Control	Urban	-29,738	-2.4%	57%	-0.0135			
Impervious Surface Reduction	Urban	21,904	1.8%	57%	0.0099			
Forest Buffers	Urban	12,177	1%	43%	0.0042			
Total Urban Reduction (urban + septic)		114,676	**9%**		0.0708	**1.60E+15**	**3.44E+14**	**7%**
Potomac River Basin Total (all sources)						**8.69E+16**	**1.86E+16**	**19%†**

†The percentage of total load reduction is calculated as the expected reduction in load from agriculture and urban non-point source sectors divided by estimated pathogen loads from all watershed sources (includes wildlife and point sources). Therefore the sum is smaller than the sum of the percentage reductions from the three individual source sectors shown in the table.

Table 6. Total Loading Reduction Estimates for the Chesapeake Bay Watershed

BMPS	Land Use	BMP acres Phase II - Chesapeake	% of land use category covered by BMP	Average Fecal Indicator Bacteria reduction	Weighted sum (efficiency x % BMP cover)	Potential reduction at edge of stream (cfu/yr)	Potential reduction main channel (cfu/yr)	% Loadings Reduced (of delivered to tidal)
Pasture Practices								
Barnyard Runoff Control	Pasture	12,055	0.49%	81%	0.0040			
Loafing Lot Management	Pasture	498	0.02%	75%	0.0002			
Pasture Alternative Watering	Pasture	83,693	3.43%	90%	0.0309			
Prescribed Grazing	Pasture	545,282	22.36%	80%	0.1778			
Precision Intensive Rotational Grazing	Pasture	277,657	11.39%	90%	0.1025			
Horse Pasture Management	Pasture	81,062	3.32%	72%	0.0239			
Forest Buffers on Fenced Pasture Corridor	Pasture	13,395	0.55%	50%	0.0027			
Grass Buffers on Fenced Pasture Corridor	Pasture	24,217	0.99%	77%	0.0076			
Stream Access Control with Fencing	Pasture	60,807	2%	36%	0.0089			
Total Pasture Reduction (pasture + feedlots)		1,098,666	**45%**	0	0.3585	**3.39E+17**	**7.28E+16**	**36%**
Agriculture Practices								
Forest Buffers	Crop	202,951	3.06%	43%	0.0131			
Wetland Restoration	Crop	86,978	1.31%	35%	0.0046			
Land Retirement	Crop	328,392	4.95%	93%	0.0460			
Grass Buffers	Crop	173,492	2.61%	69%	0.0180			
Water Control Structures	Crop	28,616	0%	67%	0.0029			
Total Crop Reduction		820,429	**12%**		0.0846	**2.91E+16**	**6.25E+15**	**8%**
Urban/Suburban Practices								
Wet Ponds & Wetlands	Urban	98,290	2.0%	48%	0.0097			
Dry Ponds	Urban	-452,870	-9.3%	80%	-0.0747			
Extended Dry Ponds	Urban	11,289	0.2%	80%	0.0019			
Infiltration Practices	Urban	545,939	11.2%	93%	0.1046			
Filtering Practices	Urban	740,706	15.3%	75%	0.1145			
BioRetention	Urban	47,980	1.0%	71%	0.0070			

Table 6. (continued)

BMPS	Land Use	BMP acres Phase II - Chesapeake	% of land use category covered by BMP	Average Fecal Indicator Bacteria reduction	Weighted sum (efficiency x % BMP cover)	Potential reduction at edge of stream (cfu/yr)	Potential reduction main channel (cfu/yr)	% Loadings Reduced (of delivered to tidal)
BioSwale	Urban	13,142	0.3%	-6%	-0.0001			
Retrofit Stormwater Management	Urban	24,513	0.5%	57%	0.0029			
Erosion and Sediment Control	Urban	-56,349	-1.2%	57%	-0.0066			
Impervious Surface Reduction	Urban	61,683	1.3%	57%	0.0072			
Forest Buffers	Urban	37,454	1%	43%	0.0033			
Total Urban Reduction (urban + septic)		1,071,777	**22%**		0.1697	**1.50E+16**	**3.21E+15**	**17%**
Chesapeake Basin Total (all sources)						**3.83E+17**	**8.22E+16**	**27%**†

† The percentage of total load reduction is calculated as the expected reduction in load from agriculture and urban non-point source sectors divided by estimated pathogen loads from all watershed sources (includes wildlife and point sources). Therefore the sum is smaller than the sum of the percentage reductions from the three individual source sectors shown in the table.

Potential magnitude of benefits

To evaluate the significance of these numbers, we considered their potential effect on human health. FIB are correlated with a number of illnesses caused by bacteria and viruses, and the illness that has been most consistently and clearly linked to water contact is increased risk of gastroenteritis (Kay et al. 1994; Fleisher et al. 1998; Wade et al. 2010), although other diseases have also been observed including respiratory illnesses, ear infections, and skin rashes, among others (Fleisher et al. 1998, 2010). Skin diseases (infections and rashes) have been most closely linked to non-point sources of pathogens (Fleisher et al. 2010) while gastroenteritis is more clearly linked to sewage (Wade et al. 2010). The gastrointestinal illnesses caused by shellfish consumption have been linked to concentrations of *Vibrio spp.* (Hlady & Klontz 1996), but *Vibrio* concentrations are widespread in the marine environment and are not highly correlated with fecal coliform (DePaola et al. 2000) and only weakly correlated with nitrogen concentrations (Pfeffer et al. 2003; Eiler et al. 2006; Johnson et al. 2010). However, concentrations of *Vibrio spp.* have been linked to increased sediment suspension in some cases (Vanoy et al. 1992; Pfeffer et al. 2003; Fries et al. 2008).

Whether or not reductions in pathogens reduces human illness from water contact or shellfish ingestion is a function of the probability of exposure to the pathogens, pathogen concentration, the number of people exposed, and the characteristics of the people that influence their susceptibility to disease (e.g., Soller et al. 2003). However, data for these characteristics are not generally available for the bay. If we draw from literature studies of other water bodies, we can see that dose-response relationships between pathogen concentrations in marine waters and cases of illness have been developed for several case studies. Cases are generally shown to have a roughly log-linear relationship (Cabelli et al. 1983; Kay et al. 1994; Wade et al. 2010), indicating that a relatively large decline in pathogens is needed to see a reduction in probability of disease.

An estimation of the reduction in cases of disease is beyond the scope of this effort, but we can reasonably assume that water bodies contain a range of pathogen concentrations that will be reduced in different proportions depending on the extent of BMP implementation in the watershed. As a type of sensitivity analysis, we can apply the 27% reduction estimated for the bay to water bodies with high and low concentrations of pathogens. Using the relationship estimated by Wade et al. (2010) (**Figure 1**), then a 27% reduction in pathogen concentration at either low or high concentrations of *Enterococcus* would translate to one fewer swimmer getting sick (e.g., a change from 1000 to 700 qPCR CCE/100ml[2] causes a decline in probability of illness from 0.13–0.12). Although GI illness is most closely associated with sewage or point sources of bacteria, stormwater can also be a source of pathogens that cause GI disease (Haile et al. 1999). We note that Wade et al. (2010) did not find a statistically significant response of skin infections and other diseases to concentrations of various bacteria species, although other studies have suggested a relationship (Haile et al. 1999; Dwight et al. 2004).

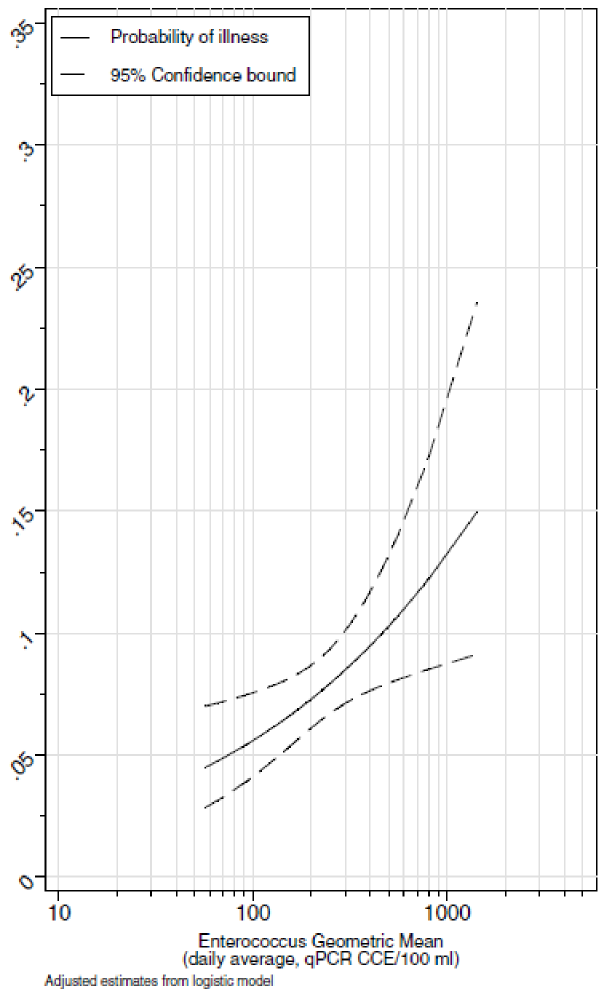

Figure 1. Probability of Illness as a Function of *Enterococcus* Concentration. From Wade et al. (2010)

Willingness to Pay

The increased water safety from reducing pathogens is likely to generate value to people in three primary ways. First, those who are in contact with the water (commercial fishermen, recreational anglers, boaters, and swimmers) are likely to have improved welfare due to illnesses avoided and may increase the number of trips they take. Second, more risk-averse recreators, who might currently be avoiding the water, might be induced to recreate in the bay, in response to improved water safety. Third, increased safety of shellfish could benefit commercial watermen, the burgeoning aquaculture industry, and seafood consumers. We would expect welfare increases from additional recreation trips, increased safety per trip, lowered costs of production for producers, and safer shellfish for consumers.

Table 7 summarizes government actions to prevent illness and reported cases of illnesses due to pathogens in Maryland and Virginia. For reference, we include a very rough estimate of the potential reduction in these adverse events due to the TMDL practices by assuming that the reduction in adverse events is equivalent to the 27% decrease in pathogens that we estimated for the bay. Although these values are modest, they have the potential to change public perception of the bay in ways that could substantially affect benefits, as further discussed at the end of this public health section.

Table 7. Summary of Adverse Events due to Pathogens and Potential Change with TMDL

	VA		MD		Annual reduction in MD and VA events w/TMDL[1]
	Baseline	w/ TMDL[1]	Baseline	w/ TMDL[1]	
Beach closures & advisories (avg annual days)	69 (out of 6900 days)	50	244 (out of 5900 days)	178	- 85
Shellfish bed closures (2013 sites)	176	128	77	56	- 68
Gastrointestinal illnesses (avg annual cases)	705	551[2]	491	383[2]	- 262
Other illnesses	n/a		n/a		
Impaired streams – TMDLs for pathogens (miles)	9,200		4,100		

[1] A rough estimate created by assuming a 27% decrease, which is the reduction value estimated for the entire bay watershed

[2] Values are estimated by first assuming that only 30% of cases are contracted from contact with open water (after Hlavsa et al. 2014) and that 27% of these cases are eliminated by the TMDL.

The potential value of beach closures and illnesses avoided has been evaluated in the economic literature. In a study that estimated willingness to pay to avoid illness, Machado and Mourato (2002) found beach goers in Lisbon, Portugal, were willing to pay on average $64.43 (2013 USD) per person to avoid gastroenteritis from contact with polluted water. The median willingness to pay (WTP) was substantially lower, $20.70 (2013 USD), per person, demonstrating a skewed distribution of values. Bin et al. (2005) summarized studies that evaluated the effects of reduced pathogens and we summarize his findings in the remainder of this paragraph, after converting his reported values to 2013 dollars. McConnell and Tseng (1999) evaluated the impacts of increased fecal coliform on use in 10 western shore beach sites of the Chesapeake Bay. For these generally wide sandy beaches within state or county parks, the study estimated the individual losses per trip of a two-fold increase in fecal coliform counts to be $2.51 per individual per trip for one site and $19.71 per individual per trip for all 10 sites. The study also valued lost availability of beach sites (due to closure) at $4.35 to $7.96 per individual per trip, depending on the site. Murray et al. (2001) estimated the seasonal value of removing one water quality advisory for Lake Erie single-day beach users in Ohio to be about $40.02 per person per year. Parsons et al. (1999) valued the estimated impact of beach closures on Delaware residents to be from $0 - $24.46 per individual per trip across six beach sites (three most and three least visited sites by Delaware residents) located in New Jersey, Delaware, and Maryland.

Conclusions for pathogens

Our literature review reveals that many BMPs being installed to reduce nutrients are effective at reducing pathogens. We provide a rough estimate of a 19% reduction in loads to tidal waters of the Potomac and a 27% reduction in loads to tidal waters of the Chesapeake Bay. Substantial new modeling and data collection would be required to improve this estimate and relate it to reduced cases of illness, beach closures, or shellfish bed closures. If we take the simple approach of assuming that adverse events decline at the same rate as pathogen concentrations in the bay (27%), we estimate this would translate into hundreds of fewer cases of reported illness and substantial welfare effects, given the potential number of beach users in the bay[4] and their willingness to pay to avoid illness. The health benefits of these reductions appear modest, based on overall reported numbers of illnesses, but could represent a significant change in safety of water contact, beach use or shellfish bed use in localized areas, if BMPs were concentrated in a watershed with high pathogen concentrations. Both beach closures and gastrointestinal illness are associated with low to moderate average willingness to pay per person, and when aggregated over the many beach users, the total WTP could be substantial.

Reduction in health risk from West Nile Virus

Current conditions and effects on ecosystem services

West Nile Virus (WNV) was first reported in 1999 in the bay watershed and the prevalence of the disease has been increasing (WVDNR 2003; CDC 2011a, 2011b, 2011c, 2012a, 2012c). In 2012 alone, over 80 cases of WNV and 8 WNV-related deaths were reported in Virginia, Maryland and DC (CDC 2012c). Although WNV has been reported in all age groups, people over 50 are most likely to have severe complications from the disease, including debilitating encephalitis and death (CDC 2012b). Because no vaccine exists for the virus, the only preventative methods available are reduction of bites from infected mosquitos and a reduction of infected mosquitos.

WNV is a zoonotic pathogen that is transferred from infected mosquitos, namely *Culex* (Cx.) *pipiens* and *C. tarsalis*, to avian and mammal hosts through mosquito bites (Winters et al. 2008). Evidence suggests that the incidence of WNV in humans is related to land use practices. In particular, and somewhat counterintuitively, increases in wetland area and the area of other types of vegetated areas suitable for bird habitat have been shown to *reduce* incidence of the disease. Both greater wetland area and bird

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[4] The number of people swimming in the Bay was not readily available but a survey estimates that 42% of US residents engage in swimming in lakes, ponds, oceans, or rivers in a given year (Cordell et al. 2005).

diversity have been observed to be negatively correlated with WNV transmission to humans (Ezenwa et al. 2007; Allan et al. 2008).

The observed reduction in disease incidence with increasing amounts of wetland or other bird habitat is thought to be driven largely by changes in the availability of hosts. The probability of WNV human infection appears to be driven both by the availability of blood meal species (such as different bird species) that are more preferred than human hosts and the probability that mosquitos will feed from competent reservoirs, meaning those species that are capable of maintaining and amplifying WNV. When preferred animal hosts are unavailable, mosquitoes carrying the virus may switch to feeding on humans, thus changing the rate of human disease incidence. In addition, bird species differ in their ability to spread the virus, so conditions that favor birds with less ability to spread the virus (low competence reservoirs) are thought to reduce transmission rates (Bradley et al. 2008; Allan et al. 2008).

In addition to the direct linkages between bird diversity and WNV, land use cover type has also been associated with changes in WNV transmission to humans. A study that modeled eight northeastern states suggested that individuals in counties with the lowest fraction of forested land use had a greater than fourfold risk of WNV infection than those individuals from counties with the highest quartile of forested land use (Brown et al. 2008). Ezenwa et al. (2007) observed that wetland cover was associated with a decline in mosquito infection rates, despite increased mosquito density. Similarly, Johnson et al. (2012) found that WNV-infected birds and mosquitos were significantly more prevalent in residential areas than large urban wetlands. The Ezenwa et al. (2007) and Johnson et al. (2012) studies also observed even lower rates in wetlands with diverse bird communities, further supporting the link between bird community composition and WNV.

Potential improvements from implementing the TMDL

The Phase II WIPs for the Chesapeake Bay add over 600,000 acres of BMPs which would increase the area of riparian buffers and wetlands, which are important habitats for birds and many other species. The habitat enhancements are likely to increase the number of bird species and change the abundance of certain key species linked to WNV transmission. Because bird diversity is positively correlated with increased forested land and wetlands, plans for tree planting and wetland creation may indirectly dilute WNV reservoirs by increasing species richness (Melles et al. 2003; Ezenwa et al. 2007). These changes, in turn, have the potential to reduce the incidence of WNV in human populations. However, further research would be needed to determine the ability of revegetation to increase bird diversity and to project the rate of WNV dilution associated with additional bird diversity.

Conclusions for West Nile Virus

Recent studies indicate that a higher proportion of wetland and forest cover (in urban and suburban settings) and the correlated increase in bird diversity lead to lower incidence of WNV, all else equal. Available information was insufficient to quantify the magnitude of change, but these studies suggest that the practices implemented to meet the Chesapeake Bay TMDL have the potential to decrease transmission of WNV to humans by increasing natural cover and associated bird diversity.

Harmful Algal Blooms (HABs)

Current conditions and effects on ecosystem services

Harmful algal blooms (HABs) are events in which algae become overabundant in a water body in response to a variety of biophysical factors (Paerl 1988; Smayda 1990; Anderson et al. 2002; Lopez et al. 2008). Blooms can be deemed harmful simply because of the high concentrations of algae, and/or because algae produce toxins. In the first situation, blooms may cause unappealing slicks and conditions such as floating mucilage, they may foul recreational fishing gear (Paerl 1988; Smayda 1990), and lead to hypoxic conditions (Anderson et al. 2002; Lopez et al. 2008).

For species that produce toxins, only small increases in the population size of toxin-producing algae may result in significant effects by killing or sickening masses of fish and other organisms that eat the fish, such as birds and people (Landsberg 2002; Anderson et al. 2002; Sellner et al. 2003; Lopez et al. 2008). Some HAB species produce toxins that are even harmful to inhale, creating risks of illness to recreational boaters and commercial fishers (Lopez et al. 2008). As a result of these risks, the presence of HABs prompts closure of fisheries and recreational boating areas, which can generate economic impacts to local businesses (Burkholder 1998; Anderson 2005; Ramsdell et al. 2005).

A global pattern of increased HABs that is associated with increased human population and eutrophication has been observed in all coastal areas (Anderson et al. 2002; Sellner et al. 2003). The Chesapeake Bay is no exception, and a seasonal increase in algae blooms is associated with spring riverine high flows that carry high nutrient loads (Anderson et al. 2002). Because the mechanisms of bloom formation are complex, they cannot always be related to nutrient enrichment. However, according to Kemp et al. (2005), "blooms of both the common bay dinoflagellate *Prorocentrum minimum* and the rarer dinoflagellate *Pfiesteria piscicida* appear to be stimulated by addition of dissolved organic nitrogen…" (Heil 2005; Glibert et al. 2012).

Recently a consensus has emerged among those who study HABs that "degraded water quality from increased nutrient pollution promotes the development and persistence of many HABs and is one of the reasons for their expansion in the U.S and other nations" (Heisler et al. 2008). Several HAB species (toxic and non-toxic) are found in bay waters (Marshall 1996; Glibert et al. 2012) and the most studied HAB species, *Prorocentrum*, occurs in the bay. Blooms of *Prorocentrum* have been associated with high concentrations of nitrogen (Glibert et al. 2012).

Potential improvements from implementing the TMDL

Observations from estuaries within and outside of the Chesapeake Bay suggest that HAB frequency and extent are related to nutrient loads and that negative effects are reversible (Anderson 2005; Heisler et al. 2008). The frequency and extent of blooms of the toxic cyanobacterium *Microcystis aeruginosa* declined sharply after wastewater treatment plants began removing phosphorus in the 1970s (Jaworski 1990; Sellner et al. 2003; Jones & Kraus 2010). In the Gulf of Mexico, a HAB species has exhibited an increasing number of blooms per year since the 1950s, which correlates with increased nutrient loading from the Mississippi River over the same time period (Dortch et al. 1997; Parsons et al. 2002). Similarly, there are examples from the Inland Sea of Japan and the Black Sea where reductions in nutrient loadings have corresponded to a reduction in the incidence of HABs (Anderson et al. 2002; Anderson 2005).

Given this evidence, the expected 25% reduction in nitrogen due to meeting the TMDL would be anticipated to reduce the frequency and/or extent of some types of HABs, based on the evidence that phytoplankton in general, and some HAB species in particular, have enhanced growth under elevated nitrogen levels. Evidence further suggests that production of toxins by some bloom species increases when nitrogen levels increase, suggesting that the deleterious effects of blooms, when they occur, may be reduced under conditions of lower nitrogen concentrations (Li et al. 2012).

Conclusions for HABs

Although data connecting HABs to nutrient concentrations is limited, a scientific consensus has emerged that eutrophication plays a substantial role in promoting their occurrence. In the bay, some toxin-producing species have been shown to decline in response to decreases in nutrients (Glibert et al. 2012), suggesting that the TMDL will reduce some particularly harmful HAB species. Any decline in blooms has the potential to generate benefits related to protection of human health, reductions in fish kills, reduction of hypoxia, and ultimately, to increased resilience of the ecosystem through the reduction of hypoxia-inducing bloom events. An added benefit of diminished incidence of HABs is reduction in spending on monitoring and detection, prevention strategies, and emergency response (Anderson et al. 2002; Hoagland et al. 2002; Sellner et al. 2003).

Difficult to value benefits of reducing health risks

Although the number of reported illnesses due to poor water quality in the bay (from pathogens and HABs) is relatively modest, and suggests that a reduction due to the TMDL would thus have only modest economic benefits, this analysis does not tell the whole story. The TMDL implementation has the potential to create something potentially more valuable – an improved feeling of safety regarding use of the bay for fishing, swimming and boating. Higher confidence in bay safety can potentially increase a variety of benefit types.

To understand why feeling safe matters, consider that the public response to an event that adversely affects health or safety can generate behavior changes that are disproportionate to the risks (Kasperson et al. 1988; Slovic 2000b) and these behavior changes can, in turn, generate disproportionate economic harms. This effect has been called the *social amplification of risk* (Kasperson et al. 1988) and it occurred in the bay area during the *Pfisteria* outbreak of 1997. In 1997, watermen reported widespread incidence of lesions and unnatural tissue growths in fish in the Lower Pocomoke River, on the Eastern Shore of Maryland. Watermen working that area also reported cold-like symptoms, skin problems, and overall poor health (Magnien 2001). Researchers later documented effects on memory and cognitive function in this group (Grattan et al. 2001). The proportion of fish affected was probably low, since sampling trawls found rates of problems of 0-6% (Magnien 2001). Eventually, the source of the problem was diagnosed as a toxic form of algae known as *Pfisteria piscicida* (Burkholder & Glasgow Jr 2001).

The public responded to news of this risk by avoiding seafood and cancelling fishing trips throughout the bay, even though the affects were limited to a few areas or a few seafood types (Magnien 2001). This had a substantial, though temporary, effect on restaurants, charter boat businesses, seafood wholesalers and others (Meyer 1997). Whitehead et al. (2003) used a survey to confirm the high risk aversion among the public to such threats and estimated that the lost benefits to seafood consumers in the Mid-Atlantic would be \$37–\$72 million in the month following a fish kill due to *Pfisteria.*

The study by Whitehead et al. (2003) further suggested that once risk becomes amplified, minor events can generate substantial economic impacts and be expensive to reverse. Their survey, conducted in the mid-Atlantic, showed that people rated seafood consumption as riskier when they were told of a major fish kill compared to those answering a survey that did not mention a fish kill, even though no relationship was made between the fish kill and the seafood. Interestingly, this higher risk perception persisted in the responses despite being told that the seafood was safe to eat. Only the use of (hypothetical) seafood inspections was sufficient to cause respondents to perceive the risk as comparable to the case without the fish kill. The same survey also found that people said that they would buy less seafood at any given price after any fish kill (major or minor).

While *Pfisteria* has not recurred, a more persistent amplification of risk occurs in the bay and has the potential to be ameliorated by the pathogen reductions that would occur as a result of the TMDL. Conversations with stakeholders and news articles indicate that some residents and visitors to the bay are hesitant to swim in open waters due to increased, although rare, occurrences of severe illnesses and deaths resulting from *Vibrio* and other bacteria in the bay (Wood 2009; Hale 2009; Dvorak 2010). Although only 30 cases were reported in Virginia in 2011 and 45 cases reported in Maryland in 2010 (including cases resulting from ingestion of raw filter-feeding shellfish), those interviewed are more cautious about swimming in the bay during warmer months or have decided not to swim in the bay at all (Wood 2009; Hale 2009; Dvorak 2010). These actions are inconsistent with state health department recommendations which suggest that more moderate approaches to reducing risk are adequate such as checking for beach closures, not swimming with open wounds, and avoiding the water for 48 hours after a heavy rainfall (VDH 2011b; AACDoH 2013).

With reduced pathogens as a result of the TMDL, we would not only expect fewer instances of serious health threats, the public could also have a dampened reaction to isolated events that are perceived as serious health threats, such as HABs, beach closures and shellfishing bans. The TMDL efforts, which will be enacted by many people throughout the watershed, could help to promote a "brand" of a clean bay–an effect that has been shown to help interrupt the social amplification of risk (Busby et al. 2009).

Ecosystem Service Benefit 2. Ecosystem Resilience and Contributions to Nonuse Values

Many people have concerns for protecting nature independent of human use (NRC 1999). People value the pure existence of natural assets, want to pass these assets along to future generations, and/or think that these assets ought to be protected for the benefit of others. Economists sometimes refer to these types of values as existence, bequest and altruistic values or, collectively, as *nonuse values* to distinguish them from the values derived from recreation, food supply, health protection or other uses.

When people are asked about their nonuse values for any natural system, including the Chesapeake Bay, they often mention concepts related to stewardship of a system and protection from disturbance (e.g., Manning et al. 1999). People also commonly recognize that some species (e.g., SAV) and water quality conditions (e.g., algae levels) serve as indicators of an ecosystem's overall condition or resilience in the face of disturbance (NRC 1999). Because people value stewardship, we can evaluate the potential magnitude of change in nonuse values by considering how the TMDL actions improve the resilience of the bay.

We define resilience as the ability of a system to self-regulate and buffer the effects of modest environmental disturbance and recover from severe disturbance. Such resilience may take on increased importance as the bay ecosystems are forced to respond to climate change and the associated increases in acidity, temperature, precipitation intensity, salinity variability, and other effects that can stress fish and other bay organisms (Najjar et al. 2010). If disturbance becomes more common, then increased resilience could translate into less time spent in a degraded state.

SAV and fish as indicators of resilience

Current conditions of SAV

The extent of seagrass and other submersed plants, collectively referred to as SAV, is a reflection of overall condition of the bay ecosystem and the magnitude of current and future benefits to people. SAV beds contribute to fishery productivity by serving as important feeding sites and refuges from predation for juvenile stages of fish and invertebrate species (e.g., striped bass, sea trout and blue crabs) in spring and summer (Perkins-Visser et al. 1996; Heck et al. 2003, 2008). During the fall, they serve as gathering grounds and as sources of food for diverse migrating or overwintering waterfowl. Further, SAV beds dampen the energy of waves and reduce storm surges, which prevents shoreline erosion and protects property during storms (Koch et al. 2009; Barbier et al. 2011, 2013).

Shallow regions of the bay were historically inhabited by extensive SAV beds that were rich with plant and animal life (Kemp et al. 1984; Borum et al. 2012). Starting in the 1960s, however, massive bay-wide declines in SAV abundance occurred (Kemp et al. 1983; Orth & Moore 1983), reducing plant cover to less than 30% of the potential plant habitat at mean water depths < 1m (Kemp et al. 2004). SAV growth is limited by algae and sediment in the water which prevents sunlight from reaching plants both by directly absorbing sunlight and indirectly by promoting growth of epiphytes (algae and other microscopic plants that grow on SAV leaves), which further shade plants (Kemp et al. 2005).

Because SAV has the ability to improve water quality, its loss exacerbates water quality problems by removing nutrient and sediment buffering capacity (Kemp et al. 2005). Water improvements occur when SAV facilitates the settling of particulates (Ward et al. 1984; Rybicki et al. 1997), prevents erosion and resuspension of materials (Madsen et al. 1993), and removes nutrients from the system (Caffrey & Kemp 1990, 1992; Tyler et al. 2003).

In the absence of SAV, water quality degradation may prevent the development of benthic algae (an important food source for some fish species) by preventing light from reaching sediments. In addition to loss of this prime fish food, SAV loss is associated with the loss of fish refuge, and both effects suggest a loss of prime habitat that would tend to increase stresses on the fish populations that use these habitats (Beck et al. 2001). Finally, the loss of SAV as quality habitat for waterfowl may have been a factor in declines in waterfowl populations and, at a minimum, has reduced diversity of waterfowl diets (Perry & Deller 1996).

Current conditions of fisheries

Fish are relatively abundant in the bay, but compared to even the recent past (1930-1970), the dominant species have shifted, the proportion of large fish has declined for some species, and certain species have shown dramatic declines (CBFEAP 2006). The oyster fishery has experienced near complete collapse, and in addition to major declines earlier this century, has declined 92% since 1980 (Wilberg et al. 2011). In addition to these changes, some fish and shellfish are showing signs of stress such as: reduced larval and juvenile production in menhaden; increased incidence of bacteria-induced lesions on striped bass; newly identified diseases in menhaden, softshell clams, and blue crabs; and, until recent management efforts were enacted, blue crab harvests were on the decline (Kemp et al. 2005; CBFEAP 2006; Houde 2011). The loss of oysters, in particular, is thought to have reduced removal of nutrients from the water column, a prominent ecological function of oysters (Newell et al. 2005; Sisson et al. 2011).

Why the TMDL may enhance resilience

For decades, ecologists and environmental managers have sought to improve understanding of the physical and biological mechanisms by which aquatic ecosystems buffer the effects of modest environmental disturbance and recover from severe disturbance (May 1972; Holling 1973). Some ecosystems are by nature adaptable and resilient, enabling them to recover quickly from external climatic and anthropogenic stresses; however, other ecosystems are prone to respond to stress by shifting, often abruptly, from one stable state to another (May 1977; Gunderson 2000). The shallow water environments that are common in the bay often exhibit complex, non-linear ecological dynamics that make systems unpredictable (Cloern 2001; Rose et al. 2009; Breitburg et al. 2009a). As a result, such systems defy simple cause-and-effect modeling of future condition because it can be difficult to judge which of the many interacting and counteracting forces will dominate (Costanza et al. 1993; O'Neill 1998; Rose et al. 2009; Breitburg et al. 2009a).

Despite these complexities, available evidence describes several mechanisms by which TMDL implementation may enhance the Chesapeake Bay ecosystem's ability to withstand stress and promote rapid recovery from disturbance. The mechanisms all fall under a common theme. Namely, that reducing nutrients and sediments alleviates multiple sources of stress to promote a greater diversity of species, more efficient functioning, and increased capacity within individual organisms and the system as a whole to respond to extreme events or novel stresses. Although nutrients are not the only stress on the bay, the literature suggests that alleviating even one stress can prevent multiple stressors from combining to create an impact more extreme than the sum of the individual effects acting independently.

The next section describes evidence that reduced nutrients and sediments in the bay may improve resilience through such mechanisms, and offers examples from severely degraded systems that may serve as cautionary tales of how failure to reduce controllable stresses can contribute to undesirable and sudden shifts in system condition. In describing the mechanisms by which eutrophication may influence resilience, we focus on two markers of bay condition, fish populations and SAV. We choose these endpoints because they serve as integrative indicators of many system processes, and because they are likely to affect people's future use and enjoyment of the bay.

Potential for SAV regrowth due to improved water quality

The evidence that the TMDL will promote improvements in SAV extent is relatively strong because SAV decline has been closely correlated with increased nutrient loads (Twilley et al. 1985; Kemp et al. 2005), and case studies have demonstrated that this process is reversible (Orth et al. 2010; Gurbisz & Kemp 2014). Recovery has been demonstrated in the Potomac River, where reductions in phosphorus loads due to wastewater treatment plant upgrades in the 1970s were associated with an almost complete elimination of blue-green algae blooms (an algal type associated with HABs) and a reappearance of SAV that appeared due to the water quality improvements (Carter & Rybicki 1986, 1990; Carter et al. 1994; Kemp et al. 2005).

Potential for fisheries improvements due to improved water quality

As a result of high natural variability and multiple, sometimes counterbalancing, drivers of change, it is difficult to project how the TMDL will influence fish populations. The evidence from estuaries and seas where nutrient inputs have been reduced from high levels to more moderate levels supports the idea that nutrient reductions reduce hypoxia, which will, in turn, improve habitat for many types of fish. However, it is not clear that this habitat improvement will translate into large changes in total fish abundance (Breitburg et al. 2009b). Models suggest that the improvements are likely to be most immediate for oysters and clams, which are most susceptible to hypoxia. However, for many mobile fish, the expected average annual effects of reduced hypoxia appear modest, based on existing models and understanding (Townsend 2012), but uncertainty of these results is high.

The lack of a dramatic response of fisheries to reductions in nutrient loads has many causes, which can be well described by a simple model relating fishery production to nutrient concentrations. Breitburg et al. (2009b) demonstrated that a concave function provided the best fit for data relating nitrogen loadings to total biomass of fish landings for a worldwide set of estuaries, supporting a theory first proposed by Caddy (1993). The model suggests that total fish biomass increases in response to nutrients over low to moderate levels of nitrogen loadings and then peaks before declining, as nitrogen continues to increase. The initial positive response by fish is due to the "fertilization" effect of nutrients on the algae and plant life (phytoplankton and epiphytes) that serves as the basis of the food web for fish that live throughout the water column (Nixon & Buckley 2002). The negative effects of nutrients on fish are expected to occur as a result of a combination of factors such as shifts in prey abundance and composition, hypoxia, harmful algal blooms, disease, and changes in habitat quality (e.g., loss of SAV) that reduce growth and increase mortality (Wu 2002; Kemp et al. 2005; Rose et al. 2009; Zhang et al. 2009; Breitburg et al. 2009a; Ludsin et al. 2009).

An implication of this model that is not well understood outside of fishery ecology, is that at moderate to high levels of nutrient inputs, the positive and negative effects of eutrophication on fisheries largely compensate for each other when viewed at the scale of the estuary (Breitburg et al. 2009a), resulting in high fish productivity under eutrophication. Multiple effects are at work to balance the system's response to positive and negative effects of eutrophication. A primary population stabilizer is that reduced productivity among bottom-feeding fish (e.g., flounder, croaker) is compensated by increased productivity of fish that live in the water column (e.g., menhaden) (de Leiva Moreno et al. 2000; Kemp et al. 2005). Also, some fish, such as striped bass, can adapt to conditions by shifting their feeding from bottom sources that are depleted to water column sources that are enhanced by eutrophication, such as the small fish that eat algae or zooplankton (Pruell et al. 2003). Further, even though some fish species within the bay may decline due to adverse conditions of eutrophication, others of that species may be thriving along the eastern seaboard, and these species can "subsidize" the bay every year, providing stability for the population (Ray 1997). However, this stabilizing effect is not as strong for species when a large fraction of the population uses the bay as primary habitat, such as striped bass, Atlantic croaker, eels, white perch, and American shad (Ray 1997).

Resilience effects from reducing the overall level of stress

Estimates from available models on fish production (particularly when based on commercial catch data)[5] are considered neither conclusive nor representative of non-average conditions because they are based on an incomplete understanding of relative influence of competing factors and feedbacks (Kemp et al. 2005; Rose et al. 2009). Fisheries have complex dynamics that suggest that they can decline rapidly if multiple stresses coincide, or conversely, can rebound from having one type of stress reduced.

A well-known case study that demonstrates how fisheries can respond to reduction in one stress, even when multiple stresses co-occur, is the decline and recovery of the striped bass in the bay. The commercial catch of striped bass peaked in 1973 and then declined 80–90% in the years before a moratorium was enacted in 1984 (Houde 2011). This reduction in stock was largely attributed to overharvesting, however, water quality was hypothesized to be a contributor to the decline because of the increased mortality of eggs and larvae that was attributed to low DO, low pH, trace metals, and temperature drops (Coutant & Benson 1990; Hall Jr et al. 1993; Richards & Rago 1999; Houde 2011).

The rebound of the striped bass fishery, which is both a commercial and recreational species, was largely attributed to continuous management of fishing pressure and inherent traits of the species. In particular, a store of old and large fish were available for spawning (Secor 2000) that produced large year classes in the 1980s, that ultimately translated into recreational catches increasing more than 400% coast wide between 1985 and 1989 (Richards & Rago 1999). Thus, the natural recuperative power of the species produced a recovery, when fishing pressure was reduced, that overcame any stresses due to systemic hypoxia and other sources of pollution.

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[5] It is understood that commercial catch data may be an inadequate indicator of fish stock since data are confounded by human adaptation to change in fish populations, such as increased effort, changes in location of effort, and shifts to formerly unexploited or less exploited species (Pauly et al. 1998; Caddy 2000; Kemp et al. 2005; Essington et al. 2006). Some evidence suggests that fish harvest may be more efficient under hypoxia since fish are known to aggregate near the edges of low oxygen zones which may increase fish catchability by commercial fishers (Craig 2012). If fish become easier to catch under hypoxia, it limits the ability to use landings to detect changes in stock abundance (Winters & Wheeler 1985).

While the striped bass fishery is currently doing well, a bacterial disease (mycobacteriosis) that produces lesions on fish and is thought to contribute to mortality is affecting a substantial percentage (10%) of young fish resident in the bay (Blankenship 2004; Kaattari et al. 2005; Houde 2011), and the overall infection rate may be as much as 50% (Overton et al. 2003; Gauthier et al. 2008). The causes of mycobacteriosis are not clear, but its rapid increase provides an example of how fisheries are frequently confronted with emergent stressors that can potentially reduce the vigor of the fishery, and when combined with other stressors, can substantially reduce survival and productivity. Thus, just as a past moratorium on fisheries catalyzed recovery, improvements in water quality might allow striped bass to better adapt and recover from new and novel stresses.

Other species have not demonstrated the same type of recovery as the striped bass and may be more likely to improve under the TMDL. Most notably, iconic bay species such as oysters, American shad, shortnose sturgeon, and Atlantic sturgeon, which were once abundant, currently remain at low percentages of their former stock levels (CBFEAP 2006; Wilberg et al. 2011). Although oysters have shown some increases in recent years, other species have sensitivities that require reducing multiple stressors. American shad larvae, for example, have been shown to have significantly reduced survival when subjected to very high suspended sediment concentrations (greater than 100 mg per liter) (Auld & Schubel 1978), suggesting that reductions in sediment loads would help these and similar species recover, particularly if frequency of very high sediment concentrations could be reduced.

Since sturgeon fishing is tightly regulated, their failure to recover is thought to be related to reduced suitability of spawning and nursery habitats (ASMFC 1998; Atlantic Sturgeon Status Review Team 2007) which appears to be due, in large part, to hypoxia (Collins et al. 2000; Niklitschek & Secor 2005) and loss of hard bottom that serves as spawning grounds (Secor et al. 2000a). Evidence that the TMDL may help in the recovery of these diminished species can be found in the case of the federally endangered shortnose sturgeon in the Hudson River. This ancient fish, beloved for its caviar, appears to have shown a marked increase in abundance in response to improved oxygen conditions in the river (Woodland et al. 2009). Further, because native juvenile sturgeon have been found in the Chesapeake Bay, it is not unreasonable to conclude that sturgeon could become more abundant, if oxygen levels were improved (Secor et al. 2000b).

Resilience due to having more oxygen

Water quality improvements are expected to enhance the vigor of fisheries by ameliorating the negative impacts on individuals. Hypoxia due to poor water quality is often cited as one of the most important pathways of harm to fish from eutrophication, and low oxygen levels are thought to directly harm fish by reducing growth, feeding rate, survival, and fecundity of individual fish (Rahel & Nutzman 1994; McNatt & Rice 2004; Shimps et al. 2005; Stierhoff et al. 2006; Thomas et al. 2006, 2007; Landry et al. 2007; Hanks & Secor 2011). Localized fish kills and crab jubilees[6] are often used as evidence of the harm that low oxygen can cause to fish. Recently, an additional pathway of harm has been identified. Namely,

[6] When crabs move to shallow water to escape hypoxic areas, they can create an abundance of easy-to-catch crabs, or a crab jubilee.

hypoxia appears to act as an endocrine disrupter on croaker with the potential to result in widespread failure of reproduction (Wu et al. 2003; Thomas et al. 2006, 2007; Landry et al. 2007).

In addition to direct stresses, hypoxia induces behavioral changes that create harm by displacing fish from their preferred habitat (Coutant & Benson 1990; Breitburg 2002; Wu 2002; Craig & Crowder 2005; Eby et al. 2005; Eggleston et al. 2005; Craig et al. 2005; Chan et al. 2008). This displacement is thought to "cost" the fish in terms of reduced growth or reproduction (Pyke 1984; Micheli 1997; Taylor & Eggleston 2000; Tyler & Targett 2007; Costantini et al. 2008; Rose et al. 2009) because fish are assumed to use the most favorable habitat for maximizing feeding, reproduction, and predator avoidance. For example, young summer flounder appear to use shallow waters to avoid large predators (Manderson et al. 2004), but shallow water hypoxia could force them into oxygenated deeper water[7] where predation risk increases. Hypoxia also can reduce the food supply for bottom-feeding fish by killing or reducing the size and diversity of shellfish, worms, and other creatures that serve as food sources (Llansó 1992; Wetzel et al. 2001; McAllen et al. 2009; Seitz et al. 2009).

Whether reducing hypoxia would have substantial effects on fisheries is unclear (Diaz & Solow 1999; Breitburg et al. 2009b). Some evidence from lakes and semi-enclosed seas, such as the Great Lakes, Baltic Sea, Black Sea, Sea of Azov, and Mediterranean Sea suggests that hypoxia, in concert with fishing pressure and other effects, has played a role in reducing the abundance of commercially exploitable fish and enhancing the dominance of invasive species, including jellyfish, (Caddy 1993, 2000; Diaz 2001; Breitburg 2002; Daskalov 2002, 2003; Oguz 2005). An estuarine example is from the Gulf of Mexico, where catch per unit effort of brown shrimp was negatively correlated with the extent of hypoxia (Zimmerman & Nance 2001; O'Connor & Whitall 2007). However, the same effect on abundance was not found to be true for the white shrimp (Zimmerman & Nance 2001), suggesting that this effect is not generalizable across species.

Resilience to climate change or new stressors

SAV

As has been seen with fish, reduction of a single source of stress is expected to help SAV be resilient to other stresses. Although SAV are expected to expand their range in response to improved water quality, the future of two important SAV species remains uncertain due to their intolerance to the combined stresses of low water clarity and expected climate change effects. Temperature increases stress on eelgrass (*Zostera marina*) by increasing their light requirements (Wetzel & Penhale 1983; Moore et al. 1997). When temperatures rise, respiration rates increase, thus requiring more light to maintain plant condition (Evans et al. 1986). Thus, the combined stress of high summertime temperatures that are expected with climate change (Preston 2004; Najjar et al. 2010) and low light as a result of eutrophic water can lead to complete bed loss, particularly if combined stresses occur in consecutive years because seed banks have become depleted (Jarvis & Moore 2010, 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 create stress on a second SAV species, wild celery or *Valisneria americana*. French

[7] Although much of the hypoxia occurs in deep water, shallow water can lose oxygen due to daily cycling of oxygen by photosynthesis, while deeper water remains oxygenated.

and Moore (2003) found that light requirements may be 50% higher when this plant is growing in higher salinity, suggested that periods of both high salinity and low water clarity will be difficult for this species to tolerate.

What remains unknown is whether bay SAV species may transition to heat or salt-tolerant species as conditions change. Evidence for such a transition has not been seen so far in the bay and a recent review of aquatic plant responses to climate change concluded that "the rate of climate change appears more rapid than aquatic plant dispersal" (Bornette & Puijalon 2011). If new species do not establish and if plants do not adapt, then, without the TMDL to improve the light levels reaching SAV, the added stress of climate change may be lethal to multiple seagrasses.

Fish

Despite the relatively optimistic picture of *overall* fishery productivity under eutrophication, shifts within or among species cause functional changes in the ecosystem that could reduce its resilience to stress (Steele 1991), particularly when multiple stressors combine (Caddy 1993, 2000; de Leiva Moreno et al. 2000). Evidence that functional changes that affect fisheries are occurring in the bay comes from several sources. For one, if menhaden are excluded from fishery landings, total fisheries landings and biodiversity have declined since 1980 (Kimmel et al. 2012). Declines in species diversity are a concern because they represent a loss of genetic and behavioral variability that may diminish a system's ability to maintain productivity year-to-year under different types and levels of stress (Loreau et al. 2001; Hooper et al. 2005; Hector & Bagchi 2007). In particular, species have different preferences for salinity and temperature, so maintaining a variety of species allows the system to be productive when these physical conditions fluctuate (Winemiller & Rose 1992), as might occur with climate change.

Reducing stress to avoid tipping points

Another line of thinking regarding how nutrient reductions may influence the stability of fishery production, SAV beds, or other aspects of the ecosystem, is the potential for promoting or preventing sudden shifts between alternative stable states. In other words, rather than systems moving gradually along either a degradation or recovery trajectory, systems may make sudden jumps between states when multiple stressors combine in novel ways (Scheffer & Carpenter 2003). Paine et al. (1998) described this effect by saying, "…disturbances leave a residual assemblage that provides a legacy on which subsequent patterns build." While nutrients are not expected to increase dramatically without the implementation of the TMDL, the current level of stress from water quality may leave the system vulnerable to shifts initiated by such events as major storms or changing climate.

A recent story of seagrass recovery provides a dramatic example of how water quality improvements may have promoted a shift to a more desirable stable state after a system reached a tipping point. The Susquehanna flats seagrass bed was a vibrant lush ecosystem that served as valuable habitat for fish and waterfowl and offered popular sites for anglers and hunters, until its peak abundance in the 1960s. After this peak, SAV cover gradually declined, coinciding with deteriorating water quality conditions. Finally, Tropical Storm Agnes in 1972 delivered the final straw. After Agnes, SAV abundance declined by > 70% in one year and the bed remained sparse for 30 years (Kemp et al. 1983).

Remarkably, this bed has recently recovered to a great extent and has been able to withstand two extreme weather events (Tropical Storms Irene and Lee in 2011) with only a temporary decline in grasses. Research by Gurbisz and Kemp (2014) suggest that the recovery was initiated by a combination of a drought that reduced nutrient loads sufficiently for beds to re-establish and a decline in nutrient loads that helped to maintain the bed. A probable mechanism for this recovery was the positive feedbacks that occurred when small improvements in water clarity promoted sparse seagrass growth. That initial growth slowed down overlying water and caused settling of suspended particles. This improved water clarity promotes more seagrass growth, further decreasing suspended particles, increasing water clarity, and so on (Gruber & Kemp 2010; Gruber et al. 2011). Thus, in the presence of sufficient water quality, positive feedback loops reinforced and accelerated small improvements in environmental conditions and promoted a more resilient bed than existed in the recent past.

Evidence that declining water quality may contribute to detrimental shifts in state, comes from ecological case studies that show the cascading effects of accumulated stresses. Scheffer and Carpenter (2003) document a recent example of how excess nutrients appear to have contributed to a dramatic shift in Caribbean coral reefs from diverse reef ecosystems with abundant fish to one in which the corals became algae-encrusted with less diverse and abundant fish communities (Hughes 1994). According to Scheffer and Carpenter (2003): "Only with hindsight were the probable mechanisms unraveled. Increased nutrient loading as a result of changed land use had promoted algal growth, but this result did not show as long as herbivorous fish suppressed the algae. With time, intensive fishing reduced the numbers of fish, but, in response, the sea urchin *Diadema antilliarum* became abundant and took the role of key herbivore. Finally, when a pathogen hit the dense *D. antilliarum* populations, algae were released from grazer control and the reefs became overgrown rapidly."

Other case study examples that are specific to estuaries or semi-enclosed seas support the idea that eutrophication sets the stage for shifting to undesirable states. In San Francisco Bay, the combined effect of a sparse benthic community (due to hypoxia) and two years of climate extremes promoted establishment of the non-native invasive corbiculid clam that appears to have contributed to the decline of several commercial fish species and the endangered delta smelt (Paine et al. 1998). Similarly, Kemp et al. (2005) describe the deterioration of fisheries in the Black Sea (Daskalov et al. 2007; Oguz & Gilbert 2007), from which they draw the conclusion that, "detrimental effects of eutrophication may not be fully manifested until a combination of excessive fishing activity, unusual climate regimes, introductions of alien species, and nutrient loading overwhelm the ecosystem's resilience."

Potential beneficiaries

It is difficult to estimate who values the bay in ways that do not involve direct or indirect use. After all, we cannot observe people valuing the bay without using it in some fashion. However, survey results consistently show that nonuse values are prevalent among many types of people, and that people who use systems like the bay for fishing and other types of recreation typically also hold nonuse values (Johnston et al. 2013). In focus groups that we have conducted with bay stakeholders, we often hear people express concerns about their children or grandchildren being able to enjoy the bay. These concerns are clear examples of nonuse values.

Conclusions for resilience reflected in SAV and fish

The complexities of estuarine responses to nutrient loading suggest that the benefits of implementing the TMDL will not be a simple gradual improvement in all conditions as nutrients and sediments decline (Boynton et al. 1983; Cloern 2001). Rather, the benefits of having lessened the stress of eutrophication may not fully manifest until the system needs to withstand or recover from new or unusual levels of stress. The literature strongly suggests that the ability to adapt to higher water temperatures, salinity variability, storm damage, or the introduction of an invasive species may be enhanced once the system is released from high eutrophication stress.

In addition to these long-term and uncertain benefits, some benefits will be readily apparent as a direct result of implementing the TMDL. Evidence is relatively strong that SAV beds will increase in extent and thus provide benefits associated with promoting fish productivity, waterfowl habitat and shore stabilization. Further, evidence suggests that improvements in water quality and associated restoration of SAV beds and shellfish would contribute to an increased ability to buffer nutrient loads by promoting settling and removal of nutrients and sediments from the system and by reducing the amount of algae that depletes oxygen during decomposition.

History and understanding of fish physiology suggest that improvements in water quality might allow fish to better adapt and recover from new and novel stresses. Yet, as a result of high natural variability and multiple, sometimes counterbalancing, drivers of change, it is difficult to project how the TMDL will influence fish populations in the absence of novel stressors. Ignoring the potential for tipping points or feedbacks in the system for the moment, the literature and available models suggest that we should expect modest increases in certain sensitive fish species, such as sturgeon, and possibly a slight shift in the relative abundance of bottom-feeding fish, such as flounder, relative to fish that feed in the water column, such as menhaden (de Leiva Moreno et al. 2000; Kemp et al. 2005). However, the tendency of estuaries to shift between alternative stable states suggests that the TMDL may play a more important role in averting system shifts to less desirable states, which could include states that have lower fish productivity than present. The net effect of the TMDL may be to promote the feedbacks that allow species to resist stress or that reinforce recovery mechanisms following disturbance.

Toxins

Current conditions and effects on ecosystem services

Toxins reach the environment through a variety of pathways including atmospheric deposition, point sources, and non-point sources. A recent EPA review of toxic contamination in the Chesapeake Bay Watershed (USEPA et al. 2012) considered a number of groups of contaminants, only some of which are likely to be affected by the TMDL practices because they are washed into the bay during rainfall and runoff events.

A number of toxic contaminants can be associated with non-point sources like agricultural lands and stormwater runoff, especially from impervious surfaces such as roads, parking lots, and driveways (Hwang & Foster 2006). The following subsections define each of these groups of toxins and, where possible, their impacts on ecosystem services.

Polycyclic Aromatic Hydrocarbons (PAHs)

PAHs are a family of compounds derived from either petroleum and coal or the combustion of fossil fuels and wood products. PAHs found in sediment can have a deleterious effect on benthic organisms which can lead to decreased growth, survival and reproductive success of the fish that consume them (USEPA et al. 2012). Chronic exposure to PAHs by fish embryos can result in death, deformities, or decreased growth (USEPA et al. 2012). In general, fish tend to metabolize, rather than accumulate PAHs.

Petroleum hydrocarbons

Petroleum hydrocarbons include a mixture of chemicals originating from crude oil. Petroleum can enter the environment through point (e.g., oil spills) or non-point (e.g., stormwater runoff) sources. In the Chesapeake Bay, petroleum contamination is localized to areas with high levels of shipping activity (EPA et al. 2012) (USEPA et al. 2012). A water standard for petroleum hydrocarbons has not been established, but a numerical standard for "oils and grease" (including non-petroleum based oils) exists. A narrative standard (i.e., no visible sheen) is typically used to evaluate compliance (USEPA et al. 2012). Data for petroleum residue in fish tissue are not available, and petroleum-related risk to wildlife is primarily associated with oil spills rather than stormwater runoff (USEPA et al. 2012).

Pesticides

Pesticides are products designed to prevent, destroy, repel or reduce pests. They have a wide range of chemical characteristics, targets and application procedures. Most pesticide use is associated with agriculture and residential land, as well as golf courses, mosquito and gypsy moth control, invasive species control, etc. (USEPA et al. 2012). Due to the wide variety of applications, many point and non-point pathways to the Chesapeake Bay Watershed exist.

The USGS National Water Quality Assessment monitors pesticide occurrence in the nation's rivers and streams. Among the major findings of this assessment are that, nationwide, the occurrence of pesticides in many streams may affect aquatic life or fish-eating wildlife, but pesticides are seldom present in concentrations that exceed human health benchmarks; however, these benchmarks are constantly being updated (USEPA et al. 2012). Among the potential environmental effects of the presence of various pesticides on fish and wildlife are the suppression of the immune system in certain fish species, correlations between atrazine concentrations in the water column near smallmouth bass nesting sites and intersex conditions in fish found at those sites, and immune system suppression in tadpoles following exposure to certain pesticides (USEPA et al. 2012). Due to the variety of pesticides in use, the regular introduction of new pesticides, and the persistence in the environment of pesticides no longer in use (e.g., DDT), the extent and severity of pesticides in the Chesapeake Bay watershed is uncertain.

Pharmaceuticals

Pharmaceuticals are chemicals used in the diagnosis, mitigation, treatment, cure and prevention of disease. They are widely used in human, veterinary, and livestock applications. The environmental effects of pharmaceuticals on aquatic ecosystems are not well understood, and at present, no aquatic-life or related water-quality or sediment-related benchmarks exist for pharmaceuticals (USEPA et al. 2012).

Two large classes of pharmaceuticals found in the environment, synthetic hormones and anti-depressants, are associated with point sources. A third class of commonly used pharmaceuticals, antibiotics, have the potential to enter the environment from point or not-point sources. The effect of antibiotics released to the environment on microbial communities is not well understood, although research is ongoing. Data on the occurrence of pharmaceuticals in the environment in the Chesapeake Bay Watershed is limited, so the extent and severity is also uncertain, but sources of these chemicals in the watershed are widespread.

Household and personal care products

Household and personal care products are a loosely defined group of products including cosmetics, detergents, soaps, and food additives that include a mixture of organic and inorganic ingredients. These products are most frequently introduced to the environment through point sources (e.g., landfills, wastewater treatment plants), but there is also potential for non-point introduction as well (e.g., through careless handling of trash).

Antimicrobials such as triclosan and triclocarban have been shown to have sublethal effects at relevant environmental concentrations on a number of species (USEPA et al. 2012). However, the severity of contamination from household and personal care products in the Chesapeake Bay watershed is poorly understood due to gaps in understanding the range of potential adverse ecological consequences of the presence of these products and their degradates at environmentally relevant concentrations (USEPA et al. 2012).

Biogenic hormones

Biogenic hormones are naturally occurring hormones created and excreted by humans and other organisms. Biogenic hormones primarily enter the environment through point sources (e.g., wastewater treatment plants), but their presence in animal waste means a potential non-point source as well. Manure management practices will determine the magnitude of this pathway. Biogenic hormones have the potential for endocrine disruption in fish (USEPA et al. 2012).

Metals and metalloids

Metals are naturally occurring constituents of rock and sediment that may also be delivered to the environment via anthropogenic sources. Mercury is one of the most prevalent sources of waterbody impairment in the Chesapeake Bay watershed; however, it enters the environment primarily through atmospheric deposition, and is therefore unlikely to be affected by the WIPs. Other metals that may enter the environment through point or non-point sources have more localized contamination in the Chesapeake Bay watershed and include:

- cadmium from tire fillers, tire wear, and lubricants (WSDOT 2007)
- chromium from moving engine parts, metal plating and brake linings (WSDOT 2007)
- copper from bearing and bushing wear, moving engine parts, brake linings, radiator repair, and copper roofs (BCMoE n.d.; UWEX 1997; WSDOT 2007)
- iron from automobiles, moving engine parts, and urban infrastructure such as bridges and guardrails (BCMoE n.d.; WSDOT 2007)

- lead from tire fillers, bearing wear, and automotive and radiator repair(BCMoE n.d.; WSDOT 2007)
- manganese from moving engine parts (BCMoE n.d.; WSDOT 2007)
- nickel from diesel fuel, lubricating oil, metal plating, brake linings and asphalt paving (BCMoE n.d.; WSDOT 2007)
- zinc from galvanized metal roofs, gutters and downspouts (UWEX 1997)

For many of these groups of toxins, their effects on water quality, sediments, fish and wildlife are not completely understood, and therefore there are no water quality benchmarks defined (e.g., pharmaceuticals, household and personal care products, biogenic hormones). However, some toxic contaminants have known effects on birds, fish, amphibians and other organisms, depending on the contaminant and concentration at which fauna are exposed. For example, lead can adversely impact reproduction, growth and development of fish at levels as low as 7 micrograms per liter (UNEP 2010). Similarly, PAHs may cause deformities and abnormalities and adversely impact reproduction, growth and development of fish at sediment levels as low as 22.8 mg/kg total PAHs (USGS 2011; EPA et al. 2012). Further, elevated levels of the pesticide atrazine in the Potomac River have been associated with reproductive development abnormalities in male fish in the Potomac River (EPA et al. 2012).

Toxic contaminants can also have developmental, neurological and carcinogenic effects on humans depending on the contamination and exposure level. Children with blood lead levels as low as 10 micrograms per decaliter can experience declines in IQ (UNEP 2010).

Potential improvements from implementing the WIPs

The TMDL has the potential to reduce toxics by implementing BMPs that reduce loadings of toxic-laden sediments, reduce impervious surfaces, and decrease runoff volume and flow. Many metals, PAHs and other organic contaminants are significantly correlated with suspended solids (Hwang & Foster 2006; Schiff & Tiefenthaler 2011; Gunawardana et al. 2012). By reducing the sediment loading to the bay and its waterways, many toxics are likely to be diverted from streams as well. Furthermore, many toxics are associated with impervious surfaces because these pollutants accumulate and are easily washed off of impervious surfaces (e.g., petroleum) (Hwang & Foster 2006).

Common agricultural practices such as the use of pesticides and spreading manure on fields can introduce toxins found in animal waste (e.g., pharmaceuticals, biogenic hormones) to the environment. A USEPA report (US EPA 2000) found that the amount of manure applied per acre of farmland in states in the Chesapeake Bay watershed was among the highest in the nation. The implementation of BMPs that reduce runoff from agricultural lands (e.g., barnyard runoff control, forest buffers, etc.) will likely result in reduced introduction of these types of toxins in the Chesapeake Bay.

The state of Maryland (in cooperation with the District of Columbia) has established trash TMDLs for the Anacostia and Patapsco rivers. In these urban areas, the interception of trash will reduce contamination associated with household products (USEPA et al. 2012).

Potential magnitude of impacts

The potential impact of the TMDL on toxins is difficult to quantify, but will be significantly tied to sediment prevented from entering the waterways, as well as the runoff volume and velocity which determines flushing capability. For example, in the Anacostia, 68–97% of PAHs have been observed to be associated with sediment particles in stormwater runoff, and approximately 75% of PAHs entering the Chesapeake Bay annually are estimated to be associated with particles (Ko & Baker 2004; Hwang & Foster 2006). The study also noted that areas with higher percentages of roadways and impervious surfaces generated higher PAH concentrations in runoff.

The size of solids captured by the BMPs will also affect the magnitude of effect. One California study found that metals were most often correlated with particles between 100 and 250 micrometers, whereas PAHs were most often associated with particles smaller than 100 micrometers (Lau & Stenstrom 2005). Since micrometer size particles are easily entrained in moving surface water, we can assume that most reductions in surface runoff over impervious surfaces will be associated with a large proportional reduction in these toxins. However, we have not estimated the absolute size of this effect.

In agricultural and suburban settings, vegetative buffers may be capable of reducing pesticides and antibiotics from entering waterways by greater than 58 and 75%, respectively (USDA-NRCS 2000; Lin et al. 2011; Everich et al. 2011). The TMDL may also assist in the overall reduction of certain pesticides by promoting degradation in areas of permanent vegetative cover (USDA-NRCS 2000). The degradation of pesticides have been observed to take place at a faster rate in soils compared to surface water and sediments (US EPA 2003).

Conclusions for toxins

Toxic contaminants found in nonpoint source runoff can have varying deleterious effects on humans, birds, fish, amphibians and other organisms, depending on the contaminant and concentration at which fauna are exposed. The Chesapeake Bay nutrient and sediment TMDL has the potential to reduce toxics by implementing BMPs that reduce loadings of toxic-laden sediments by reducing impervious surfaces, runoff volume and flow, and trash that would otherwise carry toxins to the bay.

Summary

This report describes some of the benefits that might result from implementing the TMDL, but that cannot be valued in monetary terms (summarized in **Table 8**). These potential benefits are a direct result of decreasing nutrient and sediment loads and are by-products of implementing management practices and projects to achieve the TMDL. The first section of the report discusses the potential for reduced risks to human health that might occur from substantial reductions in pathogens (at least 19–27%), reduced risk of West Nile virus transmission to people, and reduced incidence of HABs. Although the incidence of illnesses from these causes is low, people have been shown to dramatically change their behavior in response to low health risks. Therefore, small reductions in illnesses may generate disproportionate increases in welfare by increasing recreational opportunities and enhancing feelings of safety and well-being. Further, reduced incidence of health threats would be expected to prevent economic impacts to local businesses that can result from the social amplification of these risks.

The second section of the report discusses the potential for reduced nutrient, sediment, and toxic loads to enhance resilience of bay ecosystems to future changes. Increased resilience of the bay enhances the chances that the bay will continue to support fisheries far into the future and could lessen the time that the bay spends in a degraded state following major disturbance. Reduced nutrient and sediment loads are expected to increase distribution of SAV, increase abundance of selected fish species, and reduce hypoxia, all of which are thought to promote the ability of the system to tolerate and adapt to novel threats. The benefits of increased resilience may not be clear until the bay experiences more intense climate change stressors, or unless stressors combine in undesirable ways. However, recent improvements in the bay and case studies of other degraded systems suggest that improved water quality helps to create a system that recovers more readily from disturbance and avoids tipping points that could shift the system to an undesirable state.

Table 8. Summary of TMDL Effects on Ecosystem Service Benefit Indicators

Ecological Indicator	Human Welfare Effects	Expected Direction of Change in Welfare	Level of Certainty of Benefit Change
In-water pathogens	Reduced risks to human health (gastroenteritis, infections, etc.)	+ (>19-27% reduction in loads)	Moderate
Bird diversity & land cover	Reduced risks to human health (West Nile Virus)	+	Low
HAB incidences & toxicity	Reduced risks to human health (toxin-induced illnesses); food supply; recreation; local business support; nonuse benefits from improved bay resilience[1]	+	Low
Oysters & fish diversity	Increases in food supply; recreation; business support; nonuse benefits from improved bay resilience and support of local heritage	+	Moderate
Abundance of dominant fish (short-term)	Food supply; recreation; business support; nonuse benefits from improved bay resilience	- or neutral	Moderate
SAV	Property protection; nonuse benefits from improved bay resilience	+	High
Toxics (from non-point sources)	Reduced risks to human health; nonuse benefits from improved bay resilience	+	Low
Probability of system shift to undesirable state (due to novel stressors)	Nonuse benefits from improved bay resilience	+	Low

[1] *Nonuse benefits from improved bay resilience* reflects the values that many people express for stewardship of the natural environment and preserving resources for future generations.

Appendix A. Supplemental Information for Pathogen Analysis

Table A.1. Pathogen Reduction Efficiencies for Crop, Pasture, Urban and Septic BMPs

Best Management Practice*	Loading Reduction Efficiency (%)	Avg Fecal Coliform and E. Coli (FIB)† Efficiency (%)	Reference
	Crop Practices		
Forest Buffers	Fecal coliform: 43 - 57%	50%	VDEQ 2003
Grass Buffers	*E.coli*: 58-99% Total coliform 67-99% Fecal coliform: 28-100% Fecal streptococci: 70-84% *Cryptosporidium*: 93-99% *Giardia*: 26%	71%	MPCA 2009; Peterson et al. 2012b
Land Retirement	90-93%	92%	VDEQ 2003; Peterson et al. 2012b
Water Control Structures	Detention structures: 67%	67%	Leisenring et al. 2012
Wetland Restoration	*E. coli*: 40% Fecal coliform:30%	35%	VDEQ 2003
Non-urban Stream Reduction	No estimate	Not included in reduction estimate	
	Pasture Practices		
Barnyard Runoff Control	Fecal coliform: 81%	81%	(USGS 1998)
Forest Buffers	Fecal coliform: 43 - 57%	50%	VDEQ 2003
Grass Buffers	*E.coli*: 58-99% Total coliform 67-99% Fecal coliform: 28-100% Fecal streptococci: 70-84% *Cryptosporidium*: 93-99% *Giardia*: 26%	71%	MPCA 2009; Peterson et al. 2012b
Horse Pasture Management	*E. coli*: 72%	72%	Peterson et al. 2012a

* No comprehensive list defining the BMPs used in the WIPs was identified, however definitions for these agricultural practices agricultural practices can be found at: http://mda.maryland.gov/resource_conservation/WIPCountyDocs/bmpdef_pg.pdf. Summaries of the types of practices used in the urban BMPs can be found here: http://www.dnrec.delaware.gov/swc/wa/Documents/ChesapeakePhaseIIWIP/Final_Phase2_CBWIP_03302012A.pdf.

† FIB, or Fecal Indicator Bacteria; Reduction Efficiency is represented by the average reduction efficiencies of *E. coli* and fecal coliform for the purposes of this analysis.

Table A.1. (continued)

Best Management Practice*	Loading Reduction Efficiency (%)	Avg Fecal Coliform and *E. Coli* (FIB)† Efficiency (%)	Reference
Loafing Lot Management	Fecal coliform: 50%	50%	VDEQ 2003
Pasture Alternative Watering	*E. coli*: 85-95% Fecal coliform: 51-94% Fecal streptococci: 77%	82%	Sheffield et al. 1997; Byers et al. 2005
Precision Intensive Rotational Grazing	Fecal coliform: 90%	90%	MPCA 2009
Prescribed Grazing	*E. coli*: 66- 72% Fecal coliform: 90 - 96%	80%	Peterson et al. 2011a, 2011b
Stream Access Control with Fencing	*E. coli*: 37-46% Fecal coliform: 30-94%	52%	Schaetzle 2005; Peterson et al. 2011b
Ammonia Emission Reductions	No estimate	Not included in reduction estimate	
Conservation Tillage w/ Continuous No Till	No estimate: Heavily dependent on if and when animal manure has been applied.	Not included in reduction estimate	Ramirez et al. 2009
Dairy Precision Feeding	No estimate	Not included in reduction estimate	
Livestock Mortality Composting	No estimate	Not included in reduction estimate	
Livestock Waste Management Systems	*E. coli*: 97-99% Fecal coliform: 44- 99% Fecal streptococci: 46 -99% Total coliform: 99%	Not included in reduction estimate	VDEQ 2003; Redmon et al. 2012;
Manure Transport Inside CBWS	No estimate	Not included in reduction estimate	
Manure Transport Outside CBWS	Assumed to be 99%	Not included in reduction estimate	
Non-urban Stream Restoration	Fecal Coliform: 30%	Not included in reduction estimate	VDEQ 2003
Poultry Phytase	No estimate	Not included in reduction estimate	
Poultry Waste Management Systems	Fecal coliform: 75% *E. coli*: 96%	Not included in reduction estimate	VDEQ 2003; Redmon et al. 2012

Table A.1. (continued)

Best Management Practice*	Loading Reduction Efficiency (%)	Avg Fecal Coliform and *E. Coli* (FIB)† Efficiency (%)	Reference
	Urban Practices		
BioRetention	*E. coli*: 71%	71%	Leisenring et al. 2012
BioSwale	Fecal coliform: -5%[‡] *E. coli*: -6%	-6%	Leisenring et al. 2012
Dry Ponds	Fecal coliform: 80%	80%	Tilman et al. 2011
Erosion and Sediment Control	Assumed average of all urban stormwater practices: Fecal coliform: 53% *E. coli*: 60%	57%	
Filtering Practices	Fecal coliform: 60% *E. coli*: 99%	80%	Clary et al. 2008
Forest Buffers	Fecal coliform: 43 - 57%	50%	VDEQ 2003
Impervious Surface Reduction	Assumed average of all urban stormwater practices: Fecal coliform: 53% *E. coli*: 60%	57%	
Infiltration Practices	Assumed to be equivalent to Leisenring et al. 2012 retention ponds: *E. coli*: 95% Fecal coliform: 65%	80%	Leisenring et al. 2012
Retrofit Stormwater Management	Assumed average of all urban stormwater practices: Fecal coliform: 53% *E. coli*: 60%	57%	
Wet Ponds & Wetlands	Fecal coliform: 53% *E. coli*: 43%	48%	Leisenring et al. 2012
Abandoned Mine Reclamation	No estimate	Not included in reduction estimate	
Street Sweeping	Fecal coliform: 1.4- 4.3%	Not included in reduction estimate	Zarriello. et al. 2003
Tree Planting	No estimate	Not included in reduction estimate	

[‡] Negative removal efficiencies indicate that the concentrations of pathogens were increased as a result of the BMP implementation.

Table A.1. (continued)

Best Management Practice*	Loading Reduction Efficiency (%)	Avg Fecal Coliform and *E. Coli* (FIB)† Efficiency (%)	Reference
Street Sweeping	Fecal coliform: 1.4- 4.3%	Not included in reduction estimate	Zarriello. et al. 2003
Tree Planting	No estimate	Not included in reduction estimate	
Urban Stream Restoration	No estimate	Not included in reduction estimate	
	Septic Practices		
Combined Sewer Overflow Elimination	Fecal coliform: 99%	Not included in reduction estimate	CGR 2011
Septic Connections	Fecal coliform: 99%	Not included in reduction estimate	Vann et al. 2002; Petersen et al. 2009
Septic Denitrification	No estimate obtained	Not included in reduction estimate	
Septic Pumping	Fecal coliform: 5%	Not included in reduction estimate	VDEQ 2003
Treatment Plant Upgrades	No estimate: Heavily dependent on type of upgrade and technology implemented.	Not included in reduction estimate	

Table A.2. Agricultural and urban BMPs excluded from analysis because pathogen efficiency reductions were unavailable

Agricultural BMPs	Urban BMPs
Non-Urban Stream Restoration	Urban Stream Restoration
Livestock Waste Management Systems	Street Sweeping
Poultry Waste Management Systems	Septic Connections
Livestock Mortality Composting	Septic Denitrification
Poultry Mortality Composting	Septic Pumping
Manure Transport Outside CBWS	Combined Sewer Overflow Elimination
Manure Transport Within CBWS	Wastewater and Sewage Treatment Plant Upgrades
Poultry Phytase (layers+pullets)	Tree planting (distinct from forest conservation and forest buffers)
Poultry Phytase (broilers+turkeys)	Abandoned mine reclamation
Dairy Precision Feeding	
Ammonia Emission Reductions	
Continuous No-till	

Appendix B. Data Quality and Limitations

These analyses used the best available data and models at the time of analysis. Readers are advised to refer to the original data sources for detailed information about input data quality. For the pathogen analysis, results should be viewed as order of magnitude estimates, rather than precise numbers, because the approach used a model at one scale (Upper Potomac) to estimate results for a substantially coarser scale (Chesapeake Bay). The estimation approach was reviewed by two of the original model developers and they concurred that the approach was a reasonable transfer of their initial model results. Further, they were not aware of other transferable pathogen models that would improve estimation given the analysis resources available. However, the sources of estimation error could be reduced by a modeling effort designed specifically for the entire Chesapeake Bay watershed. In addition, these order of magnitude results are subject to change as the Chesapeake Bay states refine their WIPs and as new research becomes available to understand BMP effectiveness at pathogen reductions or how pathogens are attenuated in surface water bodies.

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