The Chesapeake Bay Program Modeling System: Overview and Recommendations for Future Development

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Key Words

Chesapeake Bay Program, NOAA Chesapeake Bay Office, Environmental Protection Agency, Modeling, Airshed Modeling, Watershed Modeling, Estuarine Modeling, Hydrodynamic Modeling, Biogeochemical Modeling, Living Resource Modeling, Chesapeake Assessment Scenario Tool, Total Maximum Daily Load, Chesapeake Bay Agreement

Abstract

The Chesapeake Bay is the largest, most productive, and most biologically diverse estuary in the continental United States providing crucial habitat and natural resources for culturally and economically important species. Pressures from human population growth and associated development and agricultural intensification have led to excessive nutrient and sediment inputs entering the Bay, negatively affecting the health of the Bay ecosystem and the economic services it provides. The Chesapeake Bay Program (CBP) is a unique program formally created in 1983 as a multi-stakeholder partnership to guide and foster restoration of the Chesapeake Bay and its watershed. Since its inception, the CBP Partnership has been developing, updating, and applying a complex linked modeling system of watershed, airshed, and estuary models as a planning tool to inform strategic management decisions and Bay restoration efforts. This paper provides a description of the 2017 CBP Modeling System and the higher trophic level models developed by the NOAA Chesapeake Bay Office, along with specific recommendations that emerged from a 2018 workshop designed to inform future model development. Recommendations highlight the need for simulation of watershed inputs, conditions, processes, and practices at higher resolution to provide improved information to guide local nutrient and sediment management plans. More explicit and extensive modeling of connectivity between watershed landforms and estuary sub-areas, estuarine hydrodynamics, watershed and estuarine water quality, the estuarine-watershed socioecological system, and living resources will be important to broaden and improve characterization of responses to targeted nutrient and sediment load reductions. Finally, the value and importance of maintaining effective collaborations among jurisdictional managers, scientists, modelers, support staff, and stakeholder communities is emphasized. An open collaborative and transparent process has been a key element of successes to date and is vitally important as the CBP Partnership moves forward with modeling system improvements that help stakeholders evolve new knowledge, improve management strategies, and better communicate outcomes.

1 Introduction

1.1 The Chesapeake Bay Program and its Modeling System

The Chesapeake Bay is the largest, most productive, and most biologically diverse estuary in the continental United States, providing crucial habitat for native plant and animal species, many of which are migratory (Boesch et al., 2001; Kemp et al., 2005). Natural economic benefits derived from the Bay have been valued at more than $100 billion annually (CBF, 2014). The Bay supports economically important fisheries, with blue crabs, striped bass, and oysters generating the largest revenue (Dewar et al.,

2009) and shellfish aquaculture activities growing rapidly (Hudson et al., 2016). The Bay waters also enhance coastal property values and support a vital tourist economy, including nature-based recreation industries (Klemick et al., 2018). However, increases in agricultural activity, urbanization, suburban sprawl, stream alterations, and air pollution since colonial times, and intensification since the mid-20th century, have led to excessive nutrient and sediment inputs entering the Bay (Brush, 2009), adversely affecting the health of the Bay ecosystem and the economic services it provides (CBF, 2014).

Since the mid 1900s increases in nutrient and sediment inputs to the Bay have led to a reduction in water clarity, expansion of hypoxia (DO < 2 mg O_2 L^{-1}) (Hagy et al., 2004; Williams et al. 2010; Bever et al., 2013), and increase in the occurrence of noxious biotic events like harmful algal blooms (HABs). Hypoxia reduces the catch per unit effort of fish that feed in deep waters of the Bay and can lead to fish kills (Buchheister et al., 2013). HABs can adversely affect the ecosystem by degrading water quality and can impact human health by contaminating shellfish they consume (e.g., via neurotoxic, amnesic, or diarrhetic shellfish poisoning; Glibert et al. 2005, Landsberg et al. 2006, Brookfield et al., 2021). HABs also adversely impact local seafood-related businesses through effects on shellfish populations and aquaculture (Gallegos & Bergstrom 2005, Tango et al. 2005, Marshall et al. 2008, Glibert & Burford 2017, Van Dolah et al. 2016). Recreational fisheries in the Bay are also sensitive to water clarity since visual predation is necessary for fishing lures to attract economically important game fish (MacDonald et al., 2009) and degraded water clarity is aesthetically apparent to coastal residents and tourists (Klemick et al., 2018).

The Chesapeake Bay Program (CBP) is a partnership formally created in 1983 to guide and foster restoration of the Chesapeake Bay and its watershed. This partnership includes all six states within the Bay watershed (Virginia, Maryland, Pennsylvania, West Virginia, Delaware, and New York) and the District of Columbia (D.C.), plus hundreds of federal, state, and local government agencies, academic institutions, and non-profit interest groups. The CBP Partnership formed out of concerns regarding the loss of submerged aquatic vegetation (SAV) and extensive low oxygen (hypoxic and anoxic) waters in the Bay, referred to as "dead zones", documented locally as early as the 1930's (Kemp et al., 2005).

Since its inception, the CBP Partnership has relied on a complex coupled modeling system to predict the watershed loads of total nitrogen, phosphorus, and sediment that the Chesapeake Bay can receive while still maintaining acceptable water quality in terms of dissolved oxygen concentrations, water clarity, and chlorophyll *a* concentration. The 2010 version of this coupled modeling system (Cerco et al. 2010; Linker et al., 2013a,b; Shenk and Linker, 2013) specifically estimated the total maximum daily loads (TMDLs) of nitrogen, phosphorus, and sediment that could be allowed to reach Bay waters such that the tidal water quality standards were still met, as mandated by the Clean Water Act.

The 2014 Chesapeake Bay Watershed Agreement marked a substantial advancement in the restoration effort for the Bay, with all seven major watershed jurisdictions signing onto an expanded vision of Bay management (CBP 2014). The 2014 Agreement outlines five themes related to Abundant Life, Clean Water, Climate Change, Conserved Lands, and Engaged Communities, and provides specific goals and measurable outcomes associated with targeted timelines and ecological endpoints to evaluate success within each theme. The 2014 Agreement also brought an important motivational shift in efforts to restore the Chesapeake Bay. The 2014 Agreement and its creative framework of themes, goals, and outcomes incentivizes the changes necessary to achieve the TMDL levels by clearly identifying and leveraging diverse outcomes of societal benefit and value to stakeholder communities in the watershed. However, the current CBP modeling system retains its historical focus primarily on the strict regulatory interpretation of the TMDLs and associated water quality outcomes.

Modeling outcomes inform the management plans of individual jurisdictions and the overall strategy of the CBP Partnership, specifically efforts to reduce point and nonpoint pollution sources including regulations designed to restrict pollutant transport into the Chesapeake Bay and U.S. coastal waters. Over the past four decades, the CBP modeling system has significantly evolved as understanding of processes operating in the Bay and its watershed have advanced and management questions progressed. The CBP modeling system released in 2017 has multiple components (airshed, land use, watershed, estuarine hydrodynamic and water quality models). These sub-models determine Chesapeake Bay TMDLs in that they are used to force, either directly or indirectly, the biogeochemistry model that predicts changes in oxygen concentration, water clarity, and chlorophyll *a* concentration that result from changes in nutrient and sediment loads. The CBP Partnership has also promoted the development of living resource models to advance habitat restoration for targeted estuarine species of concern. These include models of submerged aquatic vegetation (SAV) and benthic filter feeders directly linked to the estuarine hydrodynamic and water quality models, as well as stand-alone ecosystem models that simulate interactions of numerous higher trophic level species (e.g., fish, crabs) by using various outputs of the coupled modeling system as inputs.

1.2 History of the Chesapeake Bay Program Modeling System

The component models of the CBP modeling system and their coupling have been continually updated in response to emerging science and changing water quality and ecological management priorities since the 1980s (Linker et al., 2002; 2013a). Models have been periodically fixed at milestone states-of-development and then used to evaluate performance of investments implemented to meet TMDL targets for nitrogen, phosphorus and sediment, and to assess the sufficiency of strategies to raise dissolved oxygen concentrations in the Bay to levels determined necessary to support estuarine ecosystem services. It is important to emphasize that the Chesapeake Bay TMDLs are specifically designed to meet water quality standards to support living resources. For example, the limits on deep-water dissolved oxygen concentrations have been established to protect juvenile and adult fish as well as shellfish (See Table 2 in Irby and Friedrichs, 2019 and Tango and Batiuk, 2013).

Modeling results have also been used to evaluate water quality standards related to the proliferation of SAV, as well as necessary thresholds for water clarity and chlorophyll *a* concentrations (USEPA 2010). The TMDL targets were first legally formalized by the CBP Partnership in 2010 (USEPA, 2010), and they were updated most recently with the 2017 "Midpoint Assessment" (USEPA, 2018) using the 2017 version of the CBP modeling system to evaluate the contemporary state of the restoration.

Developments in the CBP modeling system from its inception in 1982 to the 2017 milestone (i.e., 2017 Midpoint Assessment) include substantial increases in spatial and temporal resolution in the component airshed, watershed, and estuarine models, and deeper integration with other modeling activities outside of the component models. An example of the deeper integration is the recent incorporation of the SPAtially Referenced Regressions On Watershed attributes (SPARROW; Ator et al., 2011) model with the most recent (2017/“Phase 6”) version of the watershed model. Web-based distribution of open source, public domain model source codes, executable models, data, results, documentation, tools to assess the effects of management actions on nutrient and sediment loads to the Bay, and general support of multiple models (Irby et al., 2016) have contributed to the development of the CBP modeling system. All of the sub-models are now open source and available for use and further development by the research and management communities, either directly through the internet or via request. These efforts have increased the transparency and accessibility of the CBP modeling system, provided opportunities for inter-model comparisons, increased stakeholder engagement, and fostered trust in the models and their predictions.

The general acceptance of the CBP modeling system for informing management decisions involved a deliberate and extensive process of review and engagement. Appendix C of the 2010 TMDL (USEPA 2010) lists 433 meetings where the TMDL and/or models used in the TMDL were the principal topics of the meeting (2005-2010) and 297 additional meetings where the TMDL and/or models were on the agenda (2008-2010). The meetings occurred both within the committee and workgroup structure of the CBP, at federal, state, and non-governmental partner organizations and through scheduled public forums and webinars. Generally, stakeholder working groups, primarily the CBP’s Water Quality Goal Implementation Team and its workgroups, determine how the models will be used to assist decision-making. These groups are also charged with determining appropriate model inputs related to land use, agricultural systems, and management actions according to the best available data. Technical working groups, primarily the Modeling Workgroup, determine the structure and parameterization of the models and inputs such as atmosphere and ocean forcing functions. The CBP's Scientific and Technical Advisory Committee (STAC) also plays two key roles in model development. STAC supports broadly attended workshops that encourage cross-fertilization of ideas and result in scientific recommendations to the CBP that drive the development of models. The foundation of this paper is one such workshop. STAC also forms committees that perform independent scientific peer reviews of the models (e.g., Easton et al., 2017). The overall review process and the roles of different groups within and outside of the CBP are discussed in section 1 of the Chesapeake Assessment Scenario Tool (CAST) documentation (CBP, 2020a).

The participatory development process has expanded in scope over time, with earlier models primarily receiving scientific review and later models increasingly receiving review and input from the stakeholder community. The process resulted in a steady evolution of the models so they were up-to-date but also grounded with empirical information. This multi-decade process of development and feedback has led to a linked modeling system with sufficient transparency and accrued trust so the results are accepted by a wide range of managers and stakeholders, as well as by the scientific community. The multi-decadal process is ongoing with strong leadership provided for many years by a small group of people at the CBP (including co-authors R. Batiuk, L. Linker and G. Shenk). This experience with the CBP modeling system provides a template for how complicated models can be developed and directly used to inform large-scale management decisions.

As part of the 2017 Midpoint Assessment, the CBP Partnership concluded there were no “fatal flaws” in the milestone 2017 modeling system (i.e., an absence of flaws substantial enough to invalidate its use for decision-making by the CBP Partnership). The 2017 modeling system provided improvements over previous versions (CBP, 2020a) and incorporated feedback from the scientific community and key stakeholders. Three key groups reviewed the 2017 CBP modeling system: (1) the Scientific and Technical Advisory Committee (STAC), composed primarily of scientists that advise the CBP; (2) the Water Quality Goal Implementation Team and its workgroups, whose membership includes managers, stakeholders, non-governmental organizations, and scientists, and (3) technical managers and scientists in the Modeling Workgroup (CBP, 2020a, section 13). Starting in 2019, federal, state, and local jurisdictions have been applying the 2017 modeling system to aid in the development of the Phase III Watershed Implementation Plans . These are plans of local management actions, designed for their jurisdictional waters to meet the TMDLs, that will guide water quality management in the Chesapeake Bay region until the scheduled Bay-wide assessment in 2025. The science- and modeling-based approach to coproduce knowledge, formulate solutions to problems and adaptively guide restoration activities is fundamental to the environmental management approach of the CBP Partnership and will continue into the foreseeable future, particularly as plans are now being made for a next generation modeling system to incorporate new science and monitoring, expand the capability of the models, and to assess the challenge of 2035 climate change to achieving Chesapeake water quality standards.

1.3 Management Perspectives

The CBP modeling system was developed specifically to inform management. Formulations and testing of the models are therefore driven by regulatory management needs. The linked models are regulatory models, distinct from parallel computational platforms for exploratory, research-oriented modeling activities that are also ongoing in the Bay region (e.g., Xu and Hood, 2006; Xu et al., 2011; Feng et al., 2015; Wiggert et al., 2017; Zhang et al., 2015; 2016; Irby et al., 2016; Irby and Friedrichs; 2019; St-Laurent et al., 2020; Ator and Garcia, 2016; Ator et al 2019; Testa et al. 2017). As a tool for management with specific deadlines and milestones, the CBP modeling system must also be available and ready to be used for the next set of questions and decisions on a schedule that meets management deadlines. Examples of major court and management policy mandated deadlines were the assessments for the 2010 TMDL, 2017 Midpoint Assessment, and upcoming 2025 assessment.

Environmental managers from watershed jurisdictions use results from the coupled modeling system to guide water quality management decisions within their local subregions. A major use of the modeling system is to develop equitable nutrient and sediment loading targets across state and local jurisdictions and inform efficient implementation of best management practices (BMPs). Managers use the modeling system to: 1) set nutrient and sediment reduction targets; 2) configure nutrient and sediment reduction plans to meet the targets; and 3) quantify progress towards the implementation of reduction plans and local and Bay-wide restoration goals. The expansion of the restoration goals in the 2014 Agreement inspired further consideration of whether and how the coupled modeling system can be used beyond the past focus on prescriptive water quality issues. Under the 2014 Agreement, managers need information to assess progress related to living resources in the Bay and its watershed and consider the effects of climate change, and would benefit from models that can populate a decision-support system to analyze trade-offs and co-benefits, and to further encourage stakeholder engagement.

Chesapeake Bay restoration accomplishments, including development of the linked modeling system, are cultivated through collaborations among government and non-government researchers (primarily academics), creating opportunities for engagement among groups of managers and scientists who monitor, measure, test, and model processes relevant to the entire socioecological system comprising the Chesapeake Bay and its watershed. These collaborations open novel opportunities for model-based experiments to test hypotheses and validate CBP model findings. This is where the regulatory CBP modeling system and research-oriented models intersect, inform, and influence each other. Model comparisons can reveal consistencies and contradictions between CBP model findings and other models or observations (e.g. Irby & Friedrichs, 2019; St-Laurent et al., 2019), resulting in an enhanced understanding of underlying assumptions and processes that ultimately improve the CBP modeling system.

The CBP modeling system informs Bay policy and funding decisions that involve billions of dollars in public and private environmental investments (see https://www.epa.gov/sites/production/files/2016-03/documents/2015-06.pdf). Therefore, the CBP modeling system has to be scientifically defensible, transparent, timely, useful to manage current environmental issues, and representative of the needs of stakeholders in multiple jurisdictions. Field research and monitoring data are crucial components used to develop and evaluate models whose outputs are used to guide management efforts. Targeted research informed by modeling is essential to efficiently advance resource management (Nichols and Williams 2006). These tools must also assist managers to quantify benefits, costs, uncertainties, and risks. Models that contribute to satisfying these requirements create a consistent documented foundation to base legislation, regulations, and investments.

1.4 Successes and Emerging Challenges

The CBP modeling system has played a crucial role in recent management successes. These include the achievement of the 2025 goals for nitrogen and phosphorus pollutant load reductions collectively from hundreds of Chesapeake Bay watershed municipal and industrial wastewater treatment facilities a decade early (Dance, 2016). In addition, trends in recent years suggest that the summertime anoxic volume (i.e., dead zone) is decreasing (Figure 1, and see Ni et al. 2020) and SAV has shown signs of recovering (Lefcheck et al., 2018; see also CBP 2020c).

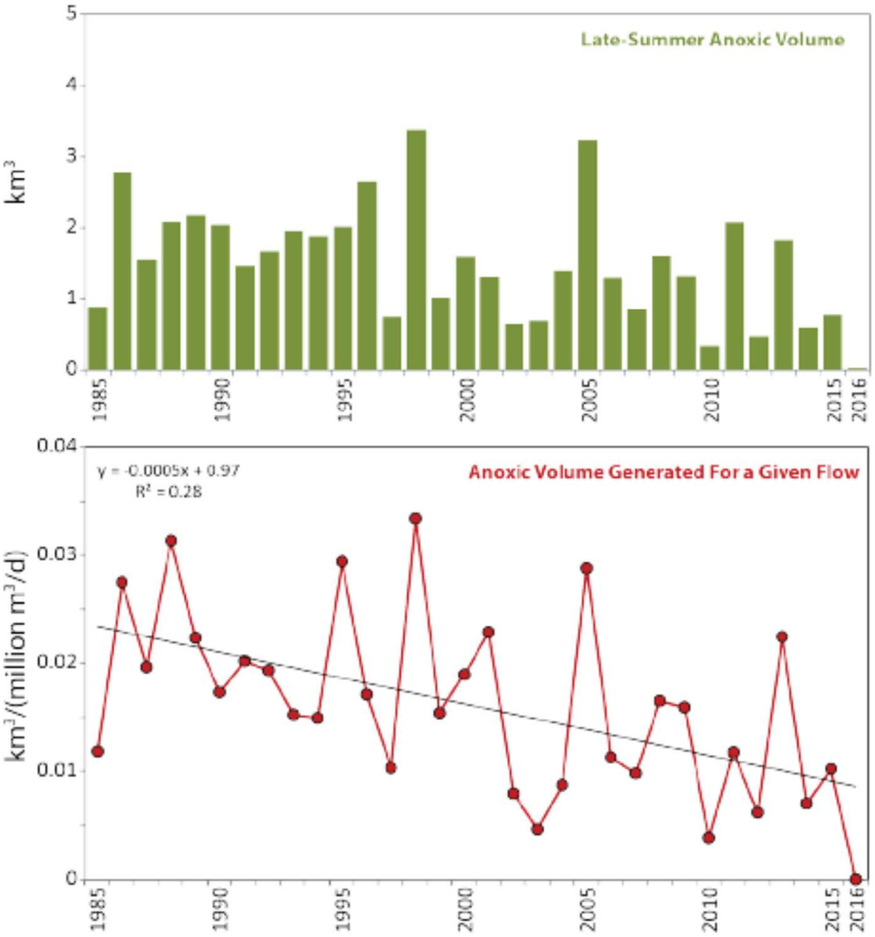


Figure 1: Late summer anoxic volume (top panel) and late summer anoxic volume normalized to flow, the latter showing a pronounced decline from 1985 to 2016.

However, the CBP modeling system will need to evolve and advance to address new challenges to provide managers with relevant information. The most daunting future challenge is ensuring the modeling system can inform management decisions under a changing natural and human environment. Globally influenced changes in regional weather patterns and sea level rise are affecting temperature, watershed dynamics, estuarine hydrodynamics, biogeochemistry, and ecology (e.g., Irby et al. 2018, Lefcheck et al. 2017, St-Laurent et al., 2020, Testa et al. 2018, Ni et al. 2020). In addition, increasing human population in the watershed will continue to influence stressors that will interact with the effects of climate change and sea level rise. The human population in the Chesapeake Bay watershed is projected to increase by about 12 percent from 2010 to 2025 (17.3 million to 19.4 million) (CBP, 2020b). The 2014 Agreement explicitly addresses climate change with goals and outcomes related to climate resiliency, monitoring, assessment, and adaptation. These considerations challenge the CBP modeling system to ensure that the modeling results can inform these goals.

In addition to its recognition of climate change relevance, another notable aspect of the 2014 Agreement is its identification of goals and targets that go beyond water quality-based metrics. For example, the 2014 Agreement highlights consideration of the effects that water quality has on tidal and nontidal living resources which, beyond SAV and benthic filter feeders, have not been a prior focus. The modeling system would need to expand its capabilities to other species in order to support multiple objective decision-making that could better encompass the associated broader set of goals, such as the simultaneous impacts to habitat quality and quantity, a variety of aquatic organisms, and fisheries harvests in response to restoration. Relating management-induced water quality responses to living resources is a formidable task considering the diverse species and habitats involved and that multiple factors beyond Bay water quality and habitat (e.g., ocean conditions and societally driven global and local harvests) can also affect most living resources.

Another class of challenges centers on the scale of the predictions from the CBP modeling system. High-resolution simulations to guide the design, implementation, and performance evaluation of optimal water quality management practices at local scales are in high demand. For example, the current watershed model in the CBP modeling system averages many conditions for a given land use within a county, potentially obscuring the importance of implementing best management practices where they can best reduce and prevent nutrient and sediment runoff (Easton et al., 2020). Local and state governments responsible for implementing management actions related to the TMDLs have expressed interest in maximizing co-benefits of their investments on nutrient and sediment controls. Co-benefits are ecosystem services that achieve nutrient and sediment reduction objectives while also addressing 2014 Agreement outcomes related to flood control, open space amenities, recreational uses, terrestrial species habitat, and healthy fisheries. Some CBP managers need tools that predict localized responses of interest (e.g., nontidal stream health) while others need tools suitable for integration across jurisdictions to achieve regional and Bay-wide goals.

2 The CBP Modeling System and Recommendations for Future Development

This paper summarizes the results of a 2018 workshop designed to identify needed modifications and advancements to the CBP modeling system to address the emerging management questions and challenges spurred by the 2014 Agreement and scheduled to be assessed in 2025. The workshop (Hood et al., 2019) involved academic and government scientists and managers active in the CBP Partnership who were specifically tasked to review the status of the modeling system (as of 2017), reflect upon the history of CBP modeling efforts, and offer guidance on future research needs and priorities. The development of the CBP modeling system offers lessons learned that are relevant to other large watershed-estuarine systems facing similar water quality impairments and management challenges.

As mentioned above, the CBP Modeling System (Figure 2) comprises airshed, land use, watershed, and estuarine models. The airshed model predicts changes in atmospheric deposition of inorganic nitrogen and other selected species on the watershed and tidal Bay due to changes in emissions. The land use model predicts changes in land use, sewage, and septic systems in response to shifts in population, expected housing and commercial property demand, and land use policy. The watershed model combines the output of these models with other data sources, such as implemented BMPs and the US Census of Agriculture, to predict the point source and non-point source (distributed) loads of nitrogen, phosphorus, and sediment entering the Bay, for the nine major tributary rivers and along shorelines of the Bay and its many estuarine tributaries. The estuarine hydrodynamic and biogeochemistry models predict variations in Bay circulation and water quality due to changes in input loads provided by the watershed model, changes in atmospheric forcing, and regional effects of climate change (i.e., sea level rise and changes in precipitation and temperature). In addition, the biogeochemistry model can simulate the impacts of changes in water quality on SAV and benthic filter feeders. Finally, there are currently two living resource models developed by the NOAA Chesapeake Bay Office (NCBO) that are not part of the CBP Modeling System but can use output from the estuarine hydrodynamic and biogeochemistry models to assess how changes in water quality due to management actions might impact higher trophic levels.

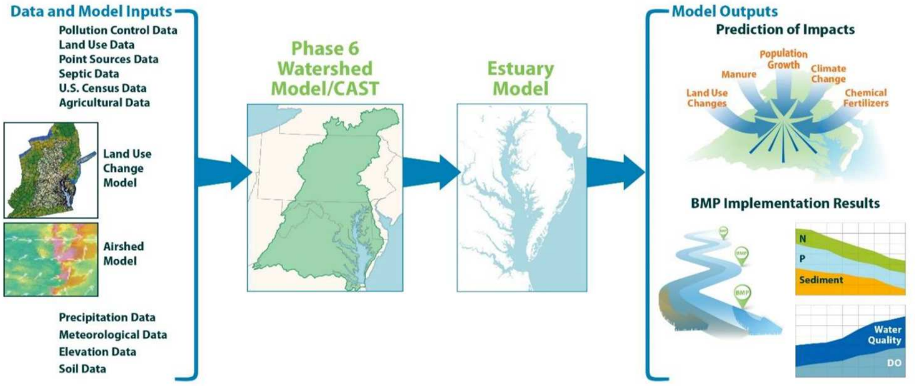


Figure 2: The Chesapeake Bay Program management modeling system. This system includes an Airshed Model, a Land Use Change Model, a Watershed model, and an Estuary Model (that includes both Hydrodynamic and Biogeochemical components). The full CBP management modeling system is specifically designed to set TMDLs, inform the development of WIPs and track progress toward achieving restoration goals in Chesapeake Bay.

The components of the CBP modeling system, including the NCBO living resource models, are all “loosely coupled” in that data are transferred manually from one sub-model to another (i.e., the output data of one model are transferred to a "downstream" model). This loose coupling has the advantage of facilitating work flow because the CBP sub-models are often used for separate tasks and at separate times. For example, the estuarine biogeochemistry model determines the allowable nutrient and sediment loads that will meet water quality standards and is run infrequently for goal setting. The watershed model typically has thousands of scenarios run each year on how to achieve the allowable loads. The airshed model provides a limited range of national emission reduction scenarios from the Clean Air Act that can be considered by CBP decision makers. Loose coupling also improves scenario analysis efficiency, that is, it is easier to work on a single model for the numerous sensitivity runs and tests that are required for each launch/application of the CBP models. Note, however, that some of the CBP sub-models are fully coupled, i.e., where the feedback loops are too critical to do otherwise. For example, the

SAV and benthic filter feeder models, mentioned above and described in section 2.4.1, are directly coupled to the estuarine biogeochemistry model so that they can provide continuous dynamic feedback to one another.

2.1 Airshed Model

2.1.1 Overview of the 2017 Airshed Model

The airshed model in the coupled system is the open-source Community Multiscale Air Quality (CMAQ; Figure 3; Foley et al., 2010). CMAQ itself consists of a series of coupled models (meteorological, emission, and photochemical air quality) that work in concert to estimate the emissions and fates of atmospheric gaseous and particulate pollutants (acid, nutrient, or toxic) and their precursors (Foley et al., 2010). CMAQ predicts the fate of these pollutants as they transport through the airshed and deposit back to Earth's surface or react to form secondary pollutants.

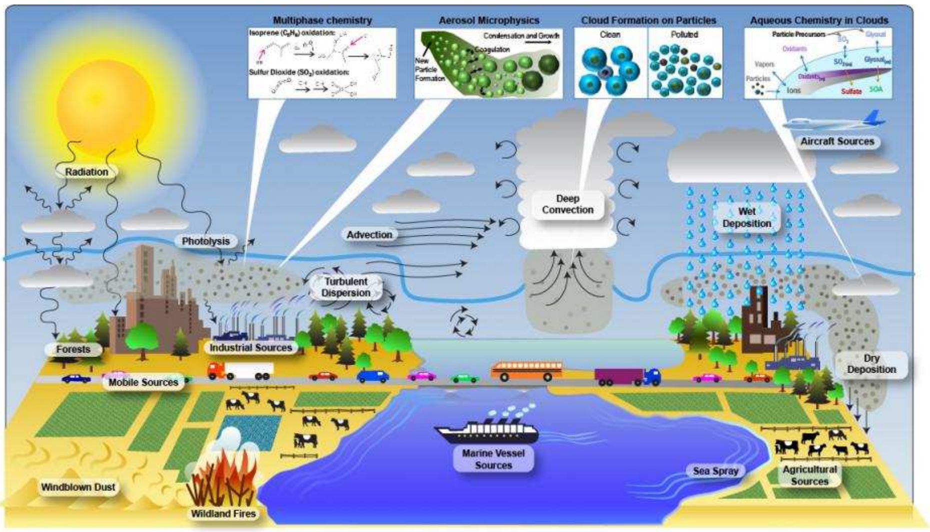


Figure 3: Schematic diagram of CMAQ airshed model that simulates transport, chemistry and deposition (gaseous and precipitation scrubbing) for ozone, particulate matter, toxics, acids, trace gases, etc., simultaneously.

CMAQ is maintained by the U.S. EPA Atmospheric Science Modeling Division, and since its initial release in 1998, CMAQ has been widely used to evaluate potential national, regional, and state-specific air quality policy management decisions. CMAQ can be used to explore different meteorological and atmospheric pollutant emission scenarios (Campbell et al. 2019). For example, CMAQ is often used to test the impact of future emission regulations on deposition and determine which individual emission sources are the largest contributors to air pollution at a site (Zhang et al. 2012). CMAQ's generalized and flexible formulation has enabled incorporation of alternate process algorithms and numerical solutions to include new science in the model to address increasingly complex air pollution issues.

CMAQ requires two primary types of inputs: meteorological information and emission rates from sources that affect air quality. The CMAQ version 5.0.2 model used with the 2017 CBP modeling system has a 3-dimensional domain that covers the North American continent at a 12 x 12 km grid scale (Figure 4) that includes the Chesapeake Bay watershed and Bay tidal waters. The model uses year-specific meteorological inputs from the Weather Research and Forecasting (WRF) model, and combines hourly emissions data from the U.S. EPA's National Emissions Inventory with the open-source Sparse Matrix Operator Kernel Emissions (SMOKE) model to estimate the magnitude and location of pollution sources. CMAQ then calculates atmospheric transport, transformation, and deposition of a suite of anthropogenic pollutants including ozone, particulate matter, toxics, acid deposition, and several forms of oxidized (e.g., NOx), and reduced (e.g., NH_3) nitrogen. The 2002 to 2012 CMAQ simulations used the bidirectional NH_3 exchange option where the surface ammonia flux is modeled as a gradient based process that can result in emissions from land use with enriched ammonium concentrations in the soil or vegetation (e.g., agriculture) or deposition to land to better capture the observed variability in NH_3 dry deposition (Bash et al. 2013).

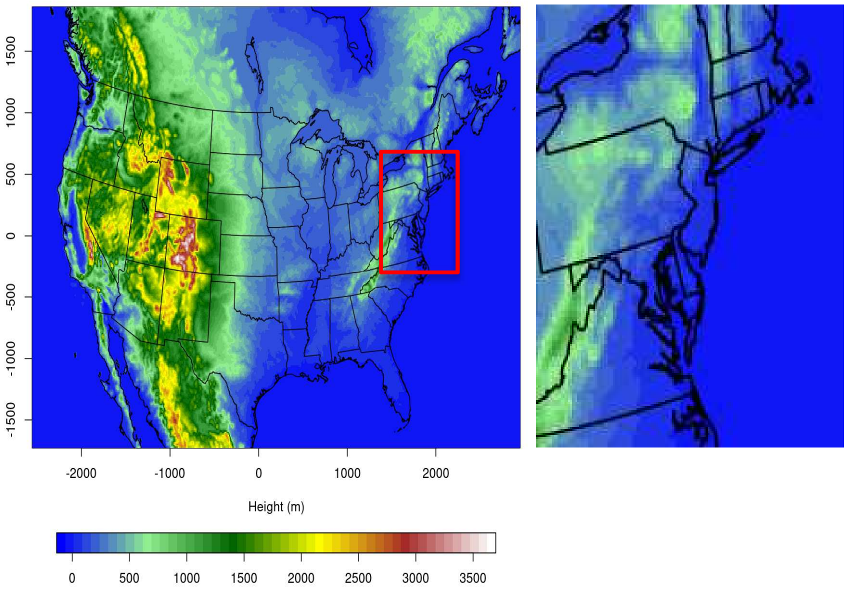


Figure 1: CMAQ model 12 x 12 km grid / topography over North America (left map) and the Chesapeake Bay region (right map). The pixilation in the topography reveals the 12 x 12 resolution.

CMAQ is continuously evaluated against network, satellite, and field sampled atmospheric chemistry and air quality observations. CMAQ effectively describes seasonal variability and trends (2002-2012) in oxidized and reduced nitrogen wet deposition and in ambient oxidized nitrogen concentrations over its broad domain, which gives confidence that wet and dry deposition of nitrogen to the Chesapeake Bay watershed are also simulated well (Bash et al., 2013; Zhang et al., 2019a).

2.1.2 Airshed Model Advantages and Limitations

The CMAQ model is based on first principles and is not calibrated to specific monitoring stations. However, CMAQ is routinely evaluated against network observations to assess its performance in capturing the magnitude and trends in ambient concentrations and wet deposition at monitoring sites (Appel et al. 2020; Zhang et al., 2019b; Kelly et al., 2019). Starting with CMAQ v5.3, there is an option for land-use-specific dry deposition (Appel et al. 2020). This option allows better integration of flux estimates from the model grid cells (with a grid spacing on the order of 10s of km) with critical loads assessments and dry deposition impacts on water quality where finer scale details are necessary due to differences in the retention and sensitivity of different land uses to pollutant/nitrogen deposition.

In the Chesapeake Bay watershed, however, where dry deposition accounts for approximately half of the atmospheric nitrogen loading, modeling of this deposition cannot be sufficiently evaluated due to the lack of a routine dry deposition monitoring network. To mitigate this uncertainty, model algorithms were developed from field scale observations (e.g., Bash et al., 2013) but this results in uncertainty for land use types where these detailed measurements are absent. Additionally, ambient atmospheric measurements of nitric acid and ammonia, primary components of nitrogen dry deposition, are uncommon making even a qualitative evaluation difficult (Wang et al., 2021). Improvements in satellite air-quality measurements, specifically NH_3 (Wang et al., 2021), are filling in many of these measurement gaps but do not yet have a sufficient history of observations to assess trends in ambient concentrations.

2.1.3 Airshed Model Summary Recommendations

In the short-term, research should focus on the influence of climate change on the atmospheric deposition of nitrogen as wet, dry, and organic nitrogen deposition. The CMAQ model should include the full characterization of organic nitrogen deposition, including pollen and other particulate forms, to better constrain mass balances of nitrogen deposition to surface waters in watersheds and coastal systems. Organic nitrogen deposition can be an important atmospheric nitrogen source in many areas, and is currently underestimated by CMAQ 5.0.2 and earlier versions in most areas.

A second area of short-term focus is better quantification of the biases in predicted oxidized nitrogen concentrations in CMAQ to improve the accuracy of model predictions for the Chesapeake Bay watershed portion of the grid. Land-use-specific deposition estimates should be adopted that have been validated against field measurements. The process of validating the deposition estimates will help guide efforts to reduce parameter uncertainty and provide loading estimates that are more relevant to watershed transport processes and mitigation (e.g., riparian buffers).

In the long-term, CMAQ should be run at a higher spatial resolution with a non-uniform horizontal grid and apply methods to enable more complete quantification of the effects of parameter uncertainty on model predictions. A more resolved model grid could improve prediction for the Chesapeake Bay subregion because the current 12 x 12 km resolution fails to fully resolve the observed spatial variability in atmospheric deposition, especially with deposition related to sea breezes, along major transportation corridors, and for other processes dependent on local scales. An unstructured grid would allow for higher resolution where it is needed, while also keeping computational demands reasonable. Output from CMAQ is used as inputs to other models and sensitivity analyses would provide the basis for propagating uncertainty through the coupled modeling system.

Other long-term priorities are developing the ability to make more direct connections to the watershed model and estuarine hydrodynamic and biogeochemistry models, and evaluation of parameterization throughout the entire model domain. More direct connections to other CBP models that receive deposition predictions (e.g., providing CMAQ with specific information about land use from the watershed model and/or specific information about temperature and heat exchange from the hydrodynamic model) would enable the model to better capture feedbacks.

2.2 Land Use Change Model

2.2.1 Overview of the 2017 Land Use Change Model

The U.S. Geological Survey began developing the Chesapeake Bay Land Change Model (CBLCM) in 2012 (Figure 5) to inform how land use planning and land conservation decisions would impact water quality and wildlife habitats. The CBLCM was developed in response to two STAC-sponsored independent scientific peer reviews of earlier modeling efforts at forecasting land change effects. The reviews emphasized, among other issues, the need to simulate multiple future scenarios and to explicitly quantify and communicate model uncertainties (Pyke et al., 2008 and Pyke et al., 2010). Results from the CBLCM are used to inform the locality-specific Phase III Watershed Implementation Plans developed by Maryland, Pennsylvania, West Virginia, and several counties in Virginia (WIP, 2019a,b,c,d).

The CBLCM is a pseudo-cellular automata urban growth model that stochastically simulates the future footprint of residential and commercial development associated with growth in population and employment (Figure 6; see also output posted on the Phase 6 viewer at: https://chesapeake.usgs.gov/phase6/map/). The 2017 version of the model incorporates data from 2013 onward to forecast annual development to 2025 at a 30-meter cell resolution, and associated conversions of forest and farmland and changes in the populations served by sewer or septic systems. The model’s forecasts are based on: (1) state-sanctioned projections of population and employment; (2) population and housing data and trends reported by U.S. Census Bureau; (3) land-cover trends derived from the National Land Cover Database (Homer et al., 2015); (4) mapped protected lands and sewer service areas; and (5) county-level zoning data (for the baseline scenario).

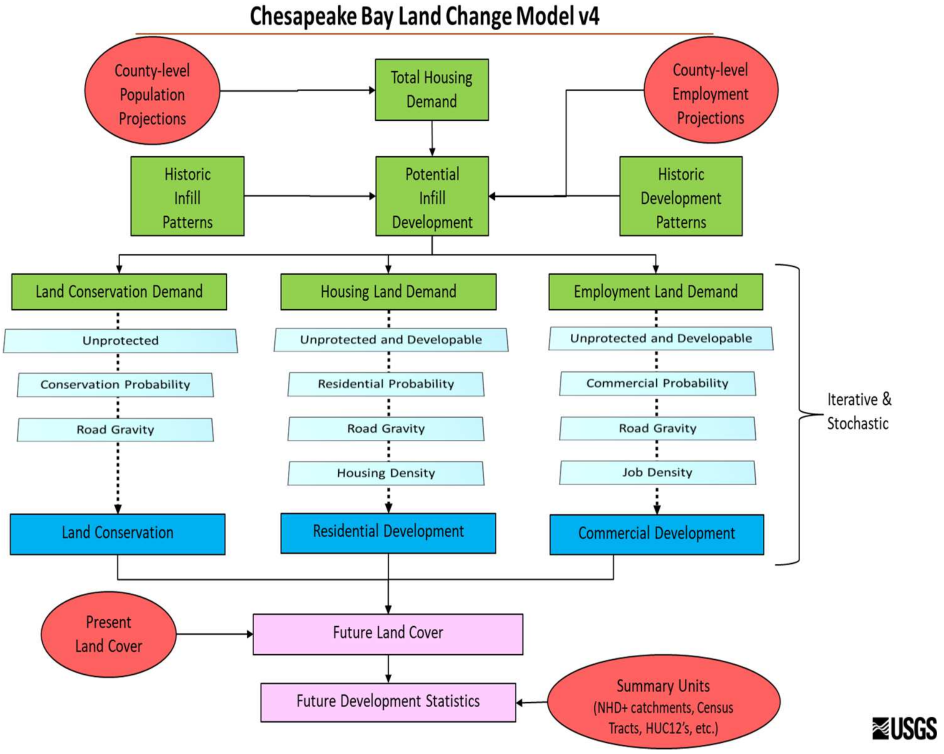


Figure 5: Diagram of the Chesapeake Bay Land Change Model.

The CBLCM simulates residential and commercial growth within individual counties by first assessing the amount of future county-level housing and employment that will occur as infill or redevelopment within Census urbanized areas. Remaining future housing and employment represent demands for greenfield residential and commercial development, respectively. Greenfield residential development is simulated by stochastically allocating seed cells of residential growth onto a residential probability surface. The residential probability surface is derived through logistic regression, comparing randomly-sampled observations of growth within residential areas (e.g., change in National Land Cover Database

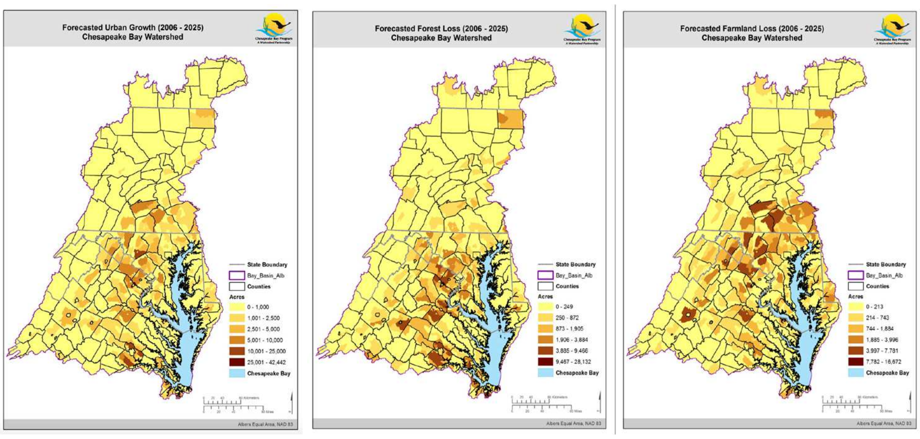


Figure 2: CBCLM projected (2006-2025) future urban growth (left panel), forest loss (middle panel) and farmland loss (right panel).

developed area, classes 21-24, within Census Block Groups with a housing to jobs ratio greater than one) with randomly-sampled explanatory variables estimated for all areas suitable for growth (i.e., unprotected, gently sloped, and undeveloped lands). A residential seed will sprout and grow into a patch of residential development if the value of the probability surface at the seed-cell location exceeds a random value assigned to the seed. The patch growth routine is the pseudo-cellular automata part of the model. Seed cells grow over a “resistance” surface weighted by proximity to the seed and proximity to the nearest road. Residential patch size potentials for each seed are randomly selected from the observed patch size distribution of residential development occurring between 2001 and 2011. As a patch is grown, households are accumulated within the patch from an underlying housing density surface. Patches stop growing when, either the maximum patch size is reached, the county-level demand for housing is met, or localized obstructions to growth (e.g., roads, steep slopes, open water, protected lands) prevent the patch from achieving its assigned size. This entire process is repeated for greenfield commercial growth using the greenfield demand for jobs and raster surfaces representing employment probability, job density, and commercial areas.

For any given future scenario, the CBLCM simulates 101 independent Monte Carlo iterations, the results of which are then averaged by watershed model land-river segment (described in Section 2.3). These stochastic iterations enable the assessment of model uncertainty associated with the growth allocation process. For every land-river segment, the relative standard deviation of future development is calculated to model uncertainty. In addition, the results of the logistic regressions are saved for every scenario and can be inspected to understand the explanatory power of the residential and commercial probability surfaces.

In addition to the baseline “current zoning” scenario, the CBLCM is capable of simulating alternative future scenarios of residential and commercial development through adjustments to the county-level population and employment projections, proportion of infill-to-greenfield development, and proportion of urban-to-rural development. Areas suitable for development, housing and employment densities, and the extent of sewer service areas can also be adjusted uniquely for any given scenario (e.g., Figure 7). To support development of the Phase III Watershed Implementation Plans, 13 alternative future scenarios representing 2025 land use conditions were created by the CBP Partnership and run through the CBLCM and watershed model. These scenarios include: “Historic Trends”, “Current Zoning”, “Forest Conservation”, “Agricultural Conservation”, “Growth Management”, and eight custom jurisdictional scenarios known as “Land Policy BMPs” for the jurisdictions of the D.C., Delaware, Maryland (3 scenarios), Pennsylvania, Virginia, and West Virginia. Descriptions of these scenarios can be found in the scenario section of the user documentation for the Chesapeake Assessment Scenario Tool (CBP, 2020b).

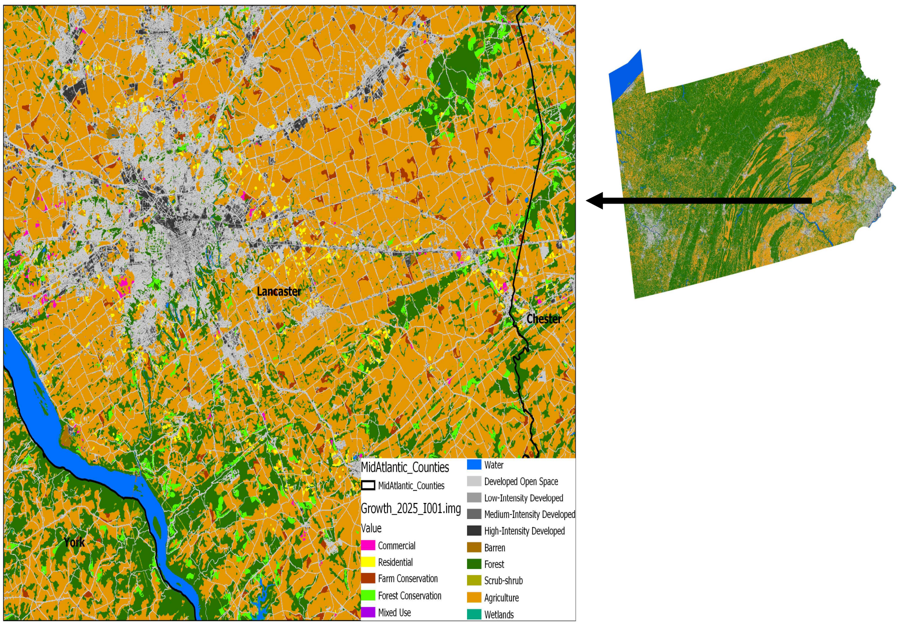


Figure 3: Simulated year 2025 residential, commercial, and mixed-use development and farm and forest land conservation in southeastern Pennsylvania, USA.

The CBLCM is designed to simulate plausible, long-term average levels of residential and commercial land use change. Unlike land cover change, which can be consistently observed by satellites over time, land use change is challenging to validate because few areas have been consistently mapped for land use over multiple time periods. Fortunately, Maryland maintains statewide tax parcel data attributed to land-use and year-built fields which can be used to validate county-level rates of residential and commercial land consumption simulated by the CBLCM. Rates were estimated as acres of consumption per year, per housing unit, and per job. Modeled residential land consumption rates were compared from 2013-2025 against observed residential rates for 2001 - 2011 and 2011 - 2019. For most counties, the CBLCM simulated plausible but higher residential land consumption rates compared to observations over the more recent 2011-2019 period and lower residential land consumption rates compared to observations over the earlier period, 2001-2011. The nationwide housing boom occurred during the former period as did high levels of suburban sprawl development. In contrast, during the latter period, the nation was recovering from an economic recession. For commercial growth, the CBLCM simulated plausible but higher rates of land consumption over the single observation period, 2006-2016. This is largely due to an under-estimation of job densities (jobs/acre), particularly in rural counties.

2.2.2 Land Use Change Model Advantages and Limitations

For input to watershed and water quality models, accurately simulating land use is more important than accurately simulating land cover because it is more relevant to nutrient and sediment loading rates. For example, the land uses of turf grass and cropland have the same cover type, "herbaceous", yet very different nutrient inputs and yields. As another example, a land use model will simulate the entire footprint of large lot residential development, and not just the impervious portions.

As a regional land use model, some specific advantages of the CBLCM are that it simulates urban infill/redevelopment, residential, and commercial greenfield development and distinguishes between growth on sewer versus septic systems. The CBLCM can also simulate multiple stochastic iterations of growth per scenario enabling the quantification of spatial uncertainty. Moreover, the CBLCM estimates residential and commercial densities, which are necessary for deriving impervious cover from land use and are essential for land use planning.

The central limitation of the CBLCM is that it is challenging to validate because no states except Maryland have data for consistently mapping land use over multiple time periods. Accurately simulating urban land use change requires estimation and prediction of the demand for land, density of development, and the portion of growth attributable to infill and redevelopment. The CBLCM attempts to estimate and predict all three of these components, but validation to date has been very limited.

2.2.3 Land Use Change Model Summary Recommendations

Areas of focus for improving CBLCM in the long-term include further specification of possible futures and improved code design. The functionality and transparency of CBLCM could be improved by leveraging results from regional transportation models (e.g., Motor Vehicle Emission Simulator, MOVES; Koupal et al.; 2013; Kall et al., 2014; Liu, 2015) and household microsimulation models (e.g., Simple Integrated Land Use Orchestrator, SILO; Moeckel, 2017) and enhanced representation of population cohorts (i.e., by age and income) and employment sectors (e.g., services, administrative/financial, warehousing).

In addition, efforts should be undertaken to incorporate temporally dynamic feedbacks between development capacity, density, growth probability, and spillover, as well as spatial allocation of infill development and redevelopment within urban areas with limits based on wastewater treatment capacity. The high-resolution land use data for the Chesapeake Bay watershed could be used to exclude already developed lands from future greenfield development more effectively. The CBLCM should also be modified to allow simulation of future: 1) changes in cropping systems, pasture, and farm animals; 2) changes in forests including changes in composition, phenology, seral stage, and disturbance; and 3) conditions consistent with a range of Representative Concentration Pathways (RCPs) and sea-level rise scenarios and their associated population and employment projections. Finally, a modular design should be adopted using open-source code and leveraging cloud computing and storage resources.

2.3 Watershed Model

2.3.1 Overview of the 2017 ("Phase 6") Watershed Model

The watershed model estimates freshwater, sediment, nitrogen, and phosphorus loads to the Chesapeake Bay from multiple sources in the watershed and determines how different management actions would affect these loadings. There are two versions of the 2017 model which are constrained to produce identical output over the long term: (1) a time-averaged (average annual loads) version widely used by the CBP Partnership for scenario assessments and Watershed Implementation Plan development; and (2) a dynamic version used in calibration and to drive the estuarine models. For full documentation of both versions and the relationship between them, see CBP 2020a.

Time-Averaged Watershed Model

The CBP uses the time-averaged version of the 2017 watershed model in the Chesapeake TMDL to set planning targets, design implementation plans, and track the progress in implementation of nutrient reduction efforts relative to their goals (Chesapeake Assessment Scenario Tool, CAST; CBP, 2020a). CAST provides estimates of average annual loads that would be expected given ten years of typical weather conditions. Typical weather conditions were defined by the CBP during the TMDL process as the period 1991-2000. Importantly, this model is intended to calculate the nitrogen, phosphorus, and sediment load annually delivered to the tidal Bay from each land use within each segment, given a set of management options. Scenarios of management options may include land use estimates from the CBLCM, atmospheric deposition from CMAQ, specification of point source and septic system discharges, and implementation of urban best management practices and agricultural conservation practices, collectively referred to as BMPs.

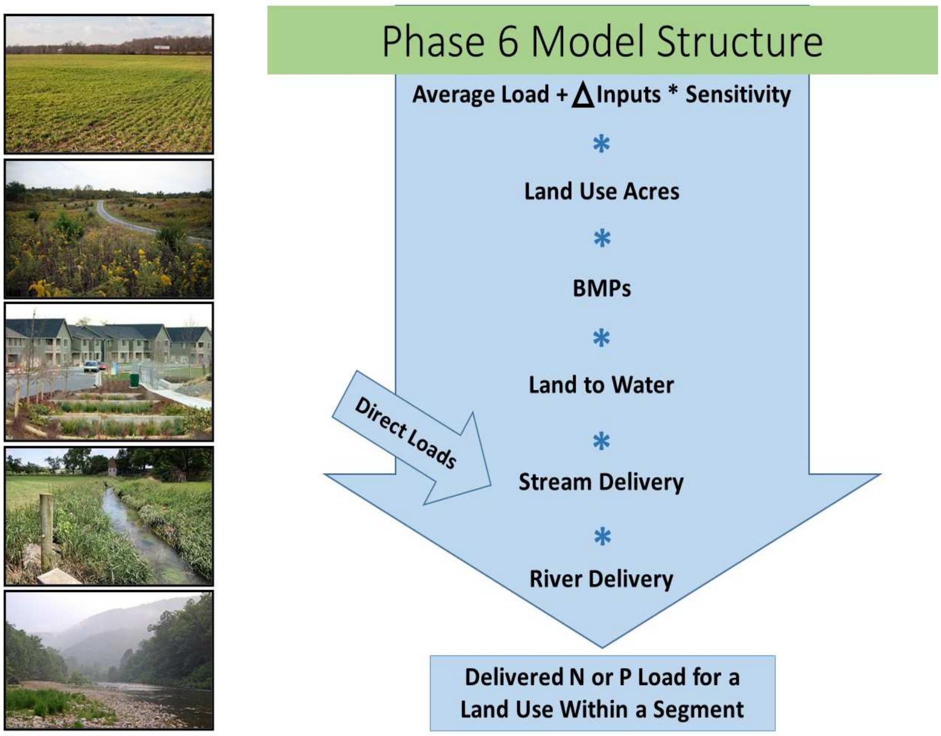


Figure 8: The Phase 6 Watershed Model Structure.

The structure of the time-averaged 2017 watershed model for nitrogen and phosphorus load predictions is organized by nine primary components (Figure 8). The approach for estimating nutrient loads involves several sequential computations. The top line in Figure 8 (*Average Loads, Inputs, and Sensitivities*) represents the calculation of water quality loads exported from a land use to a stream in a watershed segment, taking into account local applications of nutrients, but not local watershed conditions (e.g., watershed location, geology). The *average load* represents the Chesapeake Bay watershed-wide average annual load per acre for a given land use type, *Δ Inputs* represents the local deviation from the Chesapeake Bay watershed-wide mean input rate in pounds per acre for inputs such as fertilizer, manure, and atmospheric deposition. *Sensitivity* is the change in load to a stream from a unit change in an input. Sensitivity factors are specific to land use and input types.

After nutrient loads to a stream are derived from the initial step described above, the loads are then multiplied by the acres of the land use in the watershed segment (*Land Use Acres*) and modified by the effect of implemented local *BMPs*. *Land-to-Water* factors are then applied to account for spatial differences in loads due to physical watershed characteristics. *Land-to-Water* factors do not add to or subtract from the loads over the entire Chesapeake Bay watershed. Instead, they represent the spatial variability of nutrient delivery. The application of the four components results in an estimate of nutrient loads delivered to a stream or water body in a land-river segment.

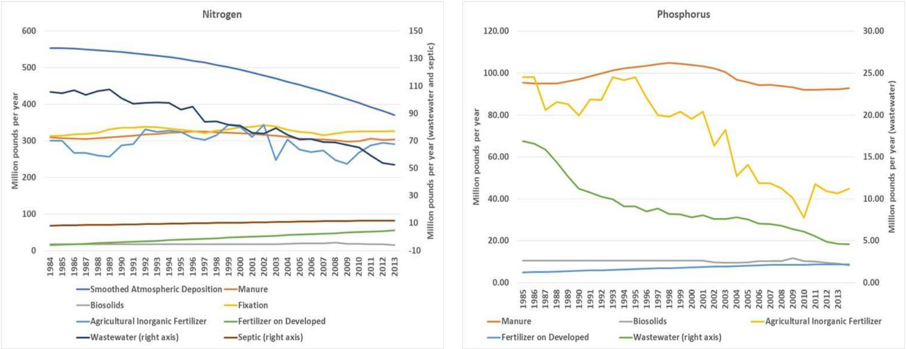


Figure 4: Major nitrogen (left panel) and phosphorus (right panel) inputs to the Phase 6 Model. Note that wastewater and septic are plotted on the right-hand axes, which is enlarged by a factor of four reflecting the approximate difference of the delivery of nutrients deposited on land and discharged directly to waterways. The atmospheric deposition is the expected deposition over the 10-year period of hydrology 1991-2000 given emissions in the indicated year.

After nutrient loads delivered to a stream estimated in the previous step, *Stream-Delivery* factors are then applied to account for processes influencing nutrient concentrations in stream flows with a mean annual discharge rate less than 100 cubic feet per second. Conceptually, these are attenuation factors that act to decrease nutrient delivery in small streams, as the loads move downstream to the boundary of the larger river reaches.

River-Delivery factors account for nutrient attenuation processes in the larger rivers. Finally, *Direct Loads* are nutrient loads that do not come from the land surface or subsurface and include point sources, stream bank erosion, and direct deposition of livestock manure in streams. Figure 9 shows the major nitrogen and phosphorus inputs to the time-averaged model from 1985 to 2019, and Figure 10 shows annual mean total nitrogen and phosphorus loads in the Chesapeake Bay watershed for the year 2017 simulated by the time-averaged model.

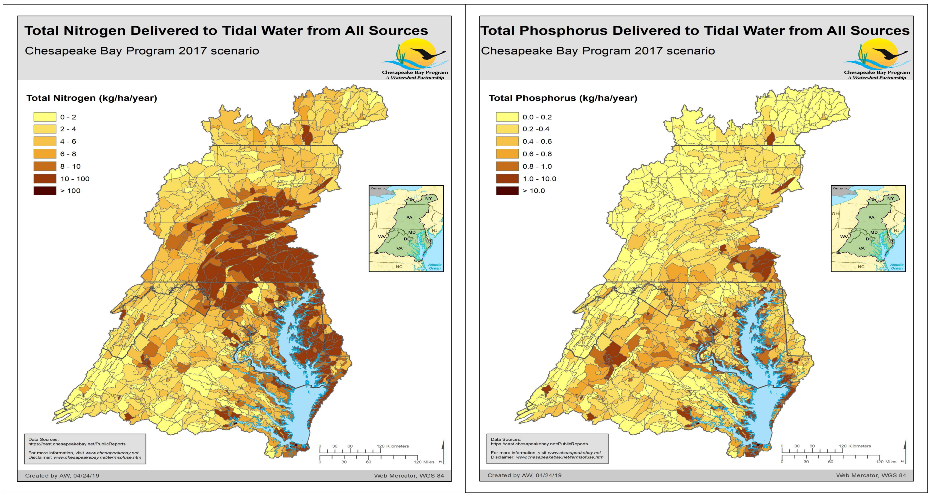
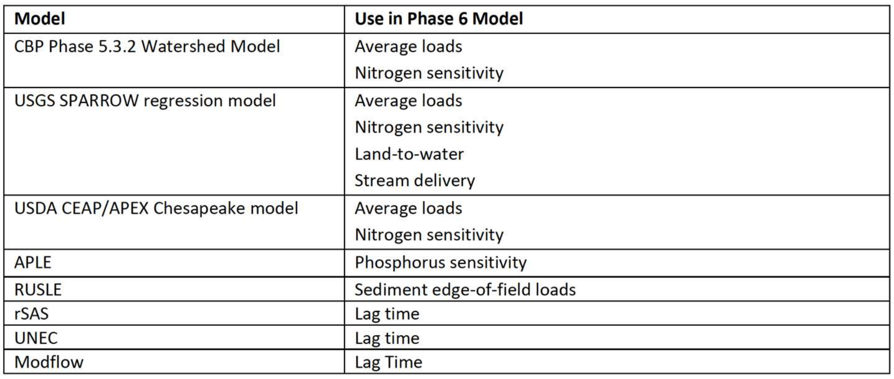


Figure 5: Spatial sources of total nitrogen (left panel) and total phosphorus (right panel) loads simulated by the time-averaged Chesapeake Assessment Scenario Tool (CAST). These maps represent annual means for 2017.

Each component in Figure 8 is represented by simple coefficients; however, the technical methods of deriving the coefficients through a collaborative process can be quite complex. The CBP Partnership has used multiple models and multiple lines of evidence from scientific observations wherever possible to estimate the coefficients. For example, average loads are calculated using the average of several fully-calibrated models (Table 1). Other coefficients are borrowed directly from companion models. Land-to-water and stream-to-river factors are taken directly from USGS SPARROW simulations of the Chesapeake Bay watershed (Ator et al., 2011), while land use acres are from the CBLCM and atmospheric deposition is from CMAQ. BMP reduction factors are estimated by a collaborative expert literature review process (e.g., Berg et al 2013). River-to-bay factors are calculated from the calibrated dynamic model. Full description of the sources of information and the CBP partnership decisions are available (CBP 2020a)

Table 1: Models incorporated into the 2017 watershed model.



Model	**Use in Phase 6 Model**
CBP Phase 5.3.2 Watershed Model	Average loads Nitrogen sensitivity
USGS SPARROW regression model	Average loads Nitrogen sensitivity Land-to-water Stream delivery
USDA CEAP/APEX Chesapeake model	Average loads Nitrogen sensitivity
APLE	Phosphorus sensitivity
RUSLE	Sediment edge-of-field loads
rSAS	Lag time
UNEC	Lag time
Modflow	Lag Time

The time-averaged structure for sediment load prediction is similar to the nine components described above for nutrients, but with some important differences in source and delivery estimation. The top line of Figure 8, rather than representing edge-of-stream nutrients, now pertains to field-mobilized sediments. These sediment loads are estimated using a spatial application of RUSLE (Revised Universal Soil Loss Equation) predictions (USDA, 2013). Land-to-water factors are now conceptualized as “delivery ratios” for mobilized sediment and are implemented based on observations of yield reductions across a range of increasing watershed drainage area sizes (Cavalli et al., 2013; Roehl, 1962). Direct load sources are similar to those for nutrients with stream erosion playing a greater role. Net average annual reductions in sediment loads within streams are assumed to be relatively low based on SPARROW results (Brakebill, et al. 2010) and sediment budgets (Noe et al 2020), therefore stream-to-river factors are set such that they counteract erosion sources. Reductions to sediment loads due to reservoir sedimentation are estimated using approaches designed for the SPARROW model load estimation approach (Brakebill, et al. 2010).

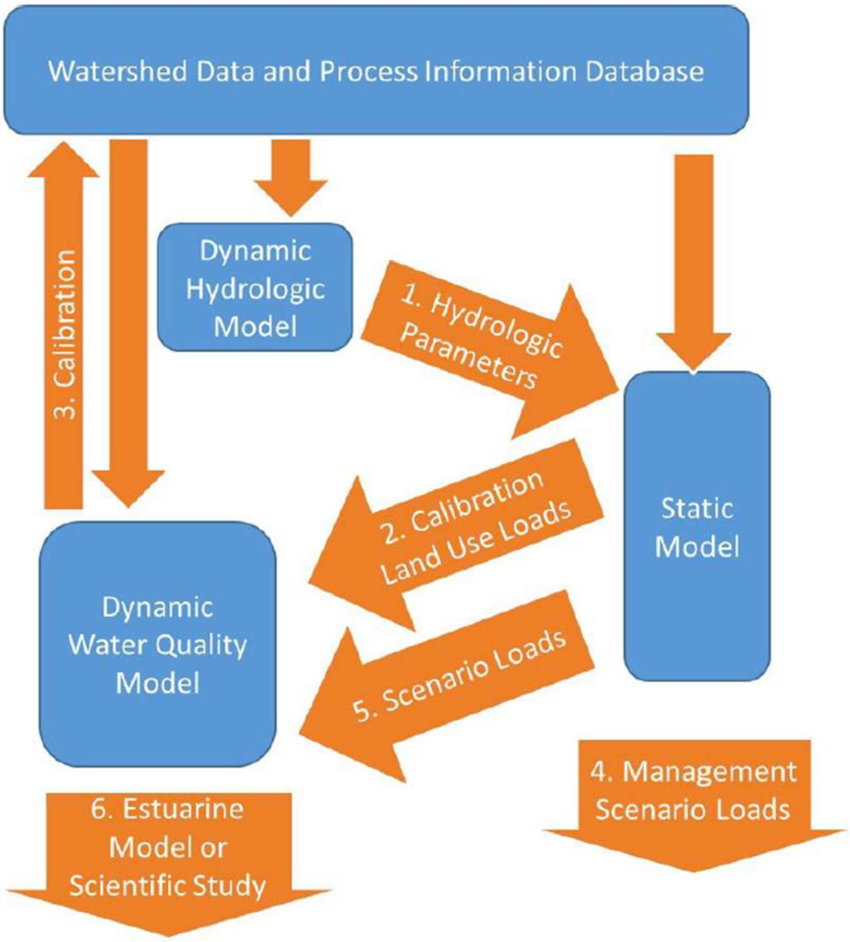


Figure 11: Relationship between the time-averaged and dynamic watershed models.

Dynamic Watershed Model

The CBP Partnership also maintains a dynamic version of the watershed model to provide daily loads to the estuarine water quality model and to estimate some parameters such as river delivery factors and stormwater runoff for use in the time-averaged model. The relationship between the time-averaged and the dynamic models are depicted in Figure 11. The dynamic model uses Hydrological Simulation Program-Fortran (HSPF, see Borah and Bera, 2004) to simulate hydrology, sediment transport, and nutrient transport in streams. HSPF simulates time-dependent hydrologic and water quality processes on land surfaces, in the subsurface, in streams, and within well-mixed impoundments. Nutrient export from the land surface and subsurface is temporally downscaled from the long-term predictions of the time-averaged model using an algorithm dependent on nutrient application timing and HSPF simulations of hydrology and sediment. The structure is documented in CBP 2020a, section 10. The simulations run for the Chesapeake Bay implementation are forced with hourly values of rainfall, snowfall, temperature, evapotranspiration, wind, solar radiation, dewpoint, and cloud cover. Input data includes land use acreage from the CBLCM and atmospheric deposition from CMAQ, as well as BMPs, fertilizer and manure applications, and point source and septic loads to calculate daily flow and associated nutrient and sediment loads.

2.3.2 Time-Averaged and Dynamic Watershed Model Advantages and Limitations

Watershed Model Advantages

The adaptable multi-model structure of the watershed model allows the ongoing leveraging of other models and analyses of monitoring data for its improvement. The flexible construction is conducive to effective adaptive management which guide better decision making, thereby improving environmental results (Easton, et al., 2017). The reduced complexity of CAST, the time-averaged version, is more understandable to the stakeholder community and has allowed for greater participation in model development. Moreover, the relatively fast run times and web interface for CAST allows users to generate their own scenarios or custom reports of previously run scenarios. Additionally, because the CAST structure is compiled from multiple sources, its use facilitates an uncertainty quantification. Finally, because of the simplified CAST structure, it is able to take advantage of spatially and temporally dense data sets for water quality measurements and daily load calculations at critical points in the watershed, including near the head of tide for major Bay tributaries.

Watershed Model Limitations

Opportunities for improvement in simulation capacity remain. The details of calculation for individual model components can be quite complex and not all of the parameters can be estimated using companion modeling approaches. As a result, no comprehensive assessment of load prediction uncertainty has been completed, particularly with respect to model quantification of load alterations due to anthropogenic changes over time (Easton, et al 2017). Demand has increased for better targeting of management practices to improve the effectiveness and lower costs of restoration, and with this, a need to develop better estimates of nutrient and sediment transport potential at a fine scale (Easton, et al., 2020). A related issue is the need to improve targeting of practices that reduce those species of nutrients and sources of sediment with greater efficacy towards water quality load reductions (Shenk, et al., 2020; Craig et al. 2008; Filoso, et al. 2015). Model upgrades will be needed to appropriately assess the effect of landscape, land use, and land management on fine-scale delivery and speciation. This is particularly true of the sediment simulation approach that requires support from measurements from small headwater streams to large rivers and delivery functions customized to varied landscape settings (Easton, et al 2017; Smith et al., 2011; Noe et al., 2020).

2.3.3 Watershed Model Summary Recommendations

2.3.3.1 General Recommendations

Development of the watershed model components of the CBP modeling system should focus on accurately predicting delivery of nutrients and sediment consistently across spatial scales and properly account for lag times in movement from watershed sources to the Bay. Special attention to scaling issues is necessary to identify water quality problems and management solutions at a site or stream segment scale, as well as cumulative impacts on the scale of a river and watershed.

The watershed models should strive to adopt agile, modular designs to facilitate investigation of varied processes and alternative algorithms, and to increase transparency for scientists working on diverse aspects of watershed hydrology as well as sediment, nitrogen, and phosphorus transport and transformation. This should include formalization of rules and procedures for linking modules across spatial and temporal scales. Modularization will provide greater flexibility and facilitate examination and testing of alternative approaches for quantification and simulation of biophysical processes at lower and higher resolutions and with different levels of mechanistic detail (Leavesley et al. 2002). Additionally, it will facilitate functional expansion of the models to simulate future issues, such as the transport and fate of contaminants of emerging concern. All data, code, output, and documentation should be made openly available on-line to enable a community modeling approach to future model development.

2.3.3.2 Watershed Hydrology Recommendations

Improved simulations of Chesapeake Bay hydrology within the dynamic watershed model would improve predictions of the effects of management actions on nutrient and sediment delivery. One area in need of improvement is representation of the hydrologic processes at sub-basin scales to better depict the spatial distribution of nutrient and sediment source problems, and to generate more finely-resolved predictions of pollutant transport within the Bay watershed. Such information would identify areas that, if managed sustainably, would be most effective to help achieve management and restoration goals (Veith et al., 2003, 2004; Easton et al., 2008, 2017; Tomer, 2018).

Another area for improvement is the development of a version of the model based on a standard watershed layer appropriate for the scale that management actions are implemented, such as the National Hydrography Dataset (NHD) or Hydrologic Unit Code (HUC). Closer matching between model and management spatial scales would allow for easier conversion and communication of model results to managers.

Over the long term, a complementary approach to the basin-wide management-scale model would be to perform high-resolution sub-basin scale dynamic simulations in a few select locations that can be used to inform the larger-scale management model. Models operating at scales commensurate with processes occurring on hillslopes and in small headwater streams and capable of resolving fine-scale locations of BMP implementation would more accurately quantify headwater inputs into higher order rivers and estuarine tributaries. These smaller-scale watershed models and the related data assembly can be developed based on regular or irregular mesh grids, or can use Hydrologic Response Unit (HRU)-based hillslope, physiographic district, or tributary scale resolution (Reger and Cleaves, 2008; Cleaves, 2003; Smith et al., 2011; Smith and Wilcock, 2015; Amin et al., 2017; Amin et al., 2018; Liu et al., 2017; Veith et al., 2019; Williams et al., 2015; Collick et al., 2015). In particular, recent developments in watershed reactive transport modeling have advanced forward to couple watershed hydrological processes and land-surface interactions with multi-component reactions to capture the dynamics of water and biogeochemical interactions, including nutrients, carbon, and sediment transport (Bao et al. 2017; Li, 2019; Zhi et al. 2019).

A good candidate dynamic watershed model that could address many of these limitations and recommendations is the Bio-Reactive Transport and Flux version of the Penn State Integrated Hydrologic Modeling System (BioRT-Flux-PIHM) (Zhi et al., 2021). In BioRT-Flux-PIHM, water flow is dictated by watershed hydrology that is influenced by meteorological conditions and other watershed characteristics. Domain discretization is fundamental to the approach and an unstructured triangular irregular network (e.g., Delaunay triangles) is generated with geometric and parametric constraints (Bhatt et al., 2014). Figure 12 shows an example of the domain decomposition of Mahantango Creek watershed into 2,606 triangular mesh elements and 509 linear stream elements. BioRT-Flux-PIHM couples flow and transport calculations within a full biogeochemical, thermodynamic, and kinetic framework (Steefel et al. 2015), thereby enabling explicit tracing of spatial and temporal evolution of geochemical species in fluid and solid phases. In particular, this code has been applied to understanding fine-scale nutrient and carbon transformation and transport processes (Wen et al., 2020; Zhi and Li, 2020). These modeling efforts, coupled with insights from data, have propelled the Shallow and Deep Hypothesis, which underscores the essential role of nutrient concentration contrast in shallow soil water and

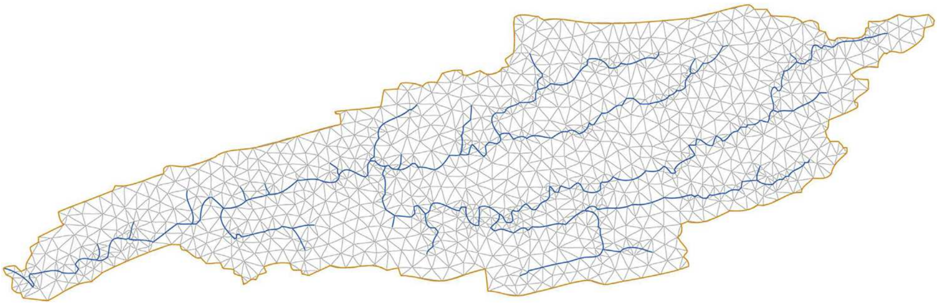


Figure 12: Domain decomposition of Mahantango Creek watershed. From Bhatt et al. (2014).

deeper groundwater in shaping stream and river concentration and discharge relationships (and therefore loads) at different flow regimes (Zhi and Li, 2020). These insights, combined river chemistry data, can be used to predict nutrient loads with simplified model structure with reduced computational cost, and to estimate nutrient removal in groundwater aquifers. The use of these reactive transport models however are not limited to nutrients and carbon. They can also be used to simulate other water quality parameters, including cations, salinity, and sediments.

2.3.3.3 Watershed Sediment Recommendations

Translating edge-of-field to edge-of-stream is difficult due to high variability across the spatial scales, watershed location, sub-regional setting, and localized land use, resulting in potentially high uncertainty in estimation (Smith and Wilcock, 2015). In addition, the balance of deposition and erosion in stream reaches is highly variable but critical to sediment budgets (Noe et al., 2020). These issues can be addressed with the creation of new empirical functions through documentation of sediment transport processes and rates for varied sediment grain size classes throughout watershed stream channel networks in varied land use settings. Better documentation is particularly important in the low-order headwater streams that traverse the steepest elevation gradients, penetrate the farthest into upland areas, and comprise over half of the total channel network length in the Bay watershed.

Time lags in sediment movement to the Bay highlight the importance of making sure that stakeholders understand that there can be temporal gaps between sediment source management investments and Bay water quality outcomes (Pizzuto, 2014). In the short-term, an updated CBP watershed model could better represent how runoff drainage networks affect sediment sources, sinks, transport, and fate. The addition of residence times of sediment located in temporary storage zones would improve predictions of sediment responses to management actions. An updated model for watershed sediment simulations would also carry over to improved simulation of constituents (e.g., nutrients) associated with sediments, and the use of sediment results as input to habitat quality and effects on living resources (e.g., SAV, oysters). Local and regional dynamics of erosion, followed by the transport, deposition, remobilization, and eventual delivery to the Bay, are often very important to local communities and stakeholders.

The spatiotemporal scales of existing empirical and rules-based models do not match process-based sediment models (Dietrich, et al. 2003). In the longer-term, the formulations related to sediment transport within the watershed model should be refined to better represent the time scales of sediment delivery and thus allow for better assessment of management practices for both the sediments and associated nutrients (Pizzuto 2014; Filoso et al. 2015; Williams et al. 2017). New science and data are needed to fill gaps in our current knowledge of watershed sediment erosion and delivery rates in prominent physiographic settings and under different land use conditions (Smith and Wilcock, 2015; Noe et al. 2020).

A CBP-sponsored 2017 legacy sediments workshop identified knowledge gaps and how they could be addressed (Miller et al. 2019a). To build on this effort, the CBP Partnership would benefit from the establishment of a sediment modeling workgroup with expertise in geomorphology and with stakeholder representation to engage in both long-term and short-term knowledge co-generation and strategies to improve the representation of sediment dynamics in the CBP watershed model (Smith et al. 2011). It is anticipated that it could take a decade or more for full development (including new data collection) and implementation (i.e., calibration, validation) of a revised sediment transport formulation in the watershed model.

2.3.3.4 Watershed Nutrient Recommendations

The time-averaged version of the nutrient watershed model should be enhanced to become more spatially explicit and mass-conserving. An enhanced version of the time-averaged model could be fit to the estimated monitored fluxes (as it is done with the SPARROW model, e.g., Ator and Garcia, 2016) and be informed by ensemble predictions of the model component coefficients in a similar manner to the 2017 time-averaged model. This new version would allow for investigation of the watershed effects of BMP performance, including localized effects and interactions with the effects of climate change (Craig et al., 2008; Filoso and Palmer, 2011). A spatially-explicit version would also allow for better quantification of nutrient sources and sinks that depend on the spatial arrangement of riparian and wetland areas (Weller and Baker, 2014), and improved representation of hydrologic connectivity to identify critical source areas that contribute disproportionately to loads that could then be targeted by management (Wallace et al., 2018). Finally, this new version would offer the opportunity to quantify nutrient legacy effects, revisit riverine biogeochemical processes - especially in active channels and floodplains - and to incorporate new sources of high-resolution data on land use and geomorphology to better represent variability in nutrient retention in forest types and forest seral stages over time and under elevated atmospheric CO_2 (Craine et al., 2018).

For both the time-averaged and dynamic watershed models, the nutrient forms and species simulated should be evaluated to ensure they can be easily matched to the forms in the estuarine biogeochemical model. For example, partitioning nutrients into particulate and dissolved phases in the watershed models would improve connectivity with the estuarine model (e.g., Dari et al., 2018). Reactivity might also be considered since effective targeting of management will require implementing practices that reduce the reactive constituents (e.g., Liu et al., 2018a; Miller et al., 2019b). Lags related to nitrogen delivery require expanded considerations of groundwater flow pathways and new approaches to quantify travel time and removal rates within drainage networks (Sanford and Pope, 2013; Phillips and Lindsey, 2003). Specification of delivery processes under varied settings and conditions will be necessary, requiring expanded forms of measurement in each of the prominent physiographic settings in the Chesapeake Bay watershed. For example, an approach that relates nutrient delivery potential to a measurement such as a topographic wetness index or connectivity index could take advantage of recent increases in land use and elevation measurement scales. Temporally and spatially dense sensor arrays in low-order streams would allow for development and validation of such approaches (Easton et al., 2020). Addition of organic carbon to the watershed models is also warranted, both to force estuarine biogeochemical models and allow more accurate representation of watershed loads of oxygen-demanding material. In the present versions of the watershed models, carbon loads are derived from the simulated nutrient and sediment loads. Implementing these recommendations will improve the watershed models' ability to identify critical source areas, especially those in hydrologically active and connected zones, and allow for smooth coupling to other models.

Implementing these recommendations will require re-analyses of existing data and collection of additional data (e.g., Ator et al., 2020). Specifically, nitrogen speciation, sources, and sinks will need to be characterized, particularly in low-order streams. Leveraging existing data will involve extensive data gathering and the development of new data analysis strategies for using relatively short time series to determine spatial and temporal variability. New data in previously unmonitored areas, small streams, and directly discharging groundwater should be collected to fill important gaps. Field-scale nutrient flux data should be collected relative to field conditions and landscape position to better identify and manage critical source areas (Buda et al., 2009; Buda, 2013; Buda et al., 2013).

2.4 Estuarine Hydrodynamics and Biogeochemistry Models

2.4.1 Overview of the 2017 Estuarine Hydrodynamics and Biogeochemistry Models

The estuarine model is composed of two independent models. A hydrodynamic model computes transport information, which is stored “offline” for repeated use by a biogeochemical model. The biogeochemical model is the decision model for projected attainment of tidal Bay dissolved oxygen, chlorophyll, and water clarity standards under TMDL scenarios.

Estuarine Hydrodynamic Model

The estuarine hydrodynamic model (Curvilinear Hydrodynamics in 3-dimensions or CH3D) is based on a model originally developed by Sheng (1986) that was modified extensively for application to the Chesapeake Bay (Johnson et al., 1991; Kim, 2013). The hydrodynamic model is forced by tides, wind, freshwater inflow, and heat exchange at the water surface. Tides are based on observations recorded near the mouth of the Bay. (NOAA Tides and Currents; https://tidesandcurrents.noaa.gov/). Wind and heat exchange are obtained from local meteorological observations. (NOAA National Center for Environmental Information https://www.ncdc.noaa.gov/cdo-web/datasets/GHCND/locations/CITY:US240002/detail). Salinity and temperature fields are prescribed on the open boundary, outside the Bay mouth, based on observations (Chesapeake Bay Program Water Quality Database https://www.chesapeakebay.net/what/downloads/cbp_water_quality_database_1984_present). Daily freshwater inflow from rivers, diffuse coastal plain surface flows, and groundwater flows are all prescribed using output from the dynamic 2017 watershed model. The CH3D model then calculates time-dependent variations in salinity, temperature, water-level elevation, velocity, and turbulent diffusivity in three dimensions with a 90-second time step.

There are up to 19 layers in the vertical dimension with a uniform layer thickness of 1.52 m, except that the top layer thickness fluctuates with sea level. The surface layer is 2.14 m thick at mean tide. Horizontally, the governing equations in the Cartesian coordinate system are recast in a boundary-fitted curvilinear coordinate system to cope with the irregular shoreline configuration and deep channel orientation. In the present Chesapeake Bay configuration, there are 11,064 surface cells and 56,920 total cells with an average grid cell dimension of 1,025 x 1,025 m (Figure 13).

Estuarine Biogeochemical Model

The velocity and diffusivity outputs from the CH3D hydrodynamic model, along with nutrient and sediment loads prescribed by the dynamic 2017 watershed model, are used to force a finite-volume biogeochemical model (Corps of Engineers Integrated Compartment Water Quality Model, abbreviated as CE-QUAL-ICM or simply ICM; Cerco and Cole, 1993; Cerco and Noel, 2013). The ICM Model uses the same grid as CH3D and is forced with hourly transport from CH3D, daily loads from the watershed model, and monthly boundary concentrations of all state variables at the mouth of the Bay. The time step is determined dynamically based on computational stability requirements and is 15 minutes on average. ICM incorporates 24 state variables that include physical variables (salinity, temperature, fixed solids) three groups of phytoplankton, dissolved oxygen, and multiple forms of carbon, nitrogen, and phosphorus (Figure 14).

Figure 13: CH3D hydrodynamic model grid.

Salinity is computed by solving the three-dimensional mass conservation equation for a conservative substance. Computation of temperature, however, includes atmospheric heat exchange at the water surface, evaluated following Edinger et al. (1974). Salinity and temperature are computed in the biogeochemical model to provide quality assurance of the linkage to the hydrodynamic model. When forced by the same boundary conditions and surface heat flux, salinity and temperature computed in the biogeochemical model should be identical to the hydrodynamic model (Dortch et al., 1992).

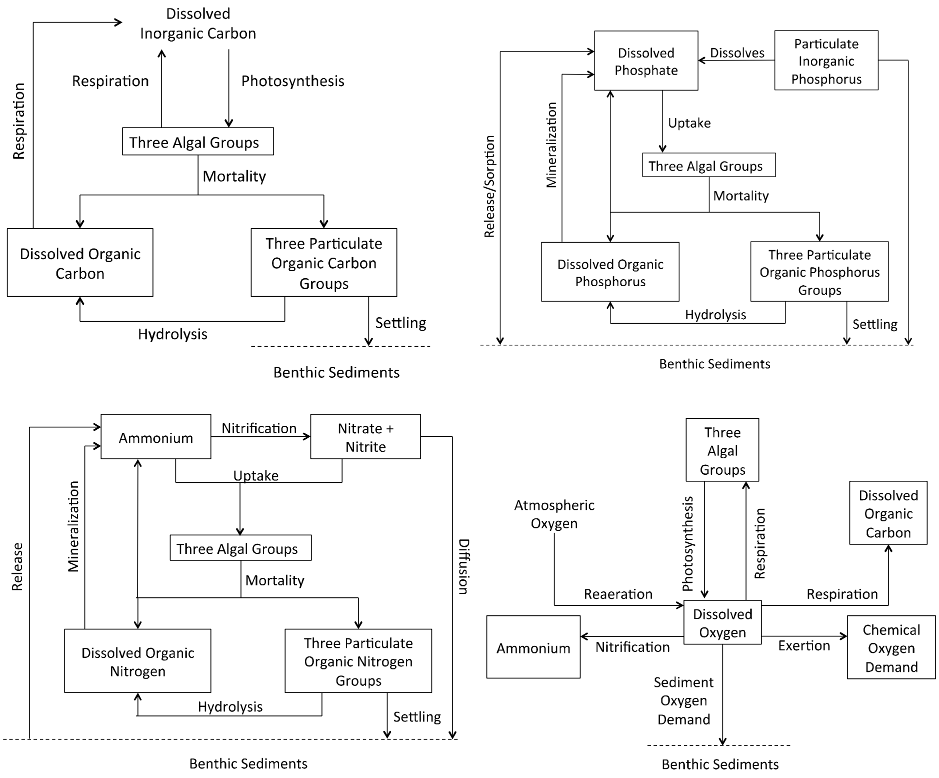


Figure 14: Schematic diagrams of the C, P, N and O_2 cycles in CE-QUAL-ICM.

Organic carbon undergoes numerous transformations in the water column. The model carbon cycle (Figure 14) is defined in this context around the process of eutrophication (Nixon 1995) and consists of the following elements: phytoplankton production and excretion; predation on phytoplankton; dissolution of particulate carbon; heterotrophic respiration;

and settling (Cerco, 2000). Algal production is the primary autochthonous organic carbon source to the water column (Cerco and Noel, 2004), although carbon also enters the system through external loading (Brookfield et al., 2021). Predation on algae releases particulate and dissolved organic carbon to the water column. A fraction of the particulate organic carbon undergoes first-order dissolution to dissolved organic carbon. Dissolved organic carbon produced by excretion, predation, and dissolution is respired at a first-order rate to inorganic carbon. Particulate organic carbon that does not undergo dissolution settles to the bottom sediments.

The model nitrogen cycle (Figure 14) includes the following processes: algal uptake and metabolism; predation; hydrolysis of particulate organic nitrogen; mineralization of dissolved organic nitrogen; settling; and nitrification. External loads provide the ultimate source of nitrogen to the system. Available nitrogen is incorporated by algae during growth and released as ammonium and organic nitrogen through respiration and predation. A portion of the particulate organic nitrogen hydrolyzes to dissolved organic nitrogen. The balance settles to the sediments. Dissolved organic nitrogen is mineralized to ammonium. In an oxygenated water column, a fraction of the ammonium is subsequently oxidized to nitrate+nitrite through nitrification. Particulate nitrogen which settles to the sediments is mineralized and recycled to the water column, primarily as ammonium. Nitrate and nitrite move in both directions across the sediment-water interface, depending on relative concentrations in the water column and sediment porewater.

The model phosphorus cycle (Figure 14) includes the following processes: algal uptake and metabolism; predation; hydrolysis of particulate organic phosphorus; mineralization of dissolved organic phosphorus; dissolution of particulate inorganic phosphorus; and settling and resuspension. External loads provide the ultimate source of phosphorus to the system. Dissolved phosphate is incorporated by algae during growth and released as phosphate and organic phosphorus through respiration and predation. Dissolved organic phosphorus is mineralized to phosphate. A portion of the particulate organic phosphorus hydrolyzes to dissolved organic phosphorus. The balance settles to the sediments. Dissolution of particulate inorganic phosphorus is also possible. Within the sediments, particulate organic phosphorus is mineralized and recycled to the water column as dissolved phosphate.

In the model carbon, nitrogen, and phosphorus cycles, three classes of particulate constituents are considered: labile, refractory, and slow refractory, corresponding to classes utilized in the benthic sediment diagenesis model. ICM includes a benthic diagenesis submodel for calculating sediment oxygen demand and sediment-water nutrient flux, and a sediment transport submodel for calculating sediment loading, deposition, erosion, and transport, which considers four solids size classes (Cerco et al., 2010; Cerco and Noel, 2013). Bottom shear stress, for computing erosion and deposition, is a combination of stress generated by currents and surface waves (Harris et al., 2013). A multi-level bed framework tracks the distribution of each size class in each layer and stores bulk properties including layer thickness, porosity, and mass (Warner et al., 2008b). An SAV model calculates the water clarity/SAV standard for the restoration of SAV and accounts for positive feedbacks that improve water clarity (Cerco and Moore, 2001). A model of benthic filter feeders (three filter feeder groups) accounts for the effects of filtration on water quality and clarity.

The sediment diagenesis submodel (Figure 15), based on DiToro (2001) (see also Brady et al., 2013; and Clark et al., 2017), is coupled to ICM to account for the response of sediment-water nutrient and oxygen exchanges to management actions in the watershed. The spatial and computational time scales of the sediment diagenesis model are the same as the water quality model. The diagenesis model considers a 10 cm thick active sediment layer that incorporates an aerobic surface layer with the remaining depth considered anaerobic. The thickness of the surface aerobic layer is time variable and is calculated based on overlying water column oxygen concentration and model calculated sediment oxygen demand. The aerobic first layer is much thinner than the anoxic second layer (1-2 mm vs. 10 cm). In the anoxic layer, diagenesis of organic matter (nitrogen, phosphorus, and carbon) creates ammonium, phosphate, and oxygen demand, respectively. The fate of these substances (i.e., release to water column, release to atmosphere, burial) is determined by processes including nitrification, denitrification, sulfate reduction and methanogenesis. Ten years of model spin-up are required to equilibrate new scenario loads with burial and refractory diagenetic processes.

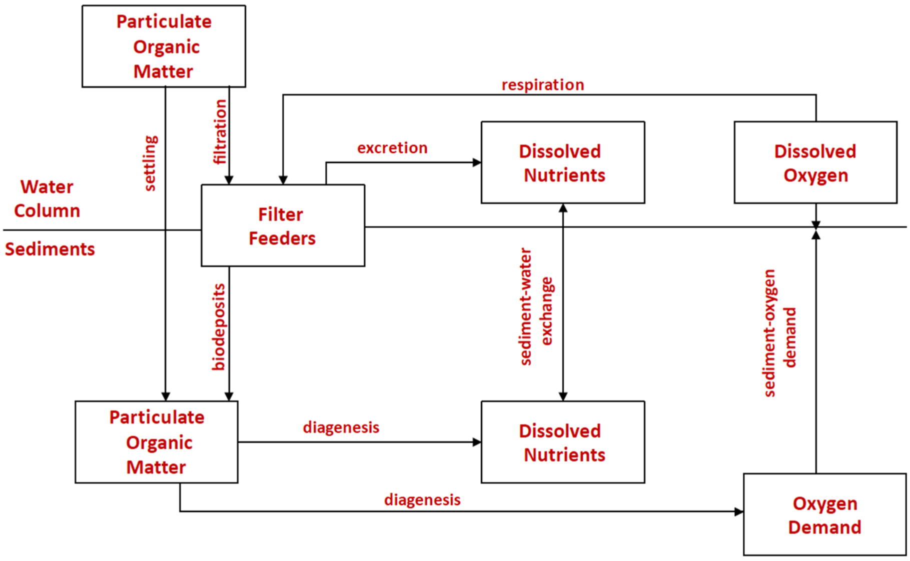


Figure 15: Schematic diagram of the sediment diagenesis submodel.

2.4.2 Estuarine Model Advantages and Limitations

Estuarine Model Advantages

The CH3D/ICM combination provides computations of estuarine transport processes and water quality in three dimensions on spatial scales of meters (vertical) to kilometers (horizontal and lateral) and on an intra-tidal time scale. The grid is based on quadrilateral elements in the horizontal-lateral plane. CH3D is distinctive, however, in the use of “non-orthogonal curvilinear coordinates.” The non-orthogonal representation implies that the grid elements are not required to meet at right angles as in models which employ orthogonal curvilinear coordinates. The non-orthogonal coordinate system allows improved representation of complicated channel geometry and irregular shorelines. The computational grid employs a Z-grid representation in the vertical axis. In the Z-grid, variations in depth are represented by varying the number of cells in the vertical direction. The cells are of constant thickness except for the surface cell which varies according to meteorological and tidal forcing. The Z-grid avoids the artificial vertical mixing which is associated with sigma coordinate systems (constant number of vertical cells, which vary in thickness, throughout). The artificial mixing associated with a sigma grid was noted early in the model application and was avoided to compute bottom-water anoxia in the Bay channel.

Perhaps the greatest advantage of the eutrophication component is the use of organic carbon throughout the model kinetics representations. Traditional water quality models were often based on alternate quantities such as biochemical oxygen demand (BOD) or “organic matter.” Phytoplankton was quantified as chlorophyll rather than as carbon biomass as in the present model. The carbon-based kinetics maximize the use of current, rigorous observations in the model calibration and verification and avoid the need to define quantities such as BOD-to-chlorophyll ratio. One rationale for the use of organic carbon is to make the water-column kinetics consistent with the carbon-based sediment diagenesis model. The model is also distinctive in that the “labile,” “refractory,” and “slow refractory” carbon, nitrogen, and phosphorus variables in the diagenesis model have direct corresponding state variables in the water column. The definition of direct corresponding state variables avoids the need to define empirical relationships between detailed representation in the sediments (i.e., three reaction classes) and less detailed representation in the water column (e.g., total organic carbon).

The phytoplankton kinetics in the model (Cerco and Noel, 2004) are distinctive in that they employ, to the greatest extent possible, quantities currently measured in field and laboratory investigations. Growth is related to maximum photosynthetic rate (g C g^{-1} chl d^{-1}) rather than a specified daily-average growth rate. Production is related to light via the Jassby and Platt (1976) relationship and is based on photosynthetically active radiation (μmole photons m^{-2} s^{-1}) rather than thermal units such as langleys. The model has been rigorously calibrated to observed photosynthetic rates and primary production (Cerco and Noel, 2004).

Estuarine Model Limitations

As the modeling effort developed, and additional capabilities were added to the CH3D/ICM combination, some disadvantages of the grid configuration became apparent. The grid went through several refinements which improved resolution in the horizontal-lateral plane. Inevitably, the limitations of representing complicated shoreline configuration with quadrilaterals have emerged. An unstructured grid that employs triangular elements could be a better approach.

The Z-grid represents changes in depth by varying the number of cells in the vertical. When changes in depth are steep, the variation in number of cells can become dramatic, resembling a “stairstep” or even a “wall.” This can create problems. For example, computing the turbidity maximum at the head of the bay, using the sediment transport module, is impeded by the sharp variation in number of cells at the head of the Bay channel. Sediment moving upstream cannot climb the “stairs” at the head of the channel. A smoothly sloping bottom would improve computation of upstream sediment transport although care must be taken not to adversely affect the representation of stratification associated with a sigma grid. Another problem arises when using the model in the shallow upper-reaches of tributaries. In such regions the model may have only one or two depth levels and hence be unable to reproduce the estuarine circulation required to effectively model salinity in the shallows.

In addition, the outer boundary of the physical model is currently located at the Bay mouth, a region of sharp changes in topography and strong currents. This is not ideal. Moving the outer boundary offshore to the continental shelf, away from the mainstem Bay, would, among other things, improve simulations of future impacts of sea level rise on Chesapeake Bay. Finally, the lack of coastal wetting and drying in the current model does not allow for consideration of impacts from sea level rise inundation of the coastline and its wetlands.

The eutrophication component incorporates representation of several “living resource” components including submerged aquatic vegetation (SAV) and bivalve filter-feeding organisms. Living resources are included based on their value to management or their necessity to the model. For example, correct representation of the spatial distribution of phytoplankton is impossible without incorporating the effects of filter feeders. The living resource components are based on mass-balance relationships. A disadvantage is that the complete, detailed, life cycles of the living resources are not represented. The simplification of the life cycles compromises the model's ability to represent the spatial and temporal distribution of the resources. For example, the distribution of SAV is largely based on light availability. While the influence of light on SAV distribution is well-established, the distribution of SAV is also influenced by recruitment and propagation, which are not considered in the model. The distribution of bivalves is also strongly influenced by recruitment. Since living resources are not the primary focus of the model, the additional calibration and computational resources required for more realistic representations may not be necessary. However, the limitations of the current representations must be recognized. Creation of specific, dedicated living resource models may be a superior alternative to adding complexities to the present models.

2.4.3 Estuarine Model Summary Recommendations

Short-term and long-term efforts should continue the present trend of resolving finer spatial scales to make the estuarine models more directly applicable to assessing the performance of management actions at scales relevant to local communities and stakeholders. Prediction of locally relevant restoration outcomes may also prove a powerful incentive to motivate further investment and implementation. Application to smaller scales requires that the models have sufficient resolution to resolve tidal tributaries and relatively fast changes in biogeochemistry, such as diel-cycling hypoxia (Tyler et al. 2009). However, refining spatial scale and increasing parameters have costs in computational time, development effort, data requirements, and parameter uncertainty. Some regions of the Chesapeake Bay may not benefit from further increases in spatial and temporal resolution and so careful consideration should be given to determining exactly where higher resolution is needed.

Multiple approaches for improving representation at local scales are available for consideration, including unstructured or nested horizontal grids (the term unstructured refers to grids composed of triangles, tetrahedra or irregularly shaped elements). An unstructured/hybrid grid (with hybrid referring to grids that combine multiple types of vertical and/or horizontal grid structures) would be a good candidate approach because it allows for inclusion of local-scale processes while maintaining efficient use of computational resources. SCHISM (Semi-implicit Cross-scale Hydroscience Integrated System Model; Zhang et al., 2015; 2016; Ye et al. 2018) is an example of an unstructured model (Figure 16) that provides advantages over the current CBP hydrodynamic model (CH3D). SCHISM is an open-source community-supported modeling system based on hybrid triangular-quadrangular unstructured grids, designed for seamless simulation of 3D baroclinic circulation across creek-lake-river-estuary-shelf-ocean scales. It uses a highly efficient and accurate semi-implicit, hybrid finite-element/finite-volume method with a Eulerian-

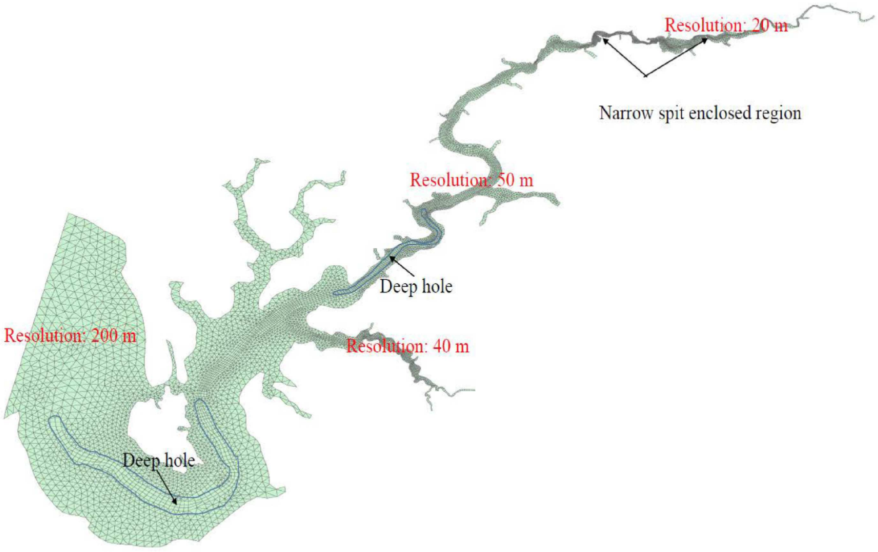


Figure 16: Example of a estuarine hydrodynamic model with an unstructured grid: Semi-implicit Cross-scale Hydroscience Integrated System Model (SCHISM) implemented in Chester River in Chesapeake Bay. From Ye et al. (2018).

Lagrangian algorithm to solve the Navier-Stokes equations (in hydrostatic form) to address a wide range of physical and biogeochemical processes. The number of vertical layers can also be varied spatially (Zhang et al. 2015), and thus a single SCHISM grid can seamlessly morph between 1D/2D/3D configurations (‘model polymorphism’; Zhang et al. 2016). The use of “shaved” cells (i.e., cells that have a sloped bottom and avoid the staircase effects associated with Z coordinates) near the bottom not only captures the underlying bathymetry/topography, but also greatly improves model accuracy for bottom-controlled processes such as salt intrusion and gravity overflow (Ye et al. 2018).

In addition, two-way “online” coupling (rather than one-way “offline”) between the hydrodynamic and biogeochemical models is critical for enabling investigation of how biogeochemical and biological processes affect physical processes. For example, increased particulates in the water column will impact estuarine bottom temperature via light attenuation (Kim et al., 2020), and SAV can impact water velocities. Such feedbacks of the biogeochemistry on the physical fields may be important in setting local TMDLs.

A second recommendation (also longer-term) is to implement a modular, experimental simulation framework that allows for testing of new and alternative biogeochemical formulations. This would allow investigation of additional processes and alternative formulations to increase certainty in the results provided to management based on the foundational version of the coupled modeling system. Such a framework or testbed approach would also expand the engagement of the CBP modeling system with academic and government research communities which would facilitate incorporation of latest scientific advancements. Such inter-model comparison approaches have been successfully performed that include the

CBP estuarine model (Irby et al. 2016; Irby and Friedrichs 2019) and in a comparison of shallow water models in the Chester River of the Bay (Friedrichs et al., unpublished). These collaborative groups of modeling teams were more effective than individual efforts in advancing the models in large part because of the balance between the teams working separately while also meeting on a regular basis to share their findings and insights. A similar collaborative approach is recommended for the next generation model of the Chesapeake Bay estuary.

Finally, the 2014 Agreement has focused attention on the prediction of management actions on living resources and thus it will be important to put more formal effort into the identification of specific types of products from the hydrodynamic and biogeochemical models that can inform living resource models. The CBP Partnership will benefit from the enhancement of the working relationship between hydrodynamic-biogeochemical and living resource modeling groups because of the ecological, economic, and societal relevance of linking the Chesapeake Bay TMDL to living resources.

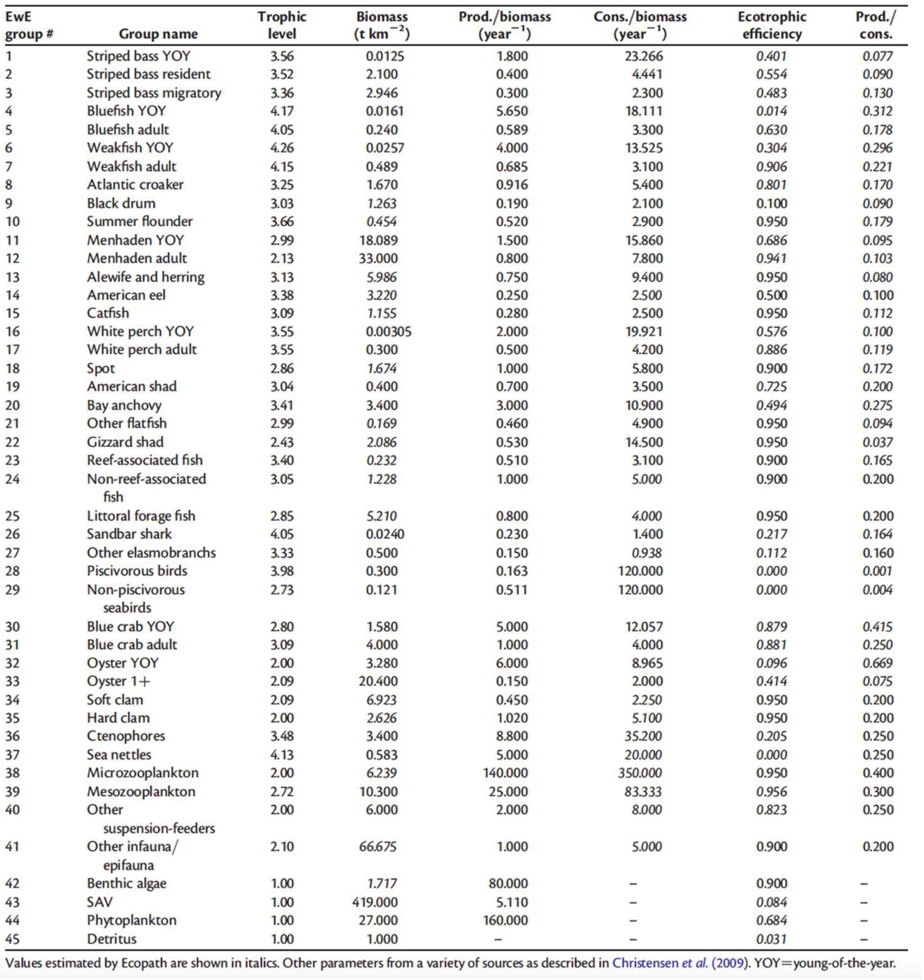
2.5 Estuarine Living Resource Models

2.5.1 Overview of the 2017 Estuarine Living Resource Models

As discussed above, the CBP modeling system includes simple sub-models for SAV and benthic filter feeders that are coupled to the estuarine hydrodynamics and biogeochemistry models, designed with water quality effects in mind. Two additional living resource models, Chesapeake Bay Fisheries Ecosystem management Model (CBFEM) and Chesapeake Bay Atlantis Model (CAM), have been developed by the NCBO to support Chesapeake Bay restoration, but these are not official components of the CBP coupled modeling system.

The CBFEM is an implementation of Ecopath with Ecosim (Christensen et al., 2009). It uses the biomass estimations of 45 trophic groups representing fisheries species of interest to the Bay, and their prey and predators (Table 2) to create a mass-balanced snapshot of the trophic linkages in the Bay as it may have been in 1950 (Townsend, 2014). The 45 trophic groups include species of commercial and ecological importance, represented by either single stocks, sub-stocks, or species groups that occupy similar foraging niches. As is typical for Ecopath with Ecosim applications, the Ecopath snapshot provides the base model for time-dependent Ecosim simulations. The CBFEM Ecosim model simulates the annual mean biomass values of the aforementioned species and groups for 53-years (1950–2002) to provide an assessment of the recent decadal dynamics of the Bay's fish species (Townsend, 2014).

Table 2: Basic parameters for the Chesapeake Bay Fisheries Ecosystem management Model (CBFEM). From Townsend (2014).



EwE group #	Group name	Trophic level	Biomass (t km^{-2})	Prod./biomass (year^{-1})	Cons./biomass (year^{-1})	Ecotrophic efficiency	Prod./cons.
1	Striped bass YOY	3.56	0.0125	1.800	23.266	*0.401*	*0.077*
2	Striped bass resident	3.52	2.100	0.400	4.441	*0.554*	*0.090*
3	Striped bass migratory	3.36	2.946	0.300	2.300	*0.483*	*0.130*
4	Bluefish YOY	4.17	0.0161	5.650	18.111	*0.014*	*0.312*
5	Bluefish adult	4.05	0.240	0.589	3.300	*0.630*	*0.178*
6	Weakfish YOY	4.26	0.0257	4.000	13.525	*0.304*	*0.296*
7	Weakfish adult	4.15	0.489	0.685	3.100	*0.906*	*0.221*
8	Atlantic croaker	3.25	1.670	0.916	5.400	*0.801*	*0.170*
9	Black drum	3.03	*1.263*	0.190	2.100	0.100	*0.090*
10	Summer flounder	3.66	*0.454*	0.520	2.900	0.950	*0.179*
11	Menhaden YOY	2.99	18.089	1.500	15.860	*0.686*	*0.095*
12	Menhaden adult	2.13	33.000	0.800	7.800	*0.941*	*0.103*
13	Alewife and herring	3.13	*5.986*	0.750	9.400	0.950	*0.080*
14	American eel	3.38	*3.220*	0.250	*2.500*	0.500	0.100
15	Catfish	3.09	*1.155*	0.280	2.500	0.950	*0.112*
16	White perch YOY	3.55	0.00305	2.000	19.921	*0.576*	*0.100*
17	White perch adult	3.55	0.300	0.500	4.200	*0.886*	*0.119*
18	Spot	2.86	*1.674*	1.000	5.800	0.900	*0.172*
19	American shad	3.04	0.400	0.700	3.500	*0.725*	*0.200*
20	Bay anchovy	3.41	3.400	3.000	10.900	*0.494*	*0.275*
21	Other flatfish	2.99	*0.169*	0.460	4.900	0.950	*0.094*
22	Gizzard shad	2.43	*2.086*	0.530	14.500	0.950	*0.037*
23	Reef-associated fish	3.40	*0.232*	0.510	3.100	0.900	*0.165*
24	Non-reef-associated fish	3.05	*1.228*	1.000	*5.000*	0.900	0.200
25	Littoral forage fish	2.85	*5.210*	0.800	*4.000*	0.950	0.200
26	Sandbar shark	4.05	0.0240	0.230	1.400	*0.217*	*0.164*
27	Other elasmobranchs	3.33	0.500	0.150	*0.938*	*0.112*	0.160
28	Piscivorous birds	3.98	0.300	0.163	120.000	*0.000*	*0.001*
29	Non-piscivorous seabirds	2.73	0.121	0.511	120.000	*0.000*	*0.004*
30	Blue crab YOY	2.80	1.580	5.000	12.057	*0.879*	*0.415*
31	Blue crab adult	3.09	4.000	1.000	4.000	*0.881*	*0.250*
32	Oyster YOY	2.00	3.280	6.000	8.965	*0.096*	*0.669*
33	Oyster 1+	2.09	20.400	0.150	2.000	*0.414*	*0.075*
34	Soft clam	2.09	*6.923*	0.450	*2.250*	0.950	0.200
35	Hard clam	2.00	*2.626*	1.020	*5.100*	0.950	0.200
36	Ctenophores	3.48	3.400	8.800	*35.200*	*0.205*	0.250
37	Sea nettles	4.13	0.583	5.000	*20.000*	*0.000*	0.250
38	Microzooplankton	2.00	*6.239*	140.000	*350.000*	0.950	0.400
39	Mesozooplankton	2.72	10.300	25.000	*83.333*	*0.956*	0.300
40	Other suspension-feeders	2.00	6.000	2.000	*8.000*	*0.823*	0.250
41	Other infauna/ epifauna	2.10	*66.675*	1.000	*5.000*	0.900	0.200
42	Benthic algae	1.00	*1.717*	80.000	–	0.900	–
43	SAV	1.00	419.000	5.110	–	*0.084*	–
44	Phytoplankton	1.00	27.000	160.000	–	*0.684*	–
45	Detritus	1.00	1.000	–	–	*0.031*	–

Values estimated by Ecopath are shown in italics. Other parameters from a variety of sources as described in Christensen *et al.* (2009). YOY=young-of-the-year.

The CBFEM Ecosim simulations have been loosely coupled to the CBP water quality model (ICM) by forcing it with time-dependent chlorophyll *a* (Townsend 2014) and SAV (Ma et al., 2010) output to assess how water quality management strategies affect living resources. For the chlorophyll *a* application, the model was used to simulate the impacts of a 40% reduction of nutrient inputs on upper-trophic-level species (e.g., the biomass of striped bass and blue crabs) and other commercially-important fished species (e.g., Atlantic menhaden, and Eastern oysters; Townsend, 2014). These simulations allow connections to be made between water quality and commercially and recreationally important species, and they can be used to assess trade-offs between water quality management goals and fisheries management goals.

The Chesapeake Bay Atlantis Model (CAM) is, in contrast to CBFEM, a spatially explicit (three-dimensional), full system (biogeochemical, physical and trophic) simulation model (Ihde et al., 2016; Ihde and Townsend, 2017). The CAM domain is composed of 97 irregular polygons and includes the brackish waters and sediments of the mainstem Chesapeake Bay and eight of its largest tributaries (Figure 17). Water movement in CAM is driven by the Navy Coastal Ocean Model (NCOM) Relocatable Model. Nutrient and sediment loads to the model are derived from the CBP dynamic watershed Phase 5.3.2 model (Shenk and Linker, 2013).

CAM includes 26 invertebrate functional groups, including primary producers and multiple bacterial groups, and 29 vertebrate groups. Most invertebrates are modeled as single state variables (mg N m^{-3}), but two invertebrate groups, blue crab and brief squid, are modeled as linked juvenile and adult state variables. All vertebrate groups are divided into 10 age classes, each tracked by abundance and weight-at-age. CAM uses

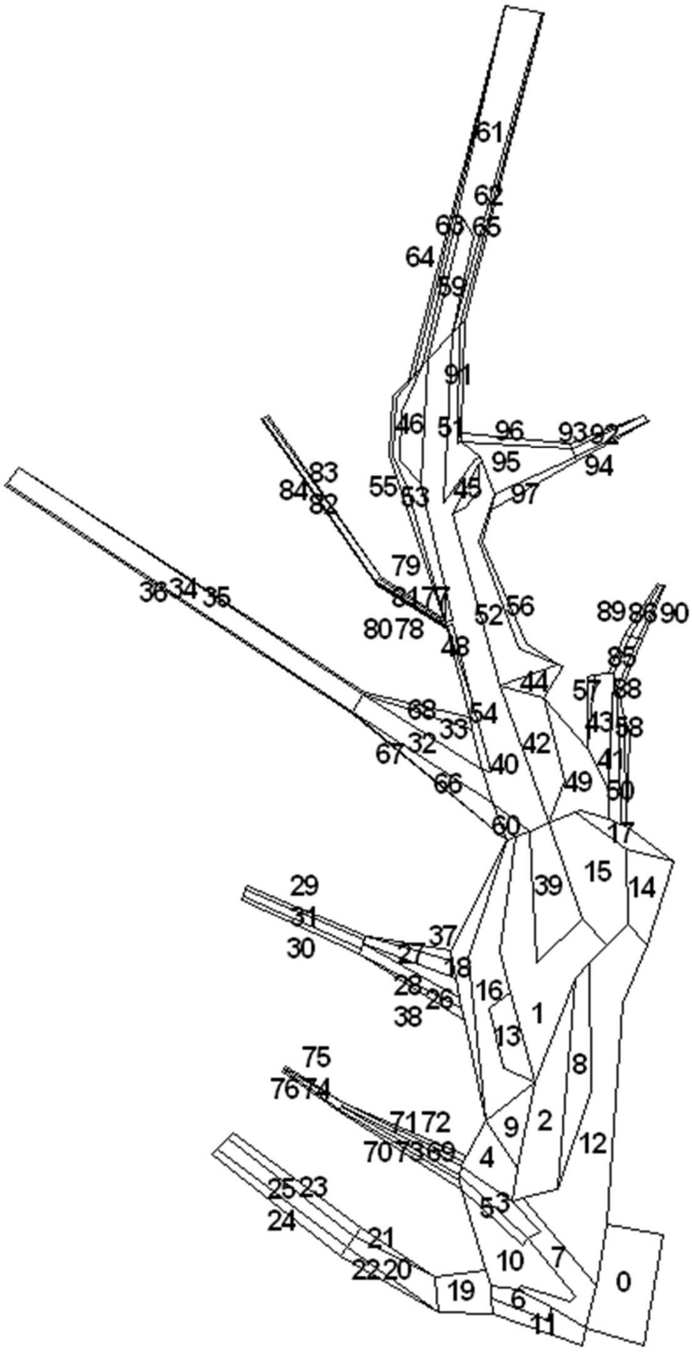


Figure 17: Spatial structure of the Chesapeake Bay Atlantis Model (CAM). The model consists of 97 irregular polygons determined by salinity, depth, bottom type (mainstem only) and management boundaries.

nitrogen as the currency for all state variables. Metabolic waste and decaying organisms form multiple forms of detritus that are cycled through bacteria to provide nutrients for both planktonic and benthic floral growth. Habitat types in CAM include both static physical factors such as mud, sand, rock, and woody debris, and dynamic biogenic functional groups such as marsh, SAV, and oyster reef, that provide refuge for prey from predator groups. Fish and other animal groups are assigned a “dependence” to one or more of the seven habitat types, and at least one such habitat must be available in a box for biomass of a group to move into that box.

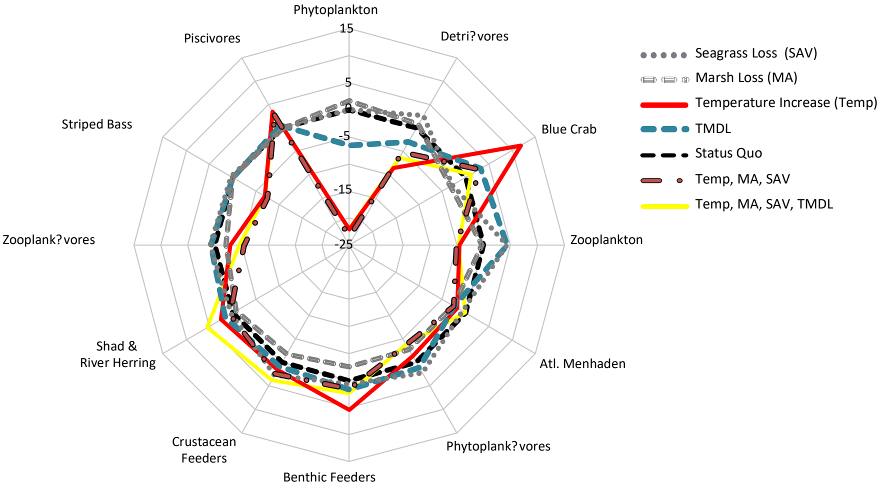


Figure 18: Effect (percent change from Status Quo scenario) of individual stressors (Submerged Aquatic Vegetation [SAV] loss [50%]; Marsh grass [MA] loss [50%]; Temperature increase [Temp; +1.5°C]; nutrient and sediment management, or “Total Maximum Daily Load” [TMDL] requirements), and combinations of those stressors, on selected CAM groups (axes) of ecological and management interest. Each scenario simulation was run for 50 years. Details of each scenario can be found in Ihde and Townsend (2017). Figure and caption modified from Ihde and Townsend (2017).

CAM has been used to estimate the higher trophic level impacts of fully achieving the goals of the U.S. EPA TMDL requirements under present day climate conditions and warmer water temperatures, habitat loss, and water quality restoration (TMDL) under assumed future climate conditions (Figure 18; Ihde et al., 2016, Ihde and Townsend 2017). These simulations used nutrient and sediment loads derived from the CBP’s Phase 5.3.2

Watershed Model (Figure 19; Ihde et al., 2016). The CAM, as well as the CBFEM, has not been directly coupled to the CBP estuarine hydrodynamic and biogeochemical models. Indeed, the CAM overlaps the functionality of the ICM because Atlantis is built on its own biogeochemical model (Murray and Parslow, 1999).

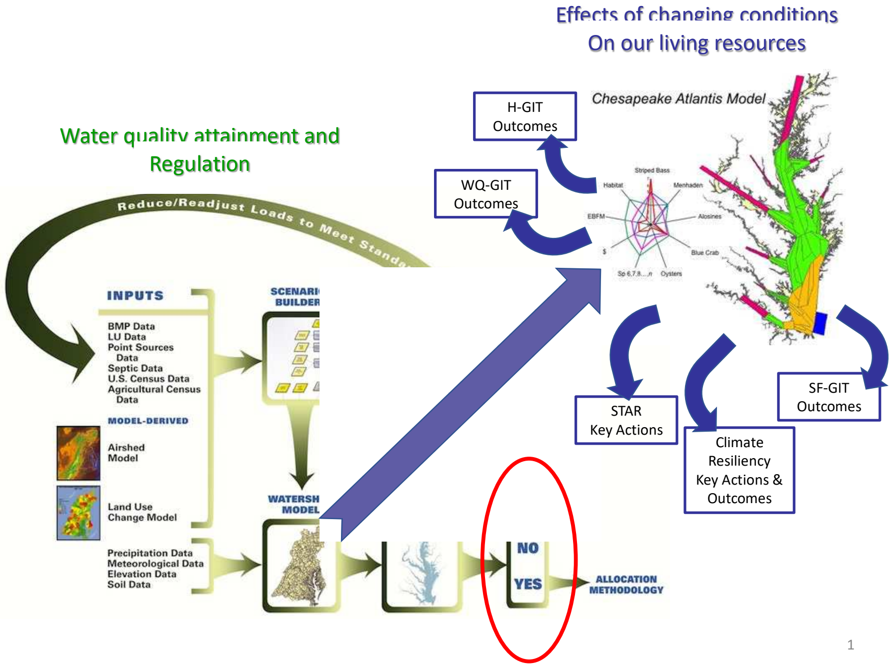


Figure 19: Relationship between the Chesapeake Bay Partnership models and the Chesapeake Bay Atlantis Model (CAM) showing how the watershed model can be used to force CAM to examine effects of restoratioin and changing conditions on living resources.

2.5.2 Estuarine Living Resource Models Advantages and Limitations

Estuarine living resource models are needed to estimate ecosystem status and predict the impacts of anthropogenically induced changes to forcing conditions on higher trophic level species, most notably the TMDL mandated nutrient loads and altered climate. Unlike more targeted living resource models that simulate individual species or just a few species, these ecosystem models put the population dynamics of modeled groups in context of the entire ecosystem with predators, prey, and competitors, providing a mechanism for achieving the larger goal of ecosystem-based management.

Simpler approaches, like CBFEM, mainly focus on trophic factors, whereas more complex approaches, like CAM also include the dominant physical and biogeochemical forcings. The CBFEM produces a mass-balanced state of the ecosystem, which is then used to simulate the system over time in response management actions. In contrast, CAM is designed to estimate cumulative effects of multiple factors acting simultaneously on the system. The structure of CAM is spatially explicit, and it is much more complex than CBFEM. As a result, the CAM model is computationally expensive and simulations can take days. In comparison, Ecopath with Ecosim produces model estimates in seconds to minutes.

Because the CBFEM and CAM models integrate a variety of different data from an array of sources, model estimates carry the burden of uncertainty inherent in each of those sources. As a result, the uncertainty of the model outputs is very large. Thus, the CBFEM and CAM models are not, at present, applied to tactical tasks like setting fishery harvest limits. Instead, they could be used to supplement tactical models, providing contextual information such as potential ecosystem impacts and trade-offs for a range of different management options to refine decision making.

Finally, it should be noted that the CBFEM and CAM model simulations are constrained to reflect the characteristics of the observed system. Yet, ultimately, these models may be needed to predict the future on timescales of multiple decades or more, in response to future forcing conditions that have not been observed in the past, thus adding even more uncertainty to the model predictions.

2.5.3 Estuarine Living Resource Models Summary Recommendations

Efforts to further incorporate living resources into the CBP modeling system predictions could start with the development of additional models, like CBFEM and CAM, that use CBP model output to drive higher trophic level models. In the short term, outputs can be used to estimate responses to habitat changes of key species in response to management actions. The current SAV model in ICM responds to water clarity; however, other factors also influence SAV growth such as propagation processes and the physical characteristics of the bottom substrate. These additional influences could be added to the ICM model to more realistically simulate SAV growth and bed expansion in response to restoration efforts and improved water clarity in the Bay. Similar expansions to the existing formulation can be implemented for benthic filter feeders within the ICM (Newell et al. 2002; Harding and Mann 2001; North et al. 2010) to increase their realism and include more feedbacks. This approach has been used for fish population dynamics in Chesapeake Bay for both menhaden (Dalyander and Cerco 2010) and bay anchovy (Adamack et al. 2017). Both efforts used an agent-based (Lagrangian) approach to simulate population dynamics, but Eulerian-based approaches could also be implemented.

Another short-term approach is to process the output of the estuarine models to assess habitat suitability of key species, for example, the effect of temperature and oxygen on striped bass (Figure 20). Habitat suitability is widely used to inform management on how altered environmental conditions will affect habitat quantity and quality (e.g., Secor 2009; Brown et al., 2013). A new generation of these habitat models, such as species distribution models, niche modeling, and bioclimatic models, are now being widely applied (Guisan and Thuiller, 2005; Ehrlén, and Morris 2015; Crear et al., 2020a,b). Models of higher trophic level habitat that use estuarine model output could be modular so that they can be easily interchangeable and allow for direct comparisons of responses across species.

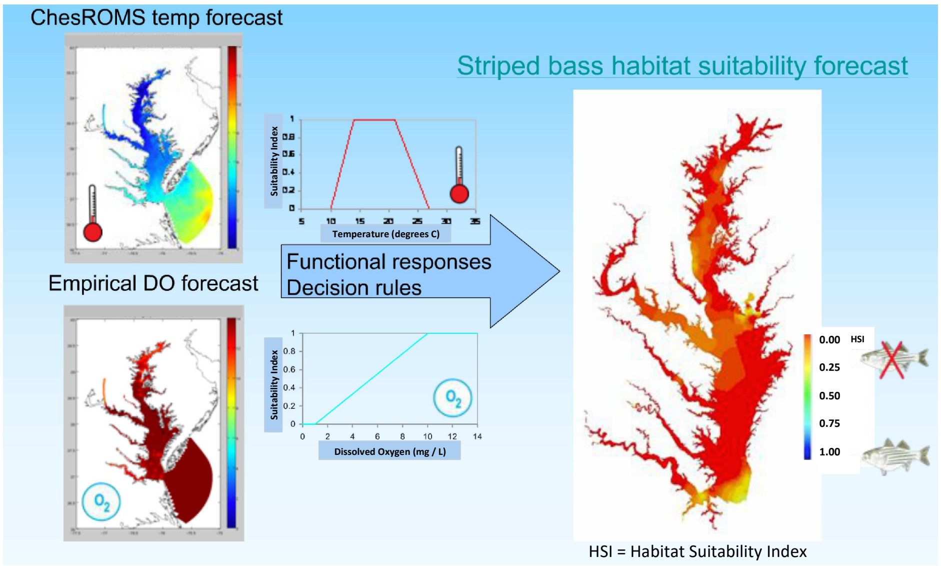


Figure 20: Schematic diagram of a striped bass habitat suitability moded. This is an example of a "secondary model" that can use output from the CBP partnership models (e.g., water quality parameters) to define habitat quality and /or impacts on higher trophic level organisms.

Further adapting and integrating one or both of the existing food web models with the estuarine models is another relatively short-term approach. The feasibility of trying to extract the higher trophic level formulations from CAM and loosely or tightly couple them to CH3D and ICM is an open question. One role that may be important for CAM is to function as a companion model to ICM to address many of the goals and outcomes of the 2014 Agreement that relate to higher trophic level species that are not simulated by ICM. In addition, or alternatively, the CBFEM could be developed further into a spatially-explicit Ecospace model, which would allow for loose coupling to CH3D-ICM, as Ecospace does not contain its own physical or biogeochemical model. Making use of the habitat capacity model within Ecospace (Christensen et al., 2014) would allow for using CH3D and ICM output as environmental drivers affecting the biomass and spatial distribution of estuarine living resources.

Comparing an Ecospace version of CBFEM with CAM would then provide a multiple model approach for higher trophic levels (Lewis et al. 2021). Issues related to commercial and recreational fish abundances, food web energetics (e.g., pelagic versus benthic pathways), system resilience, and human interactions could be quantified to allow for comprehensive assessment of the costs, benefits, and tradeoffs of different management strategies (e.g., Wainger et al. 2013).

A technical challenge to the incorporation of living resources within the CBP modeling system is how to couple the models to the watershed and estuarine models (Ganju et al., 2016). The living resources models have different temporal and spatial scales compared to the CBP watershed and estuarine models. An investment in protocols and software for coupling models (Warner et al. 2008a; Koralewski et al., 2019) will ensure consistency in the results across the living resource models that will aid in interpretation and comparisons across species and food webs. Creating these protocols in the co-production framework of the CBP will also serve to build the same transparency and trust that is already in place for the airshed, watershed, land use, hydrodynamic, and estuarine models.

The CBP Partnership would benefit from the establishment an Ecosystem Modeling Subcommittee responsible for both tidal and non-tidal aquatic systems. The collection of examples and food web models, along with other living resource models developed outside of the CBP and fisheries management models, can serve as prototypes for a more comprehensive analysis of water quality effects on Chesapeake Bay living resources. The Subcommittee should adopt a portfolio of modeling approaches for living resources that includes agreed-upon protocols for: (1) analyzing output from the CBP Modeling System from a habitat/organism perspective; (2) translating CBP Modeling System output to develop habitat or growth suitability indices; (3) using CBP Modeling System output as input for living resource and higher trophic level models; and (4) integrating organisms into the ICM water quality model as has been done, for example, with benthic filter feeders and SAV.

An explicit strategy for further incorporating living resources into the modeling system would encourage effective and efficient progress. As a starting point, the Subcommittee could consider non-linear responses of living resources to nutrients and sediment, new approaches and purposes for modeling primary producers that include botanical processes, and re-integrating consumers into the ICM biogeochemical model to facilitate coupling to higher trophic levels (i.e., zooplankton, which is not explicitly included in the ICM model). The Subcommittee could also articulate mechanisms for communication and stakeholder involvement, emphasizing that modeling living resource responses allows communication of co-benefits of restoration to stakeholders.

3. Lessons Learned

The CBP modeling system has contributed to several management successes that are due, in large part, to broad acceptance of the models by the scientific, management and stakeholder communities. This acceptance is the result of several factors. First, the members and participants of the Modeling Workgroup adhere to a set of core values that have promoted: 1) integration of the most recent air, watershed, and estuarine research and knowledge to support modeling for restoration decision making; 2) innovating, embracing creativity, and encouraging improvement in the development and support of transparent and robust modeling tools; 3) independence in making modeling decisions on the basis of best available evidence and using the most appropriate methods to produce, run, and interpret models, independent of policy considerations; and 4) inclusiveness with a strong commitment to an open and transparent process and the engagement of relevant partners, that results in strengthening the CBP's decision making tools. Adherence to these values for more than three decades has resulted in a buildup of trust among scientific, management and stakeholder communities.

In addition, the CBP models have always been developed in phases. For example, Phase 6 of the watershed model development was completed in 2017 and the CBP is now working on the next generation of models for the 2025 assessment – now designated as Phase 7. The use of phases has several advantages. It provides a subtle reminder to scientists and managers that the CBP models are continually evolving. It also reminds scientists and managers of their approval of the previous modeling phase, which facilitates approval of new models that are refined and improved versions of the previous model phase. In addition, the CBP partners and collaborators understand that whatever the current model phase, it too will be further refined and the known current model limitations will be addressed going forward in the ongoing evolution of the CBP models.

The formal procedures for model partnership development and approval are supported by CBP's long-standing commitment to being deeply collaborative, with a transparent approach to open-source model development and application. Another approach used by the CBP scientific community to increase CBP model transparency and access is convening technical transfer workshops on the models and tools to increase understanding and promote wide use. The CBP has also used its web sites and on-line documentation to create an extensive public record of what has been agreed to in CBP model development, including specifics of all major decisions and public access to the supporting technical material.

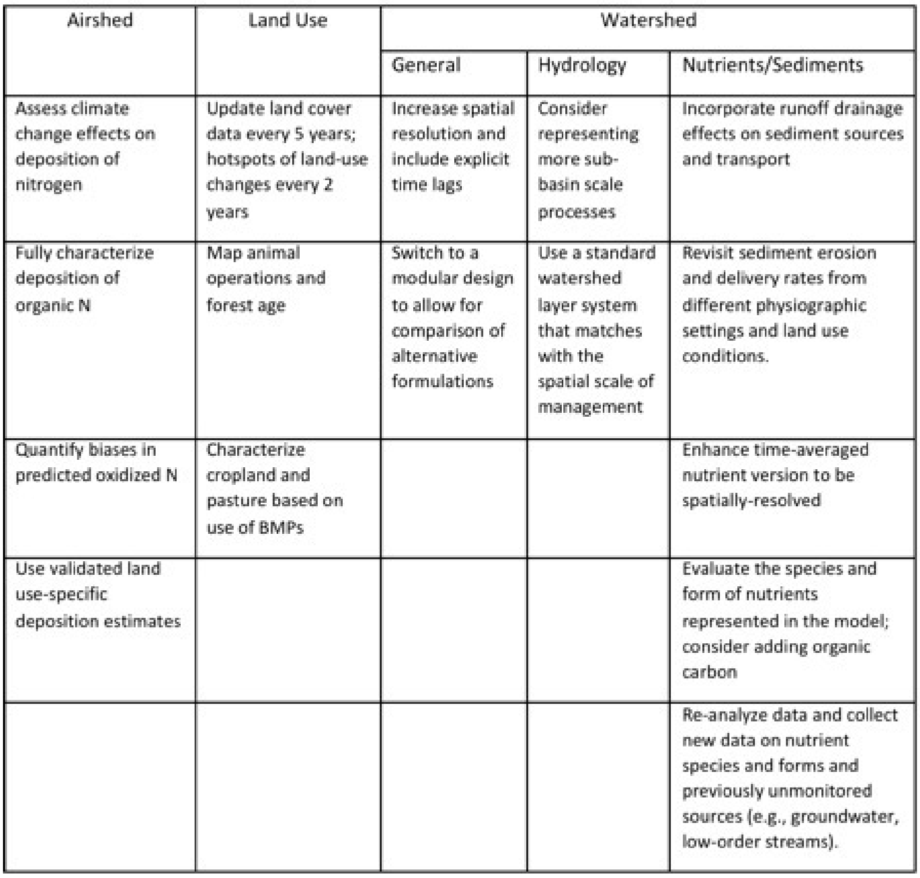
4. Summary and Going Forward

The CBP Partnership has used its linked modeling system as a planning tool to inform strategic management decisions toward Bay restoration since the 1980s. Over the last decade the modeling system has been used to formulate the 2010 Chesapeake Bay TMDL, evaluate progress and make mid-course adjustment in 2017, and inform the states' and DC's development of three phases of Watershed Implementation Plans that detail actions to be taken to reduce nutrients and sediment. Although model development has been driven by regulatory management needs, the development process is built on a foundation of monitoring, research, and collaborative engagement that cultivates the understanding necessary to manage water quality and habitat conditions in the Chesapeake Bay and its watershed. Given past successes in the CBP modeling system, there is ample evidence that new modeling tools will continue to be developed and incorporated into the modeling system to assist managers in setting, communicating, and achieving future TMDLs under uncertain future conditions influenced by varied scenarios of BMP implementation, land use, and climate change.

Envisioning the future of the CBP modeling system is timely. There have been recent advances in physical and biogeochemical process understanding, computer science, and environmental systems modeling approaches and techniques. The upcoming 2025 assessment offers an opportunity to continue this process through the use of improved models. This paper provides an overview of the 2017 CBP management modeling system and presents recommendations on potential improvements for 2025 and beyond. These improved models would better support and inform watershed management of nutrients and sediment for water quality goals and be an important step toward explicitly assessing management actions on living resources.

The recommendations are summarized for the various component models in Tables 3, 4, and 5. Tables 3 and 4 represent the Airshed, Land Use, and Watershed models by short-term (Table 3) and long-term (Table 4). The recommendations for the remaining Estuarine and Living Resources models are summarized in Table 5, without separation by time frame, but still with the order preceding as presented in the paper, which roughly places shorter-term recommendations first.

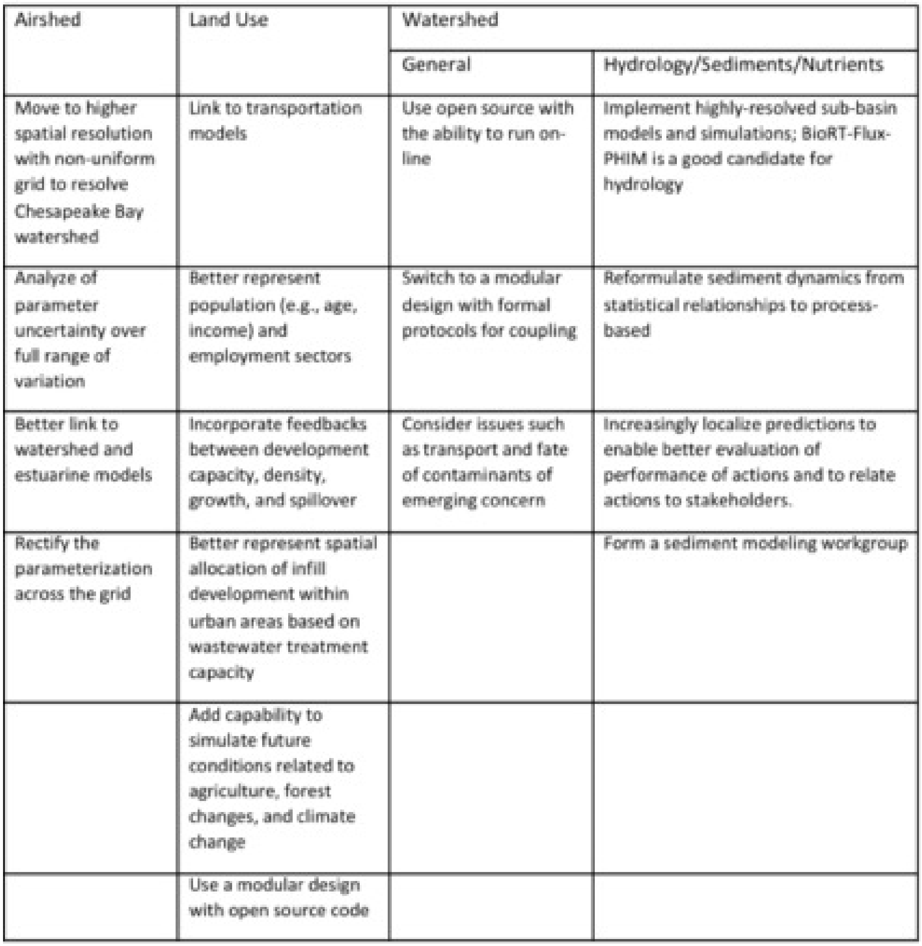
Table 3. Compilation of relatively short-term recommendations for the airshed, land use, and watershed models of the CBP modeling system. Each column is listing of recommendations in the order they appear in the text.



Airshed	Land Use	Watershed		
		General	Hydrology	Nutrients/Sediments
Assess climate change effects on deposition of nitrogen	Update land cover data every 5 years; hotspots of land-use changes every 2 years	Increase spatial resolution and include explicit time lags	Consider representing more sub-basin scale processes	Incorporate runoff drainage effects on sediment sources and transport
Fully characterize deposition of organic N	Map animal operations and forest age	Switch to a modular design to allow for comparison of alternative formulations	Use a standard watershed layer system that matches with the spatial scale of management	Revisit sediment erosion and delivery rates from different physiographic settings and land use conditions.
Quantify biases in predicted oxidized N	Characterize cropland and pasture based on use of BMPs			Enhance time-averaged nutrient version to be spatially-resolved
Use validated land use-specific deposition estimates				Evaluate the species and form of nutrients represented in the model; consider adding organic carbon
				Re-analyze data and collect new data on nutrient species and forms and previously unmonitored sources (e.g., groundwater, low-order streams).

Four major themes in the recommendations that apply to all the models are the need for: (1) finer spatial resolution; (2) improved connectivity and coupling of the component models; and (3) estimation of uncertainty. Modeling at higher and/or variable resolution would improve input to local watershed TMDLs and Watershed Implementation Plans, including fine-scale atmosphere, land use and watershed modeling capability. This starts with the need for better representation of watershed delivery mechanisms for surface water, nutrients, and sediment, as well as the changes to delivery patterns due to BMP implementation.

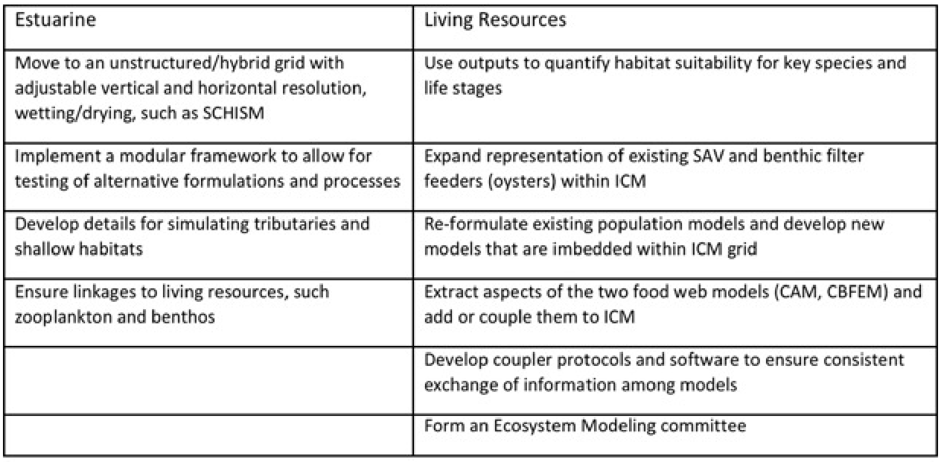
Table 4. Compilation of relatively long-term recommendations for the airshed, land use, and watershed models of the CBP modeling system. Each column is listing of recommendations in the order they appear in the text.



Airshed	Land Use	Watershed	
		General	Hydrology/Sediments/Nutrients
Move to higher spatial resolution with non-uniform grid to resolve Chesapeake Bay watershed	Link to transportation models	Use open source with the ability to run on-line	Implement highly-resolved sub-basin models and simulations; BioRT-Flux-PHIM is a good candidate for hydrology
Analyze of parameter uncertainty over full range of variation	Better represent population (e.g., age, income) and employment sectors	Switch to a modular design with formal protocols for coupling	Reformulate sediment dynamics from statistical relationships to process-based
Better link to watershed and estuarine models	Incorporate feedbacks between development capacity, density, growth, and spillover	Consider issues such as transport and fate of contaminants of emerging concern	Increasingly localize predictions to enable better evaluation of performance of actions and to relate actions to stakeholders.
Rectify the parameterization across the grid	Better represent spatial allocation of infill development within urban areas based on wastewater treatment capacity		Form a sediment modeling workgroup
	Add capability to simulate future conditions related to agriculture, forest changes, and climate change		
	Use a modular design with open source code		

While the models within the modeling system are linked, more explicit representation of the connectivity and coupling among the models would enhance their usability and interpretability. This includes how information is transferred among models and the adoption of modular approaches. There is an equally imperative demand for better simulations of the many linkages between water quality and living resources in the Bay to predict the effects of attainment of nutrient reduction goals on living resources and to create opportunities to understand and leverage co-benefits associated with restoration. Models that more fully represent higher trophic levels and ecosystem dynamics and feedbacks could provide a more complete picture of whether the conditions that support desirable living resource outcomes are being achieved.

Table 5. Compilation of recommendations for the Estuarine and Living Resources models of the CBP coupled modeling system. Each column is listing of recommendations in the order they appear in the text.



Estuarine	Living Resources
Move to an unstructured/hybrid grid with adjustable vertical and horizontal resolution, wetting/drying, such as SCHISM	Use outputs to quantify habitat suitability for key species and life stages
Implement a modular framework to allow for testing of alternative formulations and processes	Expand representation of existing SAV and benthic filter feeders (oysters) within ICM
Develop details for simulating tributaries and shallow habitats	Re-formulate existing population models and develop new models that are imbedded within ICM grid
Ensure linkages to living resources, such zooplankton and benthos	Extract aspects of the two food web models (CAM, CBFEM) and add or couple them to ICM
	Develop coupler protocols and software to ensure consistent exchange of information among models
	Form an Ecosystem Modeling committee

Many of the recommendations involve adding resolution or expanding aspects of the models and, therefore, have costs in computational time, development effort, data requirements, and parameter uncertainty. Some regions of the model domains may not benefit from further increases in resolution and so careful consideration should be given to determining where higher resolution will result in substantial management benefits. These models must be flexible and computationally efficient to enable scenario analysis with multiple runs (ensembles) to create probabilities of outcomes under different conditions.

Efforts aimed at characterizing the uncertainty in the CBP Partnership model projections (e.g., Irby and Friedrichs, 2019) should continue, in addition to independent verification and sensitivity testing to understand model skill. Many recommendations involved attempts to increase the confidence in model predictions, often by increasing resolution and by implementing more complicated process representations. Formal uncertainty analysis of large, coupled modeling systems is a challenge (Allen et al., 2007; Pianosi et al., 2016; Razavi and Gupta, 2015). A useful exercise is to also look for opportunities to simplify processes and formulations within the models, and to consider computational aspects to ensure simulations can be performed on the schedule needed by management decision-making.

Another direction to move forward is to integrate data-driven and process-based models (Karpatne et al., 2017; Reichstein et al., 2019; Shen, 2018), taking advantage of the strength of both models. As outlined in this paper, process-based models can offer process-based scientific insights and cause-consequence relationships. Machine learning techniques, on the other hand, can learn from data to facilitate the parameterization of process-based models and reduce model uncertainty. In particular, in recent years, deep learning approaches have gained momentum in hydrological forecasting (Fang et al., 2019; Rahmani et al., 2020; Shen, 2018). A recent study has also shown the promise of training a deep learning model (Long Short Term Memory, LSTM) at the continental scale using largely available hydrometeorology data to forecast dissolved oxygen (DO), an important water quality measure (Zhi et al., 2021). The model learned the theory of DO dependence on water temperature; it also indicated the critical needs of data collection under conditions that lead to DO peaks and troughs. Further data-driven model development can potentially lead to insights of temporal trends and spatial patterns that can advance hydro-biogeochemical theories and forecasting capabilities for water quality response to changing climate and human perturbations.

The development of the coupled modeling system and its use to inform management was, and will continue to be, a long-term process and investment. The CBP Partnership should continue its efforts to increase stakeholder engagement to create a shared vision of effective restoration strategies and to help guide model development and application. Linkages to the scientific community are also important to maintain the "pressure" of peer review so the models are up to date. The trust accumulated to date must be maintained into the future as recommendations are considered and implemented.

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References

Adamack, Aaron T, Kenneth A Rose, and Carl F Cerco. 2017. 'Simulating the effects of nutrient loading rates and hypoxia on bay anchovy in Chesapeake Bay using coupled hydrodynamic, water quality, and individual-based fish models.' in, *Modeling coastal hypoxia* (Springer).

Allen, JI, PJ Somerfield, and FJ Gilbert. 2007. 'Quantifying uncertainty in high-resolution coupled hydrodynamic-ecosystem models', *Journal of Marine Systems*, 64: 3-14.

Amin, M. G. M., H. D. Karsten, T. L. Veith, D. B. Beegle, and P. J. Kleinman. 2018. 'Conservation dairy farming impact on water quality in a karst watershed in northeastern US', *Agricultural Systems*, 165: 187-96.

Amin, M. G. M., T. L. Veith, A. S. Collick, H. D. Karsten, and A. R. Buda. 2017. 'Simulating hydrological and nonpoint source pollution processes in a karst watershed: a variable source area hydrology model evaluation', *Agricultural Water Management*, 180: 212-23.

Appel, K Wyat, Jesse O Bash, Kathleen M Fahey, Kristen M Foley, Robert C Gilliam, Christian Hogrefe, William T Hutzell, Daiwen Kang, Rohit Mathur, and Benjamin N Murphy. 2020. 'The Community Multiscale Air Quality (CMAQ) Model Versions 5.3 and 5.3. 1: System Updates and Evaluation', *Geoscientific Model Development Discussions*: 1-41.

Ator, S. W., J. W. Brakebill, and J. D. Blomquist. 2011. "Sources, fate, and transport of nitrogen and phosphorus in the Chesapeake Bay watershed—An empirical model: U.S. Geological Survey

Scientific Investigations Report 2011-5167." In.: http://pubs.usgs.gov/sir/2011/5167/.

Ator, S. W., and A. M. Garcia. 2016. 'Application of SPARROW modeling to understanding contaminant fate and transport from uplands to streams', *Journal of the American Water Resources Association*, https://doi-org.proxy-um.researchport.umd.edu/10.1111/1752-1688.12419

Ator, Scott W, Joel D Blomquist, James S Webber, and Jeffrey G Chanat. 2020. 'Factors driving nutrient trends in streams of the Chesapeake Bay watershed', *Journal of Environmental Quality*, 49: 812-34.

Ator, Scott W, Ana Maria García, Gregory E Schwarz, Joel D Blomquist, and Andrew J Sekellick. 2019. 'Toward explaining nitrogen and phosphorus trends in Chesapeake Bay Tributaries, 1992–2012', *JAWRA Journal of the American Water Resources Association*, 55: 1149-68.

Bao, C., L. Li, Y. Shi, and C. Duffy. 2017. 'Understanding hydrogeochemical processes at the watershed scale: 1. Development of RT-Flux-PIHM', *Water Resources Research*, doi: 10.1002/2016WR018934.

Bash, J. O., E. J. Cooter, R. L. Dennis, J. T. Walker, and J. E. Pleim. 2013. 'Evaluation of an regional airquality model with bidirectional NH3 exchange coupled to an agro-ecosystem model', *Biogeosciences*, 10: 1635-45.

Berg, J. , J. Burch, D. Cappuccitti, S. Filoso, L. Fraley-McNeal, D. Goerman, N. Hardman, S. Kaushal, D. Median, M. Meyers, B. Kerr, S. Stewart, B. Sullivan, R. Walter, and J. Winters. 2013. 'Recommendations of the Expert Panel to Define Removal Rates for Individual Stream Restoration Projects. Prepared by T. Schueler, Chesapeake Stormwater Network,and B. Stack, Center for Watershed Protection. Test-drive revisions approved by the WQGIT September 8, 2014.', *https://www.chesapeakebay.net/documents/Stream_Panel_Report_Final_08282014_Appendices_A_G.pdf.*

Bever, Aaron J, Marjorie AM Friedrichs, Carl T Friedrichs, Malcolm E Scully, and Lyon WJ Lanerolle. 2013. 'Combining observations and numerical model results to improve estimates of hypoxic volume within the Chesapeake Bay, USA', *Journal of Geophysical Research: Oceans*, 118: 4924-44.

Bhatt, G., M. Kumar, and C. J. Duffy. 2014. 'A tightly coupled GIS and distributed hydrologic modeling framework', *Environmental Modeling and Software*, 62: 70-84.

Boesch, Donald F, Russell B Brinsfield, and Robert E Magnien. 2001. 'Chesapeake Bay eutrophication: Scientific understanding, ecosystem restoration, and challenges for agriculture', *Journal of Environmental Quality*, 30: 303-20.

Borah, D. K., and M. Bera. 2004. 'Wathershed-scale hydrologic and nonpoint-source pollution models: review of applications', *Transactions of the ASAE*, 47: 789-803.

Brady, Damian C, Jeremy M Testa, Dominic M Di Toro, Walter R Boynton, and W Michael Kemp. 2013. 'Sediment flux modeling: Calibration and application for coastal systems', *Estuarine, Coastal and Shelf Science*, 117: 107-24.

Brakebill, John W, Scott W Ator, and Gregory E Schwarz. 2010. 'Sources of Suspended-Sediment Flux in Streams of the Chesapeake Bay Watershed: A Regional Application of the SPARROW Model 1', *JAWRA Journal of the American Water Resources Association*, 46: 757-76.

Brookfield, Andrea E, Amy T Hansen, Pamela L Sullivan, Jonathan A Czuba, Matthew F Kirk, Li Li, Michelle E Newcomer, and Grace Wilkinson. 2021. 'Predicting algal blooms: Are we overlooking groundwater?', *Science of the Total Environment*: 144442.

Brown, C. W., R. R. Hood, W. Long, J. Jacobs, D. L. Ramers, C. Wazniak, J. D. Wiggert, R. Wood, and J. Xu. 2013. 'Ecological forecasting in Chesapeake Bay: Using a mechanistic-empirical modeling approach', *Journal of Marine Systems*, 125: 113-25.

Brush, Grace S. 2009. 'Historical land use, nitrogen, and coastal eutrophication: A paleoecological perspective', *Estuaries and Coasts*, 32: 18-28.

Buchheister, Andre, Christopher F Bonzek, James Gartland, and Robert J Latour. 2013. 'Patterns and drivers of the demersal fish community of Chesapeake Bay', *Marine Ecology Progress Series*, 481: 161-80.

Buda, A. . 2013. 'Surface-runoff generation and forms of overland flow.' in, *Treatise on Geomorphology* (Elsevier).

Buda, A. R., P. J. Kleinman, M. S. Srinivasan, R. B. Bryant, and G. W. Feyereisen. 2009. 'Factors influencing surface runoff generation from two agricultural hillslopes in central Pennsylvania', *Hyrdologic Processes*, 23: 1295-312.

Buda, A. R., P. J. A. Kleinman, R. B. Bryant, G. W. Feyereisen, D. A. Miller, P. G. Knight, and P. J. Drohan. 2013. 'Forecasting runoff from Pennsylvania landscapes', *Journal of Soil and Water Conservation*, 68: 185-98.

Campbell, Patrick C, Jesse O Bash, and Tanya L Spero. 2019. 'Updates to the Noah land surface model in WRF-CMAQ to improve simulated meteorology, air quality, and deposition', *Journal of advances in modeling earth systems*, 11: 231-56.

Cavalli, Marco, Sebastiano Trevisani, Francesco Comiti, and Lorenzo Marchi. 2013. 'Geomorphometric assessment of spatial sediment connectivity in small Alpine catchments', *Geomorphology*, 188: 31-41.

CBF. 2014. 'The Economic Benefits of Cleaning Up the Chesapeake Bay: A Valuation of the Natural Benefits Gained by Implementing the Chesapeake Clean Water Blueprint', *Chesapeake Bay Foundation, Annapolis, MD. http://www.cbf.org/economicbenefits*.

CBP. 2014. "Chesapeake Bay Watershed Agreement." In.: https://www.chesapeakebay.net/channel_files/24334/2014_chesapeake_watershed_agreement.pdf.

———. 2020a. https://cast.chesapeakebay.net/Documentation/ModelDocumentation.

———. 2020b. "Indicator Analysis and Methods Document: Bay Watershed Population, Updated 07/21/20. ." In.: https://www.chesapeakebay.net/state/population.

———. 2020c. 'Submerged aquatic vegetation', *https://www.chesapeakeprogress.com/abundant-life/sav*.

Cerco, C.   , S.-C. Kim, and M.   Noel. 2010. "The 2010 Chesapeake Bay eutrophication model." In.: Chesapeake Bay Program Office, US Environmental Protection Agency, Annapolis, MD.

Cerco, C. F., and T. Cole. 1993. 'Three-dimensional eutrophication model of Chesapeake Bay', *Journal of Environmental Engineering*, 119: 1006-25.

Cerco, C. F., and K. Moore. 2001. 'System-wide submerged aquatic vegetation model for Chesapeake Bay', *Estuaries*, 24: 522-34.

Cerco, C. F., and M. R. Noel. 2013. 'Twentyone year simulation of Chesapeake Bay water quality using CE-QUAL-ICM eutrophication model', *Journal of the American Water Resources Association*, 49: 1119-33.

Cerco, Carl F. 2000. 'Phytoplankton kinetics in the Chesapeake Bay eutrophication model', *Water Quality and Ecosystems Modeling*, 1: 5-49.

Cerco, Carl F, and Mark R Noel. 2004. 'Process-based primary production modeling in Chesapeake Bay', *Marine Ecology Progress Series*, 282: 45-58.

Christensen, V. , A. Beattie, C. Buchanan, H. Ma, S. D. J. Martell, R. J. Latour, D. Preikshot, B. Sigrist, J. H. Uphoff, C. J. Walters, R. J. Wood, and H. Townsend. 2009. 'Fisheries ecosystem model of the Chesapeake Bay: methodology, parameterization, and model explanation', *U.S. Department of Commerce, NOAA Technical Memorandum, NMFS-F/SPO-106*: 146 pp.

Christensen, V. , M. Coll, C. Piroddi, J. Steenbeek, J. Buszowski, and D. Pauly. 2014. 'A century of fish biomass decline in the ocean', *Marine Ecology Progress Series*, 512: 155-66.

Clark, J Blake, Wen Long, and Raleigh R Hood. 2017. 'Estuarine sediment dissolved organic matter dynamics in an enhanced sediment flux model', *Journal of Geophysical Research: Biogeosciences*, 122: 2669-82.

Cleaves, Emery T. 2003. 'Conceptual model for transferring information between small watersheds', *Environmental geology*, 45: 190-97.

Collick, A. S., D. R. Fuka, P. J. A. Kleinman, A. R. Buda, J. L. Weld, M. J. White, T. L. Veith, R. B. Bryant, C. H. Bolster, and Z. M. Easton. 2015. 'Prediting phosphorus dynamics in complex terrains using a variable source area hydrology model', *Hyrdologic Processes*, 29: 588-601.

Craig, Laura S, Margaret A Palmer, David C Richardson, Solange Filoso, Emily S Bernhardt, Brian P Bledsoe, Martin W Doyle, Peter M Groffman, Brooke A Hassett, and Sujay S Kaushal. 2008. 'Stream restoration strategies for reducing river nitrogen loads', *Frontiers in Ecology and the Environment*, 6: 529-38.

Craine, Joseph M, Andrew J Elmore, Lixin Wang, Julieta Aranibar, Marijn Bauters, Pascal Boeckx, Brooke E Crowley, Melissa A Dawes, Sylvain Delzon, and Alex Fajardo. 2018. 'Isotopic evidence for oligotrophication of terrestrial ecosystems', *Nature Ecology & Evolution*, 2: 1735-44.

Crear, Daniel P, Robert J Latour, Marjorie AM Friedrichs, Pierre St-Laurent, and Kevin C Weng. 2020. 'Sensitivity of a shark nursery habitat to a changing climate', *Marine Ecology Progress Series*, 652: 123-36.

Crear, Daniel P, Brian E Watkins, Marjorie AM Friedrichs, Pierre St-Laurent, and Kevin C Weng. 2020. 'Estimating Shifts in Phenology and Habitat Use of Cobia in Chesapeake Bay Under Climate Change', *Frontiers in Marine Science*, 7: 943.

Dalyander, P. S., and C. F. Cerco. 2010. 'Integration of a fish bioenergetics model into a spatially explicit water quality model: Application to menhaden in Chesapeake Bay', *Ecological Modeling*, 221: 1922-33.

Dance, S. 2016. 'Scientists give Chesapeake Bay its highest environmental grade since 1992', *Baltimore Sun, May 17, 6:38 pm.*

Dari, B., V. D. Nair, A. Sharpley, P. J. Kleinman, D. Franklin, and W. G. Harris. 2018. 'Consistency of the threshold phosphorus saturation ratio across a wide geographic range of acid soils', *Agrosystems, Geosciences & Environment*, 1: 1-8.

Dewar, H., T. Landers, and E. Ridlington. 2009. 'Watermen blues: economic, cultural and community impacts of poor water quality in the Chesapeake Bay', *Environment Maryland Research and Policy Center, Baltimore, MD, 41 p.*

Dietrich, William E, Dino G Bellugi, Leonard S Sklar, Jonathan D Stock, Arjun M Heimsath, and Joshua J Roering. 2003. 'Geomorphic transport laws for predicting landscape form and dynamics', *Geophysical Monograph-American Geophysical Union*, 135: 103-32.

DiToro, D. 2001. *Sediment Flux Modeling* (John Wiley and Sons: New York).

Dortch, Mark S, Raymond S Chapman, and Steven R Abt. 1992. 'Application of three-dimensional Lagrangian residual transport', *Journal of Hydraulic Engineering*, 118: 831-48.

Easton, Z. M., D. Scavia, D. Alexander, K. Boomer, P. Kleinman, A. Miller, J. Pizzuto, D. Smith, and C. Welty. 2017. "Scientific and Technical Advisory Committee Review of the Chesapeake Bay Program Phase 6 Watershed Model." In *STAC Publication Number 17-007*, 47 pp. Edgewater, MD.

Easton, Z. M., K. Stephenson, A. Collick, P. M. Feleming, E. Kellner, J. Martin, M. Ribaudo, and G. Shenk. 2020. "Increasing Effectiveness and Reducing the Cost of Non-Point Source Best Management Practice Implementation: Is Targeting the Answer?" In. Edgewater, MD: STAC Publication Number 20-002.

Easton, Z. M., M. T. Walter, and T. S. Steenhuis. 2008. 'Combined monitoring and modeling indicate the most effective agricultural best management practices', *Journal of Environmental Quality*, 37: 1798-809.

Edinger, J. , D. Brady, and J. Geyer. 1974. "Heat exchange and transport in the environment." In.: Departement of Geography and Environmental Engineering, Johns Hopkins University, Baltimore, MD.

Ehrlén, Johan, and William F Morris. 2015. 'Predicting changes in the distribution and abundance of species under environmental change', *Ecology Letters*, 18: 303-14.

Fang, Kuai, Xinye Ji, Chaopeng Shen, Noel Ludwig, Peter Godfrey, Tasnuva Mahjabin, and Christine Doughty. 2019. 'Combining a land surface model with groundwater model calibration to assess the impacts of groundwater pumping in a mountainous desert basin', *Advances in Water Resources*, 130: 12-28.

Feng, Yang, Marjorie AM Friedrichs, John Wilkin, Hanqin Tian, Qichun Yang, Eileen E Hofmann, Jerry D Wiggert, and Raleigh R Hood. 2015. 'Chesapeake Bay nitrogen fluxes derived from a land-estuarine ocean biogeochemical modeling system: Model description, evaluation, and nitrogen budgets', *Journal of Geophysical Research: Biogeosciences*, 120: 1666-95.

Filoso, Solange, and Margaret A Palmer. 2011. 'Assessing stream restoration effectiveness at reducing nitrogen export to downstream waters', *Ecological Applications*, 21: 1989-2006.

Filoso, Solange, Sean MC Smith, Michael R Williams, and Margaret A Palmer. 2015. 'The efficacy of constructed stream–wetland complexes at reducing the flux of suspended solids to Chesapeake Bay', *Environmental science & technology*, 49: 8986-94.

Foley, K. M., et al. 2010. 'Incremental testing of the Community Multiscale Air Quality (CMAQ) modeling system version 4.7', *Geoscientific Model Development*, 3: 205-26.

Friedrichs, M. A. M., and R. R. Hood, et al. in prep. 'Comparison of Shallow-water Models for Use in Supporting Chesapeake Bay Management Decision-making ', *STAC Publication Number TBA*: Edgewater, MD. 36 pp.

Gallegos, Charles L, and Peter W Bergstrom. 2005. 'Effects of a Prorocentrum minimum bloom on light availability for and potential impacts on submersed aquatic vegetation in upper Chesapeake Bay', *Harmful Algae*, 4: 553-74.

Ganju, Neil K, Mark J Brush, Brenda Rashleigh, Alfredo L Aretxabaleta, Pilar Del Barrio, Jason S Grear, Lora A Harris, Samuel J Lake, Grant McCardell, and James O'Donnell. 2016. 'Progress and challenges in coupled hydrodynamic-ecological estuarine modeling', *Estuaries and Coasts*, 39: 311-32.

Glibert, Patricia M, Donald M Anderson, Patrick Gentien, Edna Granéli, and Kevin G Sellner. 2005. 'The global, complex phenomena of harmful algal blooms'.

Glibert, Patricia M, and Michele A Burford. 2017. 'Globally changing nutrient loads and harmful algal blooms: recent advances, new paradigms, and continuing challenges', *Oceanography*, 30: 58-69.

Guisan, Antoine, and Wilfried Thuiller. 2005. 'Predicting species distribution: offering more than simple habitat models', *Ecology Letters*, 8: 993-1009.

Hagy, J. D., W. R. Boynton, C. W. Keefe, and K. V. Wood. 2004. 'Hypoxia in Chesapeake Bay, 1950–2001: Long-term change in relation to nutrient loading and river flow', *Estuaries*, 27: 634-58.

Harding, J. M., and R. Mann. 2001. 'Oyster reefs as fish habitat: opportunistic use of restored reefs by transient fishes', *Journal of Shellfish Research*, 20: 1208-20.

Harris, Courtney K, J Paul Rinehimer, and Sung-Chan Kim. 2013. 'Estimates of bed stresses within a model of Chesapeake Bay.' in, *Estuarine and Coastal Modeling (2011).*

Homer, Collin, Jon Dewitz, Limin Yang, Suming Jin, Patrick Danielson, George Xian, John Coulston, Nathaniel Herold, James Wickham, and Kevin Megown. 2015. 'Completion of the 2011 National Land

Cover Database for the conterminous United States–representing a decade of land cover change information', *Photogrammetric Engineering & Remote Sensing*, 81: 345-54.

Hood, R. R., G. Shenk, R. Dixon, W. Ball, J. Bash, C. F. Cerco, P. Claggett, L. Harris, T. F. Ihde, L. Linker, C. Sherwood, and L. Wainger. 2019. 'Chesapeake Bay Program Modeling in 2025 and Beyond: A Proactive Visioning Workshop', *STAC Publication Number 19-002*: 62.

Hudson, K., and J. T. Murray. 2016. 'Virginia Shellfish Aquaculture Situation and Outlook Report', *In VSG-15-01, VIMS Marine Resources Report No. 2016–4*.

Ihde, T. F., I. C. Kaplan, E. A. Fulton, I. A. Gray, M. Hasan, D. Bruce, W. Slacum, and H. M. Townsend. 2016. 'Design and parameterization of the Chesapeake Bay Atlantis Model: A spatially explicit end-to-end ecosystem model ', *U.S. Department of Commerce, NOAA Technical Memorandum, NMFS-F/SPO-166*: 145 pp.

Ihde, T. F., and H. M. Townsend. 2017. 'Accounting for multiple stressors influencing living marine resources in a complex estuarine ecosystem using an Atlantis model', *Ecological Modeling*, 365: 1-9.

Irby, Isaac D, and Marjorie AM Friedrichs. 2019. 'Evaluating confidence in the impact of regulatory nutrient reduction on Chesapeake Bay water quality', *Estuaries and Coasts*, 42: 16-32.

Irby, Isaac D, Marjorie AM Friedrichs, Fei Da, and Kyle E Hinson. 2018. 'The competing impacts of climate change and nutrient reductions on dissolved oxygen in Chesapeake Bay', *Biogeosciences*, 15: 2649-68.

Irby, Isaac D, Marjorie AM Friedrichs, Carl T Friedrichs, Aaron Bever, Raleigh R Hood, Lyon WJ Lanerolle, Ming Li, Lewis Linker, Malcolm Scully, and Kevin Sellner. 2016. 'Challenges associated with modeling low-oxygen waters in Chesapeake Bay: a multiple model comparison', *Biogeosciences*, 13: 2011.

Jassby, Alan D, and Trevor Platt. 1976. 'Mathematical formulation of the relationship between photosynthesis and light for phytoplankton', *Limnology and Oceanography*, 21: 540-47.

Johnson, B. H. , K. W. Kim, R. H. Heath, H. L. Butler, and B. B. Hsieh. 1991. 'Development and verification of a three-dimensional numerical hydrodynamic, salinity, and temperature model of the Chesapeake Bay ', *Report HL-91-7, U.S. Army Corp of Engineers Waterways Experiment Station, Vicksburg, MS.*

Kall, N. D., C. D. Porter, and S. W. Fincher. 2014. "MOVES model sensitivity analysis to assist in prioritizing input data efforts." In

107th Annual Air & Waste Management Association Conference. Long Beach, California, June 24-27.

Karpatne, Anuj, Gowtham Atluri, James H Faghmous, Michael Steinbach, Arindam Banerjee, Auroop Ganguly, Shashi Shekhar, Nagiza Samatova, and Vipin Kumar. 2017. 'Theory-guided data science: A new paradigm for scientific discovery from data', *IEEE Transactions on knowledge and data engineering*, 29: 2318-31.

Kelly, James T, Shannon N Koplitz, Kirk R Baker, Amara L Holder, Havala OT Pye, Benjamin N Murphy, Jesse O Bash, Barron H Henderson, Norman C Possiel, and Heather Simon. 2019. 'Assessing PM2. 5 model performance for the conterminous US with comparison to model performance statistics from 2007-2015', *Atmospheric environment*, 214: 116872.

Kemp, W Michael, Walter R Boynton, Jason E Adolf, Donald F Boesch, William C Boicourt, Grace Brush, Jeffrey C Cornwell, Thomas R Fisher, Patricia M Glibert, and Jim D Hagy. 2005. 'Eutrophication of Chesapeake Bay: historical trends and ecological interactions', *Marine Ecology Progress Series*, 303: 1-29.

Kim, Changseong, Hee Yoon Kang, Young-Jae Lee, Sung-Gyu Yun, and Chang-Keun Kang. 2020. 'Isotopic variation of macroinvertebrates and their sources of organic matter along an estuarine gradient', *Estuaries and Coasts*, 43: 496-511.

Kim, S-C. 2013. 'Evaluation of a three-dimensional hydrodynamic model applied to Chesapeake Bay through long-term simulation of transport processes', *Journal of the American Water Resources Association*: doi:10.1111/jawr.12113.

Klemick, Heather, Charles Griffiths, Dennis Guignet, and Patrick Walsh. 2018. 'Improving water quality in an iconic estuary: an internal meta-analysis of property value impacts around the Chesapeake Bay', *Environmental and resource economics*, 69: 265-92.

Koralewski, Tomasz E, John K Westbrook, William E Grant, and Hsiao-Hsuan Wang. 2019. 'Coupling general physical environmental process models with specific question-driven ecological simulation models', *Ecological modelling*, 405: 102-05.

Koupal, J, TH DeFries, CF Palacios, and SW Fincher. 2013. "Study of MOVES Information for the National Emission Inventory: CRC Project A-84." In.

Landsberg, Jan H, Sherwood Hall, Jan N Johannessen, Kevin D White, Stephen M Conrad, Jay P Abbott, Leanne J Flewelling, R William Richardson, Robert W Dickey, and Edward LE Jester. 2006. 'Saxitoxin puffer fish poisoning in the United States, with the first report of Pyrodinium bahamense as the putative toxin source', *Environmental Health Perspectives*, 114: 1502-07.

Leavesley, GH, SL Markstrom, Pedro J Restrepo, and Roland J Viger. 2002. 'A modular approach to addressing model design, scale, and parameter estimation issues in distributed hydrological modelling', *Hydrological processes*, 16: 173-87.

Lefcheck, J. S., et al. 2018. 'Long-term nutrient reductions lead to the unprecedented recovery of a temperate coastal region', *Proceedings of the National Academy of Sciences of the United States of America*, doi: 10.1073/pnas.1715798115.

Lefcheck, Jonathan S, David J Wilcox, Rebecca R Murphy, Scott R Marion, and Robert J Orth. 2017. 'Multiple stressors threaten the imperiled coastal foundation species eelgrass (Zostera marina) in Chesapeake Bay, USA', *Global Change Biology*, 23: 3474-83.

Lewis, Kristy A, Kenneth A Rose, Kim De Mutsert, Shaye Sable, Cameron Ainsworth, Damian C Brady, and Howard Townsend. 2021. 'Using Multiple Ecological Models to Inform Environmental Decision-Making', *Frontiers in Marine Science*, 8: 283.

Li, L. . 2019. *Watershed reactive transport* (Mineralogical Society of America).

Linker, Lewis C, Richard A Batiuk, Gary W Shenk, and Carl F Cerco. 2013. 'Development of the Chesapeake Bay watershed total maximum daily load allocation', *JAWRA Journal of the American Water Resources Association*, 49: 986-1006.

Linker, Lewis C, Robin Dennis, Gary W Shenk, Richard A Batiuk, Jeffrey Grimm, and Ping Wang. 2013b. 'Computing atmospheric nutrient loads to the Chesapeake Bay watershed and tidal waters', *JAWRA Journal of the American Water Resources Association*, 49: 1025-41.

Linker, Lewis C, Gary W Shenk, Ping Wang, Katherine J Hopkins, and Sajan Pokharel. 2002. 'A short history of Chesapeake Bay modeling and the next generation of watershed and estuarine models', *Proceedings of the Water Environment Federation*, 14: 569-82.

Liu, J. , J. T. Spargo, P. J. A. Kleinman, R. Meinen, P. A. Moore, and D. B. Beegle. 2018a. 'Water-extractable phosphorus in animal manure and manure compost: quantities, characteristics, and temporal changes', *Journal of Environmental Quality* 47: 471-79.

Liu, J., T. L. Veith, A. S. Collick, P. J. A. Kleinman, D. B. Beegle, and R. B. Bryant. 2017. 'Seasonal manure application timing and storage effects on field- and watershed-level phosphorus losses', *Journal of Environmental Quality*, 46: 1403-12.

Liu, Xiaobo. 2015. 'A more accurate method using MOVES (Motor Vehicle Emission Simulator) to estimate emission burden for regional-level analysis', *Journal of the Air & Waste Management Association*, 65: 837-43.

Liu, Zhuo, Yinglong J Zhang, Harry V Wang, Hai Huang, Zhengui Wang, Fei Ye, and Mac Sisson. 2018b. 'Impact of small-scale structures on estuarine circulation', *Ocean Dynamics*, 68: 509-33.

Ma, Hongguang, Howard Townsend, Xinsheng Zhang, Maddy Sigrist, and Villy Christensen. 2010. 'Using a fisheries ecosystem model with a water quality model to explore trophic and habitat impacts on a fisheries stock: A case study of the blue crab population in the Chesapeake Bay', *Ecological modelling*, 221: 997-1004.

Macdonald, Paul, Chevonne H Laurenson, Arthur Johnson, and Leslie Tait. 2009. 'A comparison of catch rates of artificial lures from an automated handline fishery at Shetland, UK', *Fisheries Research*, 95: 379-85.

Marshall, HG, L Burchardt, TA Egerton, K Stefaniak, and M Lane. 2008. 'Potentially toxic cyanobacteria in Chesapeake Bay estuaries and a Virginia lake', *Advances in experimental medicine and biology*, 619: 172-73.

Miller, A, M Baker, K Boomer, D Merritts, K Prestegaard, and S Smith. 2019a. 'Legacy Sediment, Riparian Corridors, and Total Maximum Daily Loads', *STAC Publication*.

Miller, M. D., H. E. Gall, A. R. Buda, L. S. Saporito, T. L. Veith, C. M. White, C. F. Williams, K. J. Brasier, P. J. A. Kleinman, and J. E. Watson. 2019b. 'Load-discharge relationships reveal the efficacy of manure application practices on phosphorus and total solids losses from agricultural fields', *Agriculture, Ecosystems & Environment*, 272: 19-28.

Moeckel, Rolf. 2017. 'Constraints in household relocation: Modeling land-use/transport interactions that respect time and monetary budgets', *Journal of Transport and Land Use*, 10: 211-28.

Moriarty, Julia M, Marjorie AM Friedrichs, and Courtney K Harris. 2020. 'Seabed resuspension in the Chesapeake Bay: Implications for biogeochemical cycling and hypoxia', *Estuaries and Coasts*: 1-20.

Murray, A. G. , and J. S. Parslow. 1999. 'Modelling of nutrient impacts in Port Phillip Bay — a semi-enclosed marine Australian ecosystem', *Marine and Freshwater Research*, 50: 597-612.

Newell, R. I. E., J. C. Cornwell, and M. S. Owens. 2002. 'Influence of simulated bivalve biodeposition and micro-phytobenthos on sediment nitrogen dynamics: a laboratory study', *Limnology and Oceanography*, 47: 1367-79.

Ni, Wenfei, Ming Li, and Jeremy M Testa. 2020. 'Discerning effects of warming, sea level rise and nutrient management on long-term hypoxia trends in Chesapeake Bay', *Science of the Total Environment*, 737: 139717.

Nichols, James D, and Byron K Williams. 2006. 'Monitoring for conservation', *Trends in ecology & evolution*, 21: 668-73.

Nixon, S. W. 1995. 'Coastal marine eutrophication: A definition, social causes, and future concerns', *Ophelia*, 41: 199-219.

Noe, Gregory B, Matthew J Cashman, Katie Skalak, Allen Gellis, Kristina G Hopkins, Doug Moyer, James Webber, Adam Benthem, Kelly Maloney, and John Brakebill. 2020. 'Sediment dynamics and implications for management: State of the science from long-term research in the Chesapeake Bay watershed, USA', *Wiley Interdisciplinary Reviews: Water*, 7: e1454.

North, E. W., D. M. King, J. Xu, R. R. Hood, R. I. E. Newell, K. Paynter, M. L. Kellogg, M. K. Liddell, and D. F. Boesch. 2010. 'Linking optimization and ecological models in a decision support tool for oyster restoration and management', *Ecological Applications*, 20: 851-66.

Phillips, Scott W, and Bruce D Lindsey. 2003. *The influence of ground water on nitrogen delivery to the Chesapeake Bay* (US Department of the Interior, US Geological Survey).

Pianosi, Francesca, Keith Beven, Jim Freer, Jim W Hall, Jonathan Rougier, David B Stephenson, and Thorsten Wagener. 2016. 'Sensitivity analysis of environmental models: A systematic review with practical workflow', *Environmental Modelling & Software*, 79: 214-32.

Pizzuto, JE. 2014. 'Long-term storage and transport length scale of fine sediment: Analysis of a mercury release into a river', *Geophysical Research Letters*, 41: 5875-82.

Pyke, C. R. , K. Gottschalk, D. Theobald, A. Lister, T. T. Spano, and T. Johnson. 2010. "Review of land-use and land cover dataset and methodology." In. Edgewater, MD: Chesapeake Bay Program Scientific and Technical Advisory Committee, 21 pp.

Pyke, C. R., D. Parker, G. Pontius, B. C. Pijanowiski, and J. Kittle. 2008. "Chesapeake Bay land change modeling technical review." In, 26 pp. Edgewater, MD: Chesapeake Bay Program Scientific and Technical Advisory Committee.

Rahmani, Farshid, Kathryn Lawson, Wenyu Ouyang, Alison Appling, Samantha Oliver, and Chaopeng Shen. 2021. 'Exploring the exceptional performance of a deep learning stream temperature model and the value of streamflow data', *Environmental Research Letters*, 16: 024025.

Razavi, Saman, and Hoshin V Gupta. 2015. 'What do we mean by sensitivity analysis? The need for comprehensive characterization of “global” sensitivity in E arth and E nvironmental systems models', *Water Resources Research*, 51: 3070-92.

Reger, J, and Emery T Cleaves. 2008. 'Physiographic map of Maryland', *Maryland Geological Survey: Annapolis, MD*.

Reichstein, Markus, Gustau Camps-Valls, Bjorn Stevens, Martin Jung, Joachim Denzler, and Nuno Carvalhais. 2019. 'Deep learning and process understanding for data-driven Earth system science', *Nature*, 566: 195-204.

Roehl, JE. 1962. 'Sediment source areas, and delivery ratios influencing morphological factors', *Int. Assoc. Hydro. Sci.*, 59: 202-13.

Sanford, Ward E, and Jason P Pope. 2013. 'Quantifying groundwater's role in delaying improvements to Chesapeake Bay water quality', *Environmental science & technology*, 47: 13330-38.

Secor, David H, Lisa A Kerr, and Steven X Cadrin. 2009. 'Connectivity effects on productivity, stability, and persistence in a herring metapopulation model', *ICES Journal of Marine Science*, 66: 1726-32.

Shen, Chaopeng. 2018. 'A transdisciplinary review of deep learning research and its relevance for water resources scientists', *Water Resources Research*, 54: 8558-93.

Sheng, Y. P. 1986. 'A three-dimensional mathematical model of coastal, estuarine and lake currents using a boundary fitted grid', Report No. 585, ARAR Group of Titan Systems, Princeton, N.J.

Shenk, G. W., and L. C. Linker. 2013. 'Development and application of the 2010 Chesapeake Bay watershed total maximum daily load model', *Journal of the American Water Resources Association*, EPA-903-R-13-004, CBP/TRS-310-13-3.

Shenk, G., L. Wainger, C. Wu, P. Capel, M. Friedrichs, J. Hubbart, A. Iho, P. Kleinman, K. Sellner, and K. Stephenson. 2020. 'Assessing the Environment in Outcome Units.', *STAC Publication Number 20-003, Edgewater, MD. 34 pp.*

Smith, Sean, Patrick Belmont, and Peter R Wilcock. 2011. 'Closing the gap between watershed modeling, sediment budgeting, and stream restoration', *Geophysical Monograph Series*, 194: 293.

Smith, SMC, and PR Wilcock. 2015. 'Upland sediment supply and its relation to watershed sediment delivery in the contemporary mid-Atlantic Piedmont (USA)', *Geomorphology*, 232: 33-46.

St-Laurent, Pierre, Marjorie AM Friedrichs, Ming Li, and Wenfei Ni. 2019. 'Impacts of sea level rise on hypoxia in the Chesapeake Bay: A model intercomparison', *Virginia Institute of Marine Science, William & Mary. https://doi.org/10.25773/42XY-JT30.*

St-Laurent, Pierre, Marjorie AM Friedrichs, Raymond G Najjar, Elizabeth H Shadwick, Hanqin Tian, and Yuanzhi Yao. 2020. 'Relative impacts of global changes and regional watershed changes on the inorganic carbon balance of the Chesapeake Bay', *Biogeosciences*, 17: 3779-96.

Steefel, C. I., L. E. Beckingham, and G. Landrot. 2015. 'Micro-continuum approaches for modeling pore-scale geochemical processes', *Reviews in Minerology and Geochemistry*, 80: 217-46.

Tango, Peter J, and Richard A Batiuk. 2013. 'Deriving Chesapeake Bay water quality standards', *JAWRA Journal of the American Water Resources Association*, 49: 1007-24.

Tango, PJ, R Magnien, W Butler, C Luckett, M Luckenbach, R Lacouture, and C Poukish. 2005. 'Impacts and potential effects due to Prorocentrum minimum blooms in Chesapeake Bay', *Harmful Algae*, 4: 525-31.

Testa, Jeremy M, Yun Li, Younjoo J Lee, Ming Li, Damian C Brady, Dominic M Di Toro, and W Michael Kemp. 2017. 'Modeling physical and biogeochemical controls on dissolved oxygen in Chesapeake Bay: lessons learned from simple and complex approaches.' in, *Modeling Coastal Hypoxia* (Springer).

Testa, Jeremy M, Rebecca R Murphy, Damian C Brady, and William M Kemp. 2018. 'Nutrient-and climate-induced shifts in the phenology of linked biogeochemical cycles in a temperate estuary', *Frontiers in Marine Science*, 5: 114.

Tomer, Mark D. 2018. 'A twice-paired watershed experimental design to assess stacked practices through field-edge monitoring', *Journal of Soil and Water Conservation*, 73: 58-61.

Townsend, H. . 2014. 'Comparing and coupling a water quality and a fisheries ecosystem model of the Chesapeake Bay for the exploratory assessment of resource management strategies', *ICES Journal of Marine Science*, 71: 703-12.

Turner, Jessica S, Pierre St-Laurent, Marjorie AM Friedrichs, and Carl T Friedrichs. 2021. 'Effects of reduced shoreline erosion on Chesapeake Bay water clarity', *Science of the Total Environment*, 769: 145157.

Tyler, Robin M, Damian C Brady, and Timothy E Targett. 2009. 'Temporal and spatial dynamics of diel-cycling hypoxia in estuarine tributaries', *Estuaries and Coasts*, 32: 123-45.

USDA. 2013. "US Dept of Agriculture – Agricultural Research Service 2013. Science Documentation Revised Universal Soil Loss Equation Version 2." In, 355. https://www.ars.usda.gov/ARSUserFiles/60600505/RUSLE/RUSLE2_Science_Doc.pdf.

USEPA. 2010. "Chesapeake Bay Total Maximum Daily Load for Nitrogen, Phosphorus and Sediment." In.: U.S. Environmental Protection Agency Chesapeake Bay Program Office, Annapolis MD. https://www.epa.gov/chesapeake-bay-tmdl/chesapeake-bay-tmdl-document.

———. 2018. "Midpoint Assessment of the Chesapeake Bay Total Maximum Daily Load." In.: U.S. Environmental Protection Agency Chesapeake Bay Program Office, Annapolis MD. https://www.epa.gov/sites/production/files/2018-07/documents/factsheet-epa-midpoint-assessment-chesapeake-bay-tmdl.pdf.

Van Dolah, Elizabeth R, Michael Paolisso, Kevin Sellner, and Allen Place. 2016. 'Employing a socio-ecological systems approach to engage harmful algal bloom stakeholders', *Aquatic ecology*, 50: 577-94.

Veith, T. L., H. E. Gall, and K. R. Elkin. 2019. 'Seasonal manure application timing and storage effects on field- and watershed-level phosphorus losses', *Journal of Soil and Water Conservation*, in press.

Veith, T. L., M. L. Wolfe, and C. D. Heatwole. 2004. 'Cost-effective BMP placement: Optimization versus targeting ', *Transactions of the American Society of Agricultural and Biological Engineers*, 47: 1585-94.

Veith, Tamie L, Mary Leigh Wolfe, and Conrad D Heatwole. 2003. 'Optimization procedure for cost effective bmp placement at a watershed scale 1', *JAWRA Journal of the American Water Resources Association*, 39: 1331-43.

Wainger, L. , G. Van Houtven, R. Loomis, J. Messer, R. Beach, and M. Deerhake. 2013. 'Tradeoffs among ecosystem services, performance certainty, and cost-efficiency in implementation of the Chesapeake Bay Total Maximum Daily Load', *Agricultural and Resource Economics Review* 42: 196-224.

Wallace, C. W., G. McCarty, S. Lee, R. P. Brooks, T. L. Veith, P. J. A. Kleinman, and A. M. Sadeghi. 2018. 'Evaluating concentrated flowpaths in riparian forest buffer contributing areas using LiDAR imagery and topographic metrics', *Remote Sensing*, 10: 614.

Wang, Rui, Xuehui Guo, Da Pan, James T Kelly, Jesse O Bash, Kang Sun, Fabien Paulot, Lieven Clarisse, Martin Van Damme, and Simon Whitburn. 2021. 'Monthly patterns of ammonia over the contiguous United States at 2 km resolution', *Geophysical Research Letters*: e2020GL090579.

Warner, John C, Natalie Perlin, and Eric D Skyllingstad. 2008a. 'Using the Model Coupling Toolkit to couple earth system models', *Environmental Modelling & Software*, 23: 1240-49.

Warner, John C, Christopher R Sherwood, Richard P Signell, Courtney K Harris, and Hernan G Arango. 2008b. 'Development of a three-dimensional, regional, coupled wave, current, and sediment-transport model', *Computers & geosciences*, 34: 1284-306.

Weller, D. E. , and M. E. Baker. 2014. 'Cropland riparian buffers throughout Chesapeake Bay watershed: Spatial patterns and effects on nitrate loads delivered to streams', *Journal of the American Water Resources Association*, 50: 696-712.

Wen, Hang, Julia Perdrial, Benjamin W Abbott, Susana Bernal, Rémi Dupas, Sarah E Godsey, Adrian Harpold, Donna Rizzo, Kristen Underwood, and Thomas Adler. 2020. 'Temperature controls production but hydrology regulates export of dissolved organic carbon at the catchment scale', *Hydrology and Earth System Sciences*, 24: 945-66.

Wiggert, J. D., R. R. Hood, and C. W. Brown. 2017. 'Chapter 6: Modeling Hypoxia and Its Ecological Consequences in Chesapeake Bay.' in J. Dubravko, K. A. Rose, R. D. Hetland and K. Fennel (eds.), *Modeling Coastal Hypoxia: Numerical Simulations of Patterns, Controls and Effects of Dissolved Oxygen Dynamics* (Springer: Gewerbestrasse, Switzerland).

Williams, M. R., A. R. Buda, H. A. Elliot, A. S. Collick, C. Dell, and P. J. A. Kleinman. 2015. 'Linking nitrogen management, seep chemistry and stream water quality in two agricultural headwater watersheds', *Journal of Environmental Quality*, 44: 910-20.

Williams, Michael R, Gopal Bhatt, Solange Filoso, and Guido Yactayo. 2017. 'Stream restoration performance and its contribution to the Chesapeake Bay TMDL: Challenges posed by climate change in urban areas', *Estuaries and Coasts*, 40: 1227-46.

Williams, Michael R, Solange Filoso, Benjamin J Longstaff, and William C Dennison. 2010. 'Long-term trends of water quality and biotic metrics in Chesapeake Bay: 1986 to 2008', *Estuaries and Coasts*, 33: 1279-99.

WIP. 2019a. 'https://mde.maryland.gov/programs/Water/TMDL/TMDLImplementation/Documents/Phase%20III%20WIP%20Report/Final%20Phase%20III%20WIP%20Package/Phase%20III%20WIP%20Document/Phase%20III%20WIP-Final_Maryland_8.23.2019.pdf'.

———. 2019b. 'https://www.deq.virginia.gov/Portals/0/DEQ/Water/ChesapeakeBay/Final%20Phase%20III%20WIP/Virginia_Chesapeake_Bay_TMDL_Final_Phase%20III_WIP%20(2).pdf'.

———. 2019c. 'http://www.wvca.us/bay/files/bay_documents/1298_WV_WIP3_final_082319.pdf'.

———. 2019d. 'https://www.dep.pa.gov/Business/Water/Pennsylvania%E2%80%99s%20Chesapeake%20Bay%20Program%20Office/WIP3/Pages/PAs-Plan.aspx'.

Xu, J., and R. R. Hood. 2006. 'Modeling biogeochemical cycles in Chesapeake Bay with a coupled physical-biological model', *Estuarine and Coastal Shelf Science*, 69: 19-46.

Xu, J., W. Long, J. Wiggert, L. W. J. Lanerolle, C. W. Brown, R. Murtugudde, and R. R. Hood. 2011. 'Climate forcing and salinity variability in Chespeake Bay, USA', *Estuaries and Coasts*, DOI 10.1007/s12237-011-9423-5.

Ye, F., Y. Zhang, H. Wang, M. A. M. Friedrichs, I. D. Irby, E. Alteljevich, A. Valle-Levinson, Z. Wang, H. Huang, J. Shen, and J. Du. 2018. 'A 3D unstructured-grid model for Chesapeake Bay: importance of bathymetry', *Ocean Modelling*, 127: 16-39.

Zhang, Hongliang, Jingyi Li, Qi Ying, Jian Zhen Yu, Dui Wu, Yuan Cheng, Kebin He, and Jingkun Jiang. 2012. 'Source apportionment of PM2. 5 nitrate and sulfate in China using a source-oriented chemical transport model', *Atmospheric environment*, 62: 228-42.

Zhang, Qiang, Yixuan Zheng, Dan Tong, Min Shao, Shuxiao Wang, Yuanhang Zhang, Xiangde Xu, Jinnan Wang, Hong He, and Wenqing Liu. 2019b. 'Drivers of improved PM2. 5 air quality in China from 2013 to 2017', *Proceedings of the National Academy of Sciences*, 116: 24463-69.

Zhang, Y. , E. Ateljevich, H. C. Yu, C. H. Wu, and J. C. S. Yu. 2015. 'A new vertical coordinate system for a 3D unstructured-grid model', *Ocean Modelling*, 85: 16-31.

Zhang, Y., F. Ye, E. V. Staney, and S. Grashorn. 2016. 'Seamless cross-scale modeling with SCHISM', *Ocean Modelling*, 102: 64-81.

Zhang, Yuqiang, Kristen M Foley, Donna B Schwede, Jesse O Bash, Joseph P Pinto, and Robin L Dennis. 2019. 'A measurement-model fusion approach for improved wet deposition maps and trends', *Journal of Geophysical Research: Atmospheres*, 124: 4237-51.

Zhi, W. , and L. Li. 2019. 'Nitrate export regulated by different land uses', in review.

Zhi, Wei, Dapeng Feng, Wen-Ping Tsai, Gary Sterle, Adrian Harpold, Chaopeng Shen, and Li Li. 2021. 'From Hydrometeorology to River Water Quality: Can a Deep Learning Model Predict Dissolved Oxygen at the Continental Scale?', *Environmental science & technology*, 55: 2357-68.

Zhi, Wei, and Li Li. 2020. 'The shallow and deep hypothesis: Subsurface vertical chemical contrasts shape nitrate export patterns from different land uses', *Environmental science & technology*, 54: 11915-28.

Zhi, Wei, Li Li, Wenming Dong, Wendy Brown, Jason Kaye, Carl Steefel, and Kenneth H Williams. 2019. 'Distinct source water chemistry shapes contrasting concentration-discharge patterns', *Water Resources Research*, 55: 4233-51.