Disclaimer: This is provided to supplement discussion and consideration of the preliminary report from the Wetland Expert Panel to the Wetland Workgroup for September 15, 2016. Excerpts from the panel's full forthcoming report are provided below, but **these excerpts are not full chapters and have been edited or streamlined for purposes of preliminary report.** Effort has not been made to correct formatting or editorial issues in this document, while these issues will be addressed in the full report. General formatting of the text, such as labels for figures or other graphics may not be complete. Some parts of the full report are in final stages of editing or development; these sections have been removed ("editor's notes" or "…" have been inserted to help indicate where these changes were made). This will help avoid future confusion. Most of the following text is likely to be in the full report (either verbatim or revised), but **this document should not be cited nor used for review purposes** whatsoever once the full report is made available. Reading this document could, however, make reading and reviewing the full report easier for the reader since it serves as a partial preview.

Chapter 4. Review of available science – Non-Tidal wetland effects on water quality: an updated landscape perspective

Advancing a conceptual model to explain how wetland water quality and habitat benefits vary across space and time.

Predicting water quality and habitat benefits of wetlands across regional scales requires a systematic understanding of how hydrogeologic factors and watershed position combine to influence wetland form and function (Bedford, 1999). Hydrogeologic frameworks emphasize the importance of climate, surface relief and slope, thickness and permeability of soils, and the geochemical and hydraulic properties of underlying geologic materials (Winter, 1988, 1992). Stream classifications describe systematic changes and hydrologic interactions along the river corridor, from headwater reaches and associated wetlands to delta ecosystems (e.g., Brinson, 1993b; Church, 2002; Rosgen, 1994; Vannote et al., 1980). Hydrogeomorphic (HGM) frameworks combine these conceptual models to describe how wetland hydrodynamics and hydrologically-influenced geochemical variables vary across space and time (Brinson, 1993; Brooks *et al.*, 2014; Euliss *et al.*, 2004); thus when the HGM framework is presented in the context of a physiographic setting, it provides a compelling basis to capture variability in wetland function and to predict water quality benefits of different wetland types within a region. Accordingly, the panel combined these frameworks to describe how biogeochemical processes affecting transport and delivery of excess nutrients and sediment might vary in wetlands across the Chesapeake Bay watershed. Results build on the work of Lowrance and others (1997) by emphasizing linkages between wetland function and watershed position, given physiographic setting.

The hydrogeologic setting controls ground- and surface-water interactions and the role of wetlands as nutrient and sediment sinks, sources, and transformers (Winter, 1999). In upland areas, depth to bedrock, soil infiltration capacity, and topographic relief strongly influence the

amount of rainfall and the rate at which it is delivered to waterways versus infiltration to the shallow groundwater system. Shallow bedrock and steep terrain typical of mountainous ridge and valley regions result in rapid runoff rates, narrow stream/river corridors, and wetlands development primarily in valley bottoms. Steep upland land surfaces can cause erosion and transport of sediment and phosphorus to streams. In contrast, deep, unconsolidated sedimentary deposits across flat terrains, such as those defining much of the Coastal Plain, allow development of broad, expansive wetlands along entire stream networks. The relative influence of surface runoff versus infiltration controls the quantity and rate at which contaminants of concern are delivered to down-gradient wetlands. In addition, the chemical structure of a contaminant strongly influences delivery mechanisms. For example, while phosphorus and sediments are transported primarily through overland processes, nitrogen primarily enters streams in the form of nitrate dissolved in groundwater.

Where productive shallow groundwater systems develop, the potential for wetlands to capture excess nitrate depends on the thickness of the surficial aquifer above a confining layer (e.g., fine-grain, clay stratum, consolidated hardpan, or capstone bedrock) and the resulting hydrologic connectivity with wetland soils. This stratigraphy determines the potential for nitrate-enriched groundwater to flow through reduced, organic-rich wetland sediments ideal for denitrification (Hill *et al.*, 2004; Vidon and Hill, 2004a, 2006). Phosphorus retention depends on physical factors affecting erosion and deposition as well as hydrochemical conditions affecting phosphorus chemistry. Flat open areas typical of valley bottoms and bottom lands slow flow velocities and allow sedimentation. Steep upland land surfaces can cause erosion and transport of sediment and phosphorus to streams.

Consideration of watershed position can further expand the basis for evaluating how wetland function varies across space and time (Brinson, 1993b). Stream classifications describe variation in hydrobiological function in position along a stream network, recognizing systematic changes as headwater streams converge ultimately to form large-order rivers (e.g., Brinson, 1993b; Church, 2002; Rosgen, 1994; Vannote et al., 1980). Most describe the 'riverine landscape' to include the open water channel zone, headwater wetlands, and adjacent riparian or floodplain zones. In less disturbed systems, the relative importance of overland flow, ground-water contributions, and surface water inundation changes systematically along this up-stream to downstream continuum:

- Upland areas include the majority of a watershed and are defined as where stream channels connect directly to hillslopes and where sediment mobilized on upland slopes moves directly into the stream channel at the slope base (Church, 2002). In these areas, headwater wetlands, including many depressional and sloping, and riparian wetlands, provide important nutrient, sediment and carbon sinks (Church, 2002; Cohen et al., 2016). Uplands are groundwater recharge areas where soils and surficial sediments are permeable.
- Upland valley regions refer to portions of the stream network that function primarily as transfer zones (Church, 2002). These low-order streams tend to have the greatest

capacity to transport sediments downstream (i.e., stream power; Bagnold, 1966) thus often limiting in-stream biota (Church, 2002). These reaches also have the greatest frequency of adjacent sloping wetlands where advective groundwater flow controls water table position and the delivery of nutrients (Devito et al., 1999).

- The main valley forming the "backbone" of the drainage system accumulates alluvial materials along the channel and within adjacent floodplains due to much lower gradients (Church, 2002). Here, "sediment recruitment and onward transfer become purely consequences of erosion of the streambed and banks", with the former dominating further upstream and depositional processes becoming increasingly important downstream toward the distal end of stream networks (Church, 2002).

Combining the underlying principles of hydrogeology and stream classification, Brooks and others (2011) refined a hydrogeomorphic (HGM) classification of wetlands (Brinson, 1993) for the Mid-Atlantic Region (MAR), including the entire Chesapeake Bay watershed. The model broadly includes flats, depressions, and slope wetlands; lacustrine fringe, riverine floodplains, and tidal and non-tidal fringe wetlands. Importantly, the authors recognized distinct patterns in the distribution and hydrologic characteristics of wetlands across major physiographic provinces of the region (e.g., Ator *et al 2005*; Cole and Brooks, 2000), including the Appalachian Plateau, Appalachian Ridge and Valley, Piedmont and Coastal Plain (Figure XX). Each of the major wetland classes described below can occur in the different physiographic provinces, but the distribution and predominant geochemical controls vary across that space. Wetlands are most common in the relatively flat Coastal Plain followed by the Piedmont, and occur less frequently in the other physiographic provinces (Table 5). While information presented herein provides a generalized framework to better account wetland water quality functions within a TMDL framework, it is critical to recognize that the water quality services provided by an individual wetland strongly depends on hydrologic connectivity with sources of excess nutrients and sediment.

Flats develop where a combination of flat topography and slow infiltration results in precipitation accumulation at the surface. Accordingly, short-term weather patterns including evapotranspiration, primarily influence water table dynamics. In the Chesapeake Bay watershed, flats tend to occur on Coastal Plain interfluves (higher ground between two watercourses in the same drainage system) (Brinson, 1993). They are particularly common along the central topographic high of the Delmarva Peninsula between the Chesapeake Bay watershed and the Delaware Bay and Atlantic Ocean drainages. While flats sustain denitrifying conditions, these wetland sediments often do not intercept nitrate-enriched groundwater (Denver et al., 2014) or capture large quantities of surface overland flow because of their location along watershed drainage divides and small contributing areas. However, interception may occur where drainages drop down into flats at lower topographical positions within the watershed.

Depressional wetlands occur in topographic hollows and are controlled mainly by precipitation runoff, evapotranspiration, and also local interflow. Typically, these small wetlands lack surface water inlets or outlets. They form in areas up-gradient of headwater reaches and thus can

provide important areas of focused groundwater recharge. The small contributing areas often limit external supply of nutrients (Craft and Casey, 2000), however, because of their high ratio of perimeter to surface area and their frequent distribution across the landscape, depressional wetlands initially intercept surface runoff, thus providing important deposition areas (Cohen et al., 2016). Where these wetlands are located in agricultural fields, they can intercept and denitrify nitrate in or potentially entering groundwater (Denver et al, 2014). Areas with prior-converted cropland and hydric soils that are former depressional wetlands also can be areas of denitrification when soils are saturated. Further, low surface connectivity reduces exports to mitigate impacts on downstream waters, and retention rates are relatively high (Craft and Casey, 2000). Low pH (4 to 5.5) due to the predominant influence of precipitation, limits production and decomposition especially during wet seasons. Within the Chesapeake Bay watershed, depressional wetlands include the Delmarva Bays of the Outer Coastal Plain and ridge top wetlands of the Appalachian Ridge and Valley.

Sloping wetlands, including riparian corridors, often occur in association with headwater reaches where geologic discontinuities or breaks in topographic slope result in groundwater discharge to the land surface. As a result, the water table remains near the land surface (within 10 cm) and the plant rooting zone effectively is permanently saturated (Almendinger and Leete, 1998). Groundwater flow tends to occur in one direction, in relation to topographic gradients. Although saturated conditions retard decomposition and often result in the development of organic-rich peat soils, supplies of oxic, nitrate-rich ground water and generally neutral pH create biogeochemically active areas especially conducive to removing excess nitrogen (Gu *et al.*, 2008; Hill and Cardaci, 2004; Schipper *et al.*, 1993; Vidon and Hill, 2004b). These wetlands have the highest reported denitrification rates, although sub-oxic conditions also can enhance phosphorus availability and exacerbate downstream eutrophication, especially where human impacts have altered water chemistry (Boomer and Bedford, 2008; Dupas *et al.*, 2015; Lucassen *et al.*, 2004; Smolders *et al.*, 2010; Verhoeven *et al.*, 2008). Further, contaminated groundwater can bypass sloping wetlands and limit natural filter treatment, especially where surficial aquifer thickness is significantly greater than the depth of associated anoxic wetland sediments (Bohlke and Denver, 1995; Puckett, 2004; Tesoriero et al., 2009).

Riverine floodplains occur adjacent to waterways where overbank storm flow provides the dominant water source (Brinson, 1993). These surface-water driven systems generally have more variable water level fluctuations related to season and storm events compared to other wetland types, and also greater external supplies of nutrients. As a result nutrient availability, primary production, and decomposition rates are higher, especially where forested wetlands can establish stabilizing root zones. In addition, groundwater inflows from the local contributing area sustain water quality functions similar to sloping wetlands.

The Importance of Physiographic Setting: The form and distribution of wetlands strongly depend on climate and physiographic setting. Defining characteristics including topographic relief and geology strongly influence the relative importance of runoff vs infiltration to
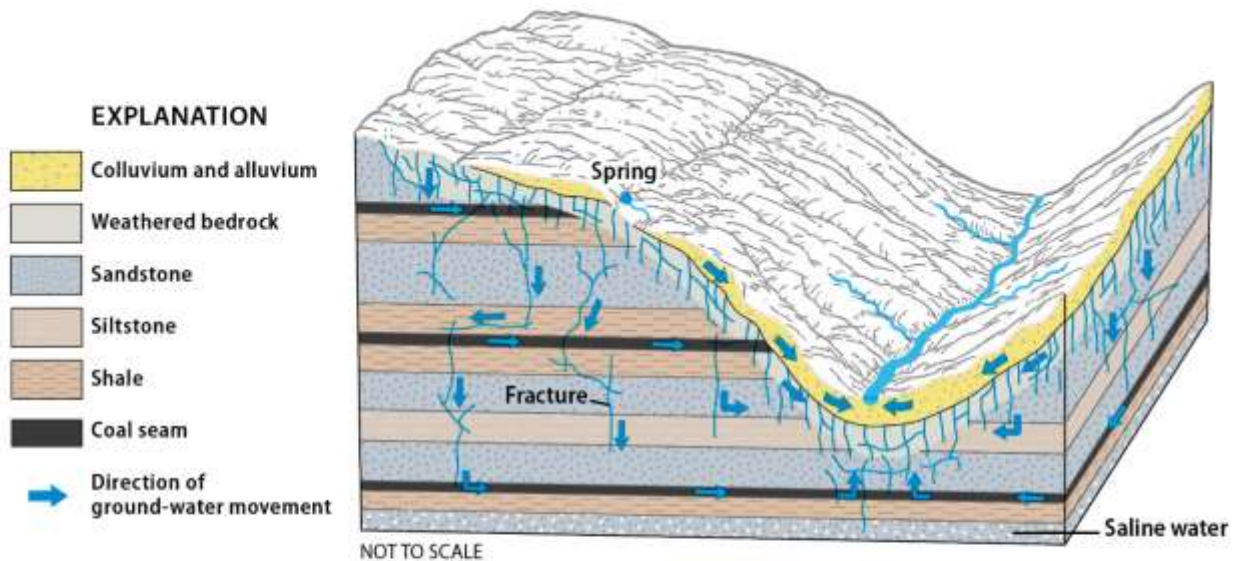
groundwater, where near-surface groundwater and surface water interactions support wetland development, and also the evolution of land use history. Together, these factors influence the distribution of different wetland types and the potential delivery of excess nutrients and sediment to these wetland systems. The Chesapeake Bay watershed can be divided into five major physiographic regions with additional sub-classes to summarize key characteristics that predominantly influence the form and function of wetlands throughout each sub-region (Figure xx). The distribution of wetlands varies widely across the physiographic regions.

The Appalachian Plateau extends across the most remote areas from the Bay, including the New York portion of the Bay watershed, across more than half of western Pennsylvania, and through small westernmost areas of Maryland and Virginia. The region is characterized by overlaying, consolidated sandstone and carbonate sedimentary rocks that are almost flat-lying to gently folded, but highly fractured, especially in more weathered units closer to the land surface (Figure xx; Trappe Jr. and Horn, 1997). In the unglaciated subregions, which includes much of the Appalachian Plateau in the Bay watershed, the region includes highly dissected waterways with adjacent slopes covered by thin accumulations of regolith; therefore, most precipitation runs to streams and only a small portion infiltrates to the groundwater system (Trappe Jr. and Horn, 1997). About 5 percent of the land in this area is wetlands, most of which are in floodplains in wide valleys and topographic lows formed upstream of erosion resistant bedrock stream contacts (Figure XXa; Fretwell *et al.*, 1996). Depression and sloping wetlands also occur where permeable, water-bearing strata outcrop dissected valley walls and sustain ground-water fed springs (Figure XXa; Fretwell *et al.*, 1996). In the glaciated regions of northern Pennsylvania and New York, depressional wetlands occur in association with glacial moraine deposits (Fretwell *et al.*, 1996). The average dissolved solids concentration is 230 milligrams per liter with a median pH of 7.3. Contaminated waters, notably from coal mining, generally are acidified and have higher concentrations of iron, manganese, sulfate, and dissolved solids (Trappe Jr. and Horn, 1997), all of which can strongly influence nutrient biogeochemistry. Limited development and agriculture in the region reduces the risk of contamination by excess nutrients and sediment.

Topography and shallow fracture systems determine ground-water movement in the aquifers of the Appalachian Plateaus. Water infiltrates weathered bedrock and moves mostly through near-surface factures; some water moves in a steplike fashion vertically along deeper fractures and horizontally through fractured sandstone or coal beds. Because of the absence of deep ground-water circulation and regional flow systems, saline water is at shallow depths.

Modified from Harlow, G.E., Jr., and LeCain, G.D., 1993, Hydraulic characteristics of, and ground-water flow in, coal-bearing rocks of southwestern Virginia: U.S. Geological Survey Water-Supply Paper 2388, 36 p.
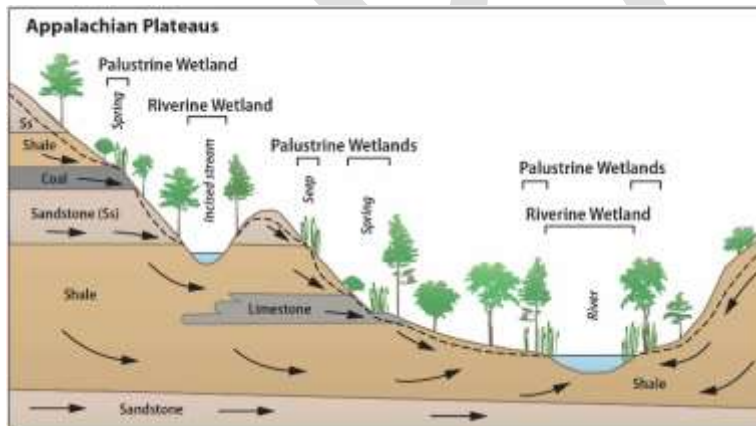
The **Appalachian Ridge and Valley** province is defined by alternating, distinctly linear valleys and ridges that trend southwest from northern New Jersey, through central Pennsylvania and Virginia, down to northern Georgia and Alabama. Similar to the Appalachian Plateau, bedrock consists mostly of sandstone, shale, and carbonate, with some locally important coal-bearing

units (Trappe Jr. and Horn, 1997). The stratum underlying the region's distinct topography, however, are highly deformed and folded and also more fractured (Trappe Jr. and Horn, 1997). Valley floor bottoms tend to have deeper accumulations of regolith. Groundwater tends to flow through ever-larger, subsurface conduits, until discharging at springs where wetlands are formed and cover about 2 percent of the land in this region. Three types of springs occur within the region (Trappe Jr. and Horn, 1997), including 1) contact springs where a water-bearing unit and underlying aquitard emerge at the land surface; 2) impermeable rock springs fed by fractures, joints or bedding planes in rocks; and 3) tubular springs that from where solution channels emerge. The latter are common in carbonate-rich, karst regions, described below in more detail. Water chemistry also is similar to resources across the Appalachian Plateau, although more variable and slightly more dilute: the average dissolved-solids concentration is 115 mg/L and median pH is 7.4. Contamination sources of water are generally from mining in the ridge areas; in the valleys, especially in areas underlain by carbonate rocks, high nitrate concentrations from agricultural sources are common.
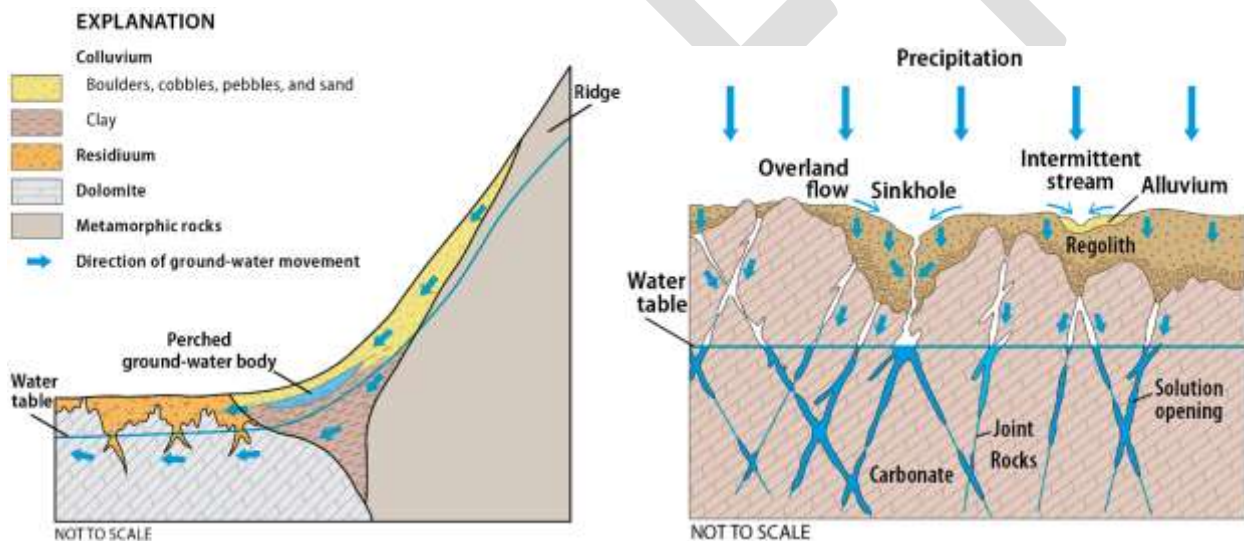
Figure 81 Thick wedges of colluvium on the lower flanks of ridges store large quantities of water that subsequently move into aquifers in the valleys. The colluvium commonly contains perched bodies of ground water that are separated from the main water table by clay confining units. Modified from Nutter, L.J., 1974a, Hydrogeology of Antietam Creek Basin: U.S. Geological Survey Journal of Research, v. 2, p. †249-252.

Modified from Nutter, L.J, 1973, Hydrogeology of the carbonate rocks, Frederick and Hagerstown Valleys: Maryland Geological Survey Report of Investigations 9, 70 p.

The **Blue Ridge Province** is characterized by its surrounding steep, mountainous slopes and numerous streams that feed into a broad valley with heavy rolling terrain, and deeply incised, fast flowing streams (Trappe Jr. and Horn, 1997). Underlying bedrock consists of highly faulted, folded, and fractured crystalline and siliciclastic bedrock (Denver *et al.*, 2010). As a result, the

groundwater system is unique to the sedimentary aquifers typical of other physiographic provinces in the region (LeGrand, 1988). Deep groundwater moves mainly through bedrock fractures. A mix of unconsolidated materials, which varies greatly in thickness, composition, and grain size, lays over top, resulting in highly variable hydraulic properties. The regolith is more permeable than the bedrock (Trappe Jr. and Horn, 1997), and groundwater flow generally is constrained to the unconfined aquifer. Flowpaths are relatively short, from recharge areas in uplands to local streams and springs; baseflow contributes more than 50 percent of annual stream discharge (Denver *et al.*, 2010). Wetlands occupy less than one percent of the region.

The **Piedmont** has similar geology to the adjacent Blue Ridge Province, but is distinguished by its low, gently rolling hills and moderate relief. To the east, the Fall Line demarcates where deeply weathered igneous and metamorphic rocks often exposed in the Piedmont are covered by sediments and separates the Piedmont from the Coastal Plain. With its terrain and shallow upland soils (less than 1 m thick) with slow infiltration rates, the Piedmont is predominantly an erosive environment (Markewich *et al.*, 1990). Groundwater occurs in unconfined conditions, in the bedrock fractures or in the overlying mantle of weathered regolith (Johnston, 1964). For more than 200 years, extensive forest clearing, agriculture, and milling operations have contributed significantly to the naturally deep valley floor deposits (Lowrance *et al.*, 1997; Walter and Merritts, 2008). As a result of natural and anthropogenic processes, the river-scape is entrenched or channelized through legacy sediments more than other regions in the Chesapeake Bay watershed (Donovan *et al.*, 2015). Baseflow supplied by the unconfined aquifer ranges between 50 and 75 percent of watershed discharge (Lowrance *et al.*, 1997). Wetlands typically are small and spring-fed, associated with slope changes in riparian or bedrock fracture zones (Fretwell *et al.*, 1996). Where connected and functioning, floodplain wetlands also provide significant nutrient and sediment trapping capacities (Schenk et al. 2013, Hupp et al. 2013). Overall, wetlands cover about 4 percent of the land area. Dissolved solids concentrations in natural waters of the Piedmont average 120 mg/L with a median pH of 6.7.

Carbonate deposits (karst terrain) in the Appalachian Plateau, Ridge and Valley, and Piedmont Provinces provide unique karst features that influence regional hydrology and the distribution of wetlands. Chemical dissolution of the bedrock creates a network of tunnels, caves, and related features that significantly increase groundwater transmissivity. Rapid groundwater drainage limits extensive wetland development (Fretwell *et al.*, 1996). Limestone outcrops, however, discharge calcium-bicarbonate rich waters that create unique ground-water fed wetland habitats and also uniquely influence wetland water chemistry. Ancient sink holes associated with subterranean karst network support depressional wetlands that typically are not directly connected by surface water flows to regional water ways, but may be connected in through spring discharge in other areas.

The **Coastal Plain** describes the broad wedge of unconsolidated sediments that occurs along the Atlantic Ocean coastline. Within the Chesapeake Bay watershed, the Coastal Plain deposits extend from the land surface, at the Piedmont Fall Line, on the Chesapeake Bay's western shore, to a depth of more than 8,000 feet along the Atlantic coastline (Debrewer *et al.*, 2007). Wetlands

are widely distributed throughout the Coastal Plain. The region can be divided into three sub-areas with distinctly different trends in wetland distributions and functions. The Inner Coastal Plain includes areas west of the Chesapeake Bay characterized by gently rolling hills. This area has the lowest percentage of wetlands (5 percent) compared to other Coastal Plain subregions. On the Eastern Shore, the Outer Coastal Plain Uplands include both poorly drained divides (wetlands cover about 34 percent of the land area) and well-drained regions (wetlands cover about 15 percent of the land area). In interior areas on poorly drained soils depression wetlands and expansive flats form, often along the major watershed divides. Narrow bands of palustrine wetlands provide riparian and floodplain functions. The Coastal Plain lowlands, includes low-lying areas on both sides of Chesapeake that occur generally within 25 feet of sea level. Here, the flat terrain and shallow regional water-table depth results in broad, unconstrained channels and expansive backwater areas (e.g., slacks or bottom-bottomland hardwood forests). These riverscapes are characterized by continuous inundation mainly driven by seasonal conditions rather than storm events, and limited directional flow (Brooks *et al.*, 2014). Precipitation, runoff from upland areas, and ground water from local and regional aquifer discharge also can contribute significantly to bottomland wetland water budgets (Fretwell *et al.*, 1996). Despite slow advective flow, however, bottomland wetlands provide important nutrient and sediment sinks (Noe and Hupp, 2005). Similar to the Piedmont and Great Valley regions, the Coastal Plain has sustained intensive development and agricultural land use, and contamination by excess nutrients and sediments occurs frequently.
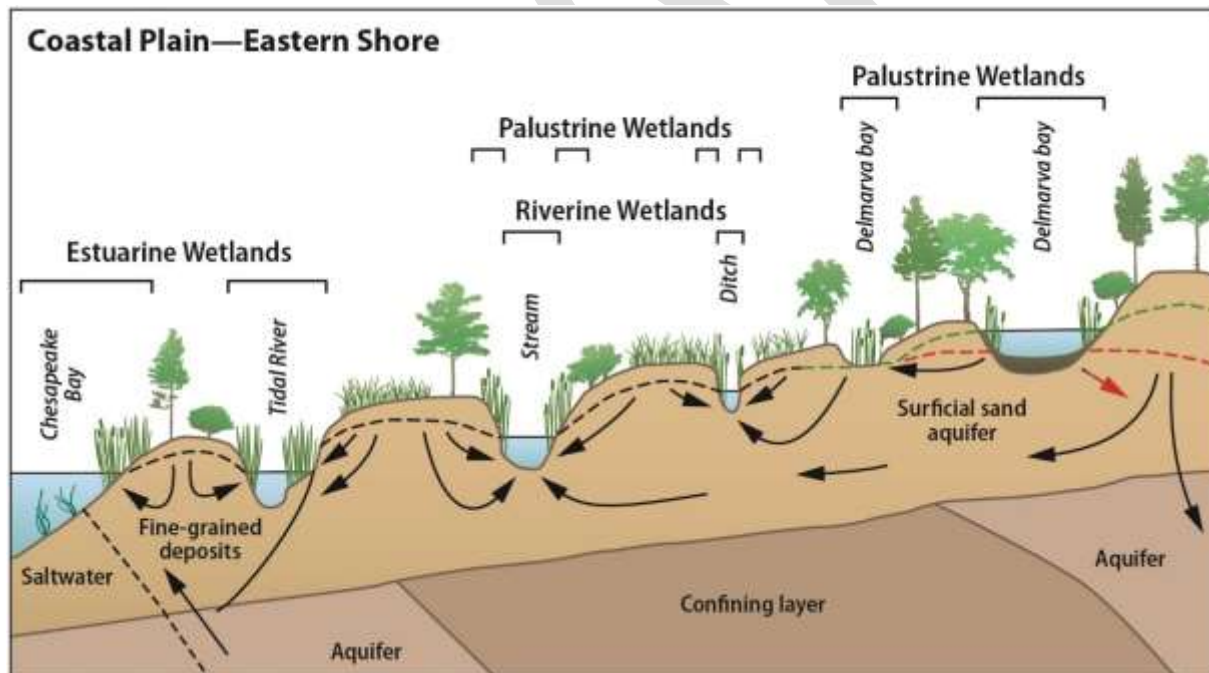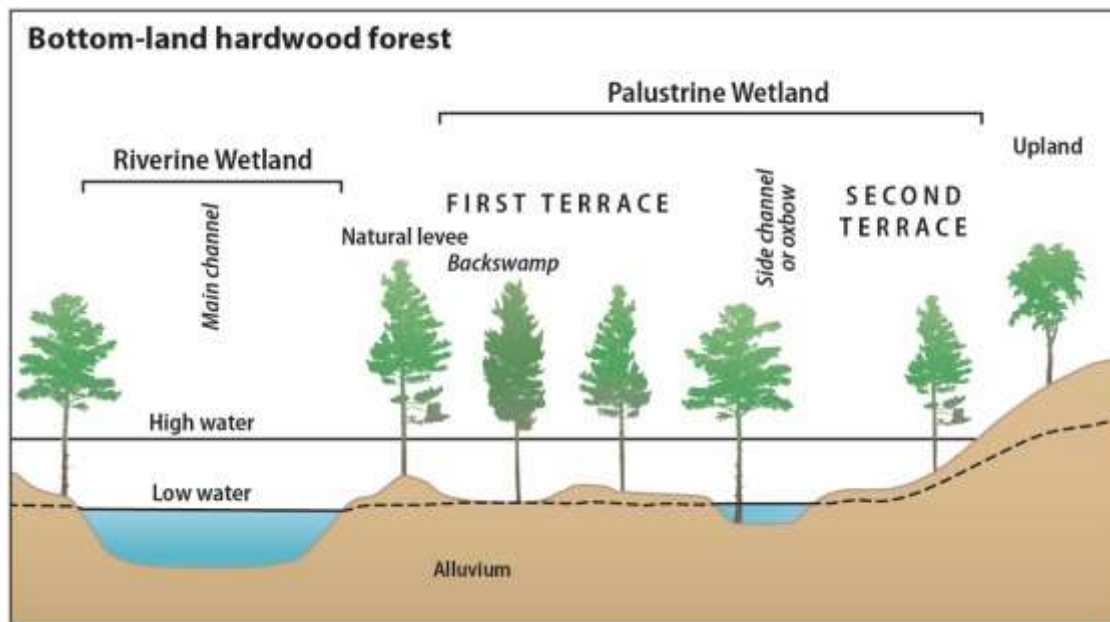
[Editor's note: A table, text and one or more maps will be provided in the full report to illustrate and summarize these physiographic regions and the distribution of nontidal wetlands within them.]

Advances in understanding how hydrogeologic setting influences wetlands nutrient dynamics

Nitrogen—transport and removal from groundwater and surface water

Our understanding of landscape controls on N transport and transformations has increased substantially over the past decade. Agricultural fields are a major source of nitrogen in many parts of the watershed (Ator *et al.*, 2011). In the Mid-Atlantic region, approximately 15% of applied fertilizer and manure leaches to the shallow aquifer (Puckett *et al.*, 2011). The most significant shallow aquifer contamination occurs in irrigated, well drained soils (e.g., carbonate-rich, karst terrain or the well-drained Outer Coastal Plain) where as much as 30% of applied nitrogen has been shown to leach into groundwater (Bohlke and Denver, 1995; Puckett *et al.*, 2011). Once delivered to the aquifer, nitrate often remains in that form, with limited biogeochemical transformation, due to high dissolved-oxygen levels and/or lack of carbon substrate which limits microbial denitrifier populations (Parkin and Meisinger, 1989; Yeomans *et al.*, 1992). Nitrate removal does not occur until the contaminated groundwater intersects carbon-rich soils, typically in wetlands (Carlyle and Hill, 2001; Duval and Hill, 2007; Green *et al.*, 2008; Hill and Cardaci, 20002004; Koretsky *et al.*, 2007). The distribution of wetlands, therefore likely provides an important control on nitrogen transport and stream water quality (Alexander *et al.*, 2007; Curie *et al.*, 2007; Oehler *et al.*, 2009).

The effectiveness of nitrogen removal via wetlands is dependent on the connectivity between wetlands and nitrogen sources (Goldman and Needleman, 2015; USEPA, 2015). The relative importance of stream baseflow contributed from groundwater versus stormflow generated by overland runoff affects the timing and form of N delivery to regional waterways. Where surface runoff dominates contributions to streams, such as in steep rocky terrains of the Appalachian Ridge and Valley Region, most N is in organic or ammonia forms and concentrations are generally low. As groundwater contributions to total stream flow increase, such as in the flat, unconsolidated Coastal Plain, nitrate typically becomes the dominant source of N. Most nitrate is formed in the soil zone and infiltrates to groundwater through the unsaturated zone.

On average, almost half of the N in surface water of the Chesapeake Bay watershed, including nonpoint and point sources, is nitrate from groundwater. Contributions of nitrate from groundwater at individual sites ranges between 17 to 80 percent (Bachman et al, 1998). The variability is due to differences in nitrogen application and hydrogeologic setting that affect the physical transport of water and nutrients, and the geochemical conditions that are encountered along surface and subsurface flowpaths. In general, Bay-wide areas with carbonate and crystalline rock aquifers have higher median nitrate concentrations in groundwater and streams than in areas with siliciclastic rocks (Ator and Ferrari, 1997). In the Coastal Plain, areas with thick sandy aquifer sediments have higher nitrate concentrations than in areas with thinner sequences of sandy sediments at the land surface (Ator et al., 2000). Areas with higher concentrations of nitrate in streams are directly correlated to higher inputs, even considering the potential for nitrate reduction by riparian and other wetlands.

Surface- and ground-water nitrogen may potentially be intercepted, especially where water sources enriched with nitrate intersect organic-rich substrates and enhance removal via denitrification. In general, such areas include headwater depression and sloping wetlands, riparian wetlands, and at the upland-wetland interface of floodplains bordering streams and rivers and poorly drained areas including shorelines of lakes, ponds, and the Chesapeake Bay. These settings commonly occur where near-surface ground- and/or surface-water interactions combined with finer-textured sediments slow water flow, resulting in saturated substrates that reduce decomposition rates and provide organic matter conducive to denitrifying conditions. While denitrification primarily occurs in carbon-rich wetland environments, this redox-sensitive processes also occur in older, less oxygenated groundwater of shallow aquifers in buried organic-rich estuarine deposits, near the boundary layers of overlying geologic stratums, or in contaminant plumes from landfills and other contaminant sources which provide carbon substrate to the denitrifying bacteria (Smedley and Edmunds, 2002). Denitrification in the shallow aquifer may account for as much as 10 percent of TN loss groundwater, or 1 to 2 percent of the total N load (Puckett *et al.*, 1999).

For water that is already in streams, overbank flooding of stormwater into floodplains has been shown to trap particulate N, absorb ammonia, and reduce nitrate in water that infiltrates through the organic-rich sediments (Noe, 2013). Several studies of flow-through wetlands (including restored wetlands) show significant reductions in N from wetland inlets to outlets (Woltemade

and Woodward, 2008; Seldomridge and Prestegaard, 2014; Kalin et al., 2013; Jordan et al., 2003). Uptake of nitrogen was affected by residence time and water temperature, and was most effective over longer residence time. Noe and Hupp (2005) noted retention of nitrogen in the floodplain where it is connected to streams in the Coastal Plain, but the disconnection of the river to the floodplain by channelization at one site resulted in very limited retention. Coastal Plain floodplains typically trap a large proportion of their annual river load of N, similar to the proportion of river load that is particulate N (Hoos and McMahon, 2009; Noe and Hupp, 2009).

Riparian-zone denitrification in slope wetlands is most effective where aquifer sediments are very thin in alluvial valleys and the discharging groundwater mostly passes through near-stream reducing conditions. This denitrification can occur in near-stream wetland sediments and the hyporheic zone (Pucket, 2004; Puckett et al., 2008; Ator and Denver, 2015). These conditions are common in the Coastal Plain on the Western Shore of the Chesapeake Bay and near the fall-line in the northern part of the Eastern Shore (Krantz and Powars, 2000; Ator et al, 2005). Models developed by Weller et al. (2011) indicated a potential high nitrate removal relative to upland inputs in this area, although groundwater data were not collected to verify upland nitrate concentrations. They can also exist in the Ridge and Valley provinces where water-bearing geologic units emerge at the land surface or where topographic slope changes between the valley walls and alluvial sediments (Winter et al., 1998).

Where the surficial aquifer is thick and groundwater flows along deeper flowpaths, much of the discharging groundwater can bypass reducing conditions in the near-stream riparian zone leading to limited potential for denitrification and elevated concentrations of nitrate in water discharging to a stream (Puckett, 2004; Böhlke and Denver, 1995; Baker, et al., 2001). This setting occurs in areas of the Piedmont with thick weathered bedrock sediments at the land surface and in parts of the Coastal Plain with a thick surficial aquifer, as is common on the Eastern Shore (Bachman et al., 1998; Ator and Denver, 2015). It also occurs in carbonate areas where most water in streams originates in springs that are fed by solution channels in the underlying carbonate rocks (Bachman et al., 1998). The widespread distribution of high nitrate concentrations in streams indicates that settings resulting in groundwater bypassing reducing conditions in near-stream areas are common in parts of the Chesapeake Bay region.

The potential for nitrogen removal by wetlands is highly variable and dependent on numerous factors, many of which are difficult to determine without local studies of particular areas. It is important to consider all types of available information and to include local hydrogeology for nitrate transport. Data sources that only look at the land surface are not adequate to determine subsurface processes, but are critical for understanding inputs and potential hydraulic flow paths from upland source areas to discharge areas in streams and rivers.

Phosphorus—fate, transport, and removal from groundwater and surface water

The highly dynamic and complicated pathways that regulate downstream P delivery continue to challenge our ability to predict P fluxes in relation to landscape setting and management practices. Because dissolved P concentrations originating from arable upland areas generally are

low or below detection in groundwater (Denver et al, 2014; Lindsey et al, 2014), storm-based sediment transport and floodplain deposition have been considered the primary mechanisms controlling delivery of excess phosphorus to downstream aquatic habitat (Kröger *et al.*, 2012). Increasing evidence of P-saturated soils and potential for increasing P bioavailablity, however, have raised concerns about the role of wetlands for P management (Sharpley *et al.*, 2014). While organic-rich, wetland soils can provide critically important ecosystem storage compartments for long-term P storage (Bridgham *et al.*, 2001; Dunne *et al.*, 2007; Reddy *et al.*, 1999), anoxic conditions can also contribute to downstream eutrophication (House, 2003; Smolders *et al.*, 1995). The following provides a brief overview of how different wetland types may influence P-availability throughout the Chesapeake Bay watershed, recognizing that these natural filter processes are strongly influenced by local topography and water chemistry along a stream network.

At the watershed-scale, hillslope processes strongly influence P transport and storage: 50 to 90 percent of P is tied up in recalcitrant forms, and physical processes including erosion, sediment transport and deposition, and burial are considered the primary mechanisms regulating P availability across the landscape. Approximately 80 percent of annual river loads of P are attached to sediment (Hupp *et al.*, 2009). Vegetated wetlands provide important deposition zones. With inundation, water flow velocity tends to slow across wetlands and allow sedimentation (Zedler, 2003).

Physical processes more specific to wetland environments, specifically variation in the frequency, magnitude, duration, and timing of flooding regulate P storage and export. Prolonged flooding reduces decomposition rates and increases accumulation of organic matter (Gambrell and Patrick, 1978; Mitsch and Gosselink, 2000), thus providing a long-term storage pool (Dunne *et al.*, 2007). Conversely, water table drawdown and soil aeration more typical of floodplain wetlands enhances decomposition, organic matter mineralization, and P release (Venterink *et al.*, 2001). Importantly, P dynamics vary across individual sites; for example, soil P mineralization varies laterally across Chesapeake floodplains associated with gradients or water flux, nutrient inputs, soil texture, and soil pH (Noe *et al.*, 2013).

The interaction of natural waters and organic-rich substrates creates a unique biogeochemical environment that strongly influences P dynamics depending on pH and redox conditions (Reddy *et al.*, 1999). In acidic, mineral wetland soils, more typical of flats and intermittently inundated floodplains, P sorption is closely related to hydrogen ion activity, organic matter content, and subsequent effects on amorphous (non-crystalline) aluminum and iron dynamics (Axt and Walbridge, 1999; Richardson, 1985). Under circumneutral pH conditions, redox conditions play a more prominent role than pH-controls in regulating P availability (Carlyle and Hill, 2001; Lamers *et al.*, 2002; Lucassen *et al.*, 2005; Smolders *et al.*, 2010). In particular, the redox-sensitive Fe-bound P-pools are highly dynamic and affected by short-term hydrologic condition and subsequent effects on water chemistry (House 2003, Richardson 1985, Walbridge and Struthers 1993). Under aerobic conditions, iron-oxides rapidly precipitate with P sorbing to the mineral surfaces (Patrick and Khalid, 1974). For example, in areas of the Outer Coastal Plain,

naturally high phosphorus and iron concentrations occur in groundwater associated with reduced, estuarine deposits; in wetlands where the groundwater emerges at the land surface, exposure to the atmosphere enhances iron mineral precipitation and P co-precipitation, thus reducing P availability (Bricker *et al.*, 2003). More typically, however, reduced wetland soils enhance P availability (Reddy *et al.*, 1999) and can result in eutrophication, especially where nitrate- or sulfate-contaminated waters enhance iron-P dissolution (Lucassen *et al.*, 2004; Smolders and Roelofs, 1993; Smolders *et al.*, 2006, 2010). In alkaline, reduced environments, likely to occur where calcium-bicarbonate rich water discharge, co-precipitation with calcium minerals can limit phosphorus availability (Moore and Reddy, 1994). Alkaline conditions (pH greater than 9) with Ca-concentrations greater than 100 mg/L) limit P solubility by enhancing Ca-P precipitation (Diaz et al. 1994; Plant and House 2002).

Although soils have a high capacity to sorb phosphorus, the filtration process can be overloaded, resulting in groundwater P contamination (Lory and Program, 1999). For example, sandy soils commonly formed across the Outer Coastal Plain provide limited mineral sorption sites. In addition, macropore features associated with karst geology or created by organism burrowing or root growth and decay (Harvey and Nuttle, 1995), often increase groundwater recharge and limit opportunities for wetland biogeochemical processes. While these processes can elevate phosphorus concentrations in stream baseflow, however, impacts to surface water quality are relatively small when compared to the quantity of sediment sorbed P delivered by surface water (Denver *et al.*, 2010).

Sediment—fate, transport, and removal from surface water

Sediment transport and deposition processes related to wetlands play an important role in regulating downstream water clarity and water quality. The relatively flat terrain of all wetland types compared to the surrounding contributing area results in significant sediment deposition at the upland-wetland edge. For any given wetland, the importance of this function depends largely on the form of the wetland (e.g., size, slope, soil conditions) and also the size of the local contributing area and land use and land management within that area (Burkart *et al.*, 2004; Tomer *et al.*, 2015; Wilkinson *et al.*, 2009). Where runoff is distributed via sheet or rill flow (i.e., not channelized), sloping, riparian wetlands along low order streams provide especially important sites for sediment retention, removing 80 to 90 percent of the gross erosion occurring on adjacent uplands (Brinson, 1993b; Lowrance *et al.*, 1997; Tomer *et al.*, 2003; Whigham *et al.*, 1988). The edge-of-wetland benefit also has been documented as a critical consideration to headwater (e.g., depressional) wetlands management (Cohen *et al.*, 2016), although retention rates are more variable, perhaps due to typically small (<100 km2) contributing areas and potential for more direct impacts from anthropogenic disturbance (Craft and Casey, 2000). Upland-wetland edges of floodplains also provide important sediment deposition zones (McClain *et al.*, 2003).

In addition to edge-of-wetland function, floodplain wetlands are widely recognized for the ability to capture sediment during flood events, specifically where overbank flow rates are slowed and surface water interacts with floodplain vegetation (Whigham *et al.*, 1988). Floodplains along

lower reaches of a river system provide key opportunities to capture nutrient-laden fine clay particles (Craft and Casey, 2000). For example, sediment deposition measurements in Coastal Plain floodplains indicated that these wetlands can capture 100% of associated annual river loads (Noe and Hupp, 2009). In contrast to the edge-of-wetland benefit, however, flood deposition occurs infrequently, only during high-magnitude storm events (Alexander *et al.*, 2015).

Advanced understanding of human impacts, especially due to changes in timing, rate, and chemistry of sources waters

Human alterations influence wetland water quality and habitat functions largely through effects on hydroperiod and water chemistry (Bedford and Preston, 1988). Resulting changes in the distribution of HGM types within a regional watershed or across physiographic provinces of the Chesapeake Bay undoubtedly has altered cumulative wetland functions and benefits significantly (Bedford, 1996; Brooks et al., 2014). For example, most streams and rivers in poorly drained areas of the Delmarva Peninsula have been channelized and, in many areas, drainage ditch construction extended entire stream networks by thousands of miles. As a result, many flats and depressional wetlands were drained to form what are referred to as prior-converted croplands w. Where ditching has lowered the watertable, it is common for the groundwater flow path to bypass the natural riparian wetlands, swamps, and tidal marshes where processing of nutrients and trapping of sediment occurs (Bricker et al. 2003). In the Piedmont, the long history of intensive agriculture and timber harvest caused extensive watershed erosion, which resulted in burial of many floodplain wetlands and the formation of incised streams which currently provide major sources of sediment to downstream locations (Donovan *et al.*, 2015). The steep relief and limited extent of navigable waterways historically limited human impacts to wetlands in the Appalachian Ridge and Valley Region and also the Appalachian Plateau. Wetland loss occurred mainly along river main stems, where development often occurs within river floodplains. Across the Bay watershed, expanding impervious surface area, channelization, and general watershed hardening has increased surface water runoff and reduced groundwater recharge, resulting in more significant flooding, altered hydroperiods and shifts in sediment loads throughout entire river corridors (Brooks and Wardrop, 2014; Hupp *et al.*, 2013; Strayer *et al.*, 2003). Compared to physical alterations imposed by human land use, less attention has been focused upon effects of shifting water chemistry. For example, increased nitrate loads ultimately can enhance P availability, especially where pyrite-rich geologic deposits can influence near-surface iron-sulfate-phosphorus chemistry (Smolders *et al.*, 2010). The human impacts to wetlands, however, provide key opportunities for targeted wetland restoration.

Remote sensing capabilities and advances in spatial modeling provide enhanced understanding of near-surface processes in relation to physiographic setting

Remote sensing capabilities and advances in spatial modeling in recent years have provided a better understanding of near-surface processes with respect to the potential for nutrient processing by wetlands. High resolution elevation data made available through LiDAR has been especially important to understanding surface flow and potential areas of interception and infiltration of water containing nutrients in extremely flat areas commonly associated with wetlands. This type of data will be especially useful for understanding phosphorus as most P transport takes place over the land surface. For nitrogen, there is still a need to include subsurface transport pathways as that is the main pathway for nitrogen transport. Combining LiDAR –derived elevation data with data on aquifer configuration can be used to understand potential subsurface flow pathways.

There has been limited research on the efficiency of wetlands to treat nonpoint source nutrients, such as from agriculture, within the Chesapeake Bay watershed (Goldman and Needleman, 2015). The ratio of wetland to watershed area has been used as a surrogate for hydrologic retention time (Simpson and Weammert, 2009), but this approach does not consider site-specific conditions that affect N removal and only weakly fits the data used to develop the model (Goldman and Needleman, 2015). New regional models that include a broader suite of factors that may influence nutrient transport and transformation are needed.   Monitoring targeted to supply needed data for model development will be important to the success of improved models.

Regional differences in surface and subsurface processes affecting nitrogen transport in the environment, including wetland interception, have been generally defined in the Chesapeake Bay watershed in the context of explanation of processes in different hydrogeomorphic or hydrogeologic settings. The Chesapeake Bay watershed was divided into simplified hydrogeomorphic regions by Bachman, et al, 1998. These regions work well for understanding general processes in the hard-rock regions above the Fall-Line. In the Coastal Plain, however, further work has refined understanding, especially with respect to subsurface processing of nitrogen (Ator et al, 2005; Krantz and Powars, 2000). Digital datasets are available to incorporate these interpretations on a regional basis for use with other pertinent data sets such as digital elevation models, soil characteristics, and land use and wetland maps.

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Chapter 5. Recommendations for Wetlands as land-use and BMPs in Phase 6 Watershed Model

Wetland land uses in the Phase 6 CBWM

[Editor's Note: this section has been removed for purposes of this document]

Wetland BMPs

Review of existing Phase 5.3.2 wetland restoration BMP

The CBP Scientific and Technical Advisory Committee (STAC) and the Mid-Atlantic Water Program have previously attempted to evaluate the effectiveness of wetlands as a BMP. During the April 2007 STAC workshop on quantifying the role of wetlands in achieving nutrient and sediment reductions, a first order kinetic equation was proposed to describe the exponential decline of nutrient and sediment over time related to detention time of runoff in a wetland. The kinetic equation was originally developed by Dr. Tom Jordan from the Smithsonian Environmental Research Center (SERC) and provided in both the STAC Report *Quantifying Role of Wetlands in Achieving Nutrient and Sediment Reductions in Chesapeake Bay* and the 2009 *Developing Nitrogen, Phosphorus and Sediment Reduction Efficiencies for Tributary Strategy Practices BMP Assessment: Final Report* by the Mid-Atlantic Water Program at the University of Maryland. The Mid-Atlantic Water Program was tasked with defining BMPs and determining effectiveness estimates that are representative of the overall Bay watershed.

Data have shown that longer detention times improve the nutrient removal efficiency of wetlands (Simpson and Weammert 2009). The kinetic equation assumes that wetland detention time is proportional to the ratio of the area of wetland to the area of the watershed. First order kinetics also describe, generally, the finding that the rate of removal is proportional to the concentration, making first order kinetics a practical way to express efficiency as a percentage of the inflow pollutant removed by the wetland.

The first order kinetic equation was developed to represent the removal efficiency of restored wetlands, based on the assumptions that:

- removal is an exponential function of detention time;
- detention time is proportional to the proportion of the watershed that is wetland; and
- there is zero removal when there is no wetland in the watershed

Nonlinear regression was used to fit the model to the removal data in the literature. This yielded the equation:

$$\text{Removal} = 1 - e^{-k(area)}$$

Where:

- Removal: proportion of the input removed by the wetland
- Area: proportion of the watershed area the is wetlands
- k: fitted parameter

    - TN, k=7.90, 95% confidence limits [4.56, 11.2]

    - TP, k=16.4, 95% confidence limits [8.74, 24.0].
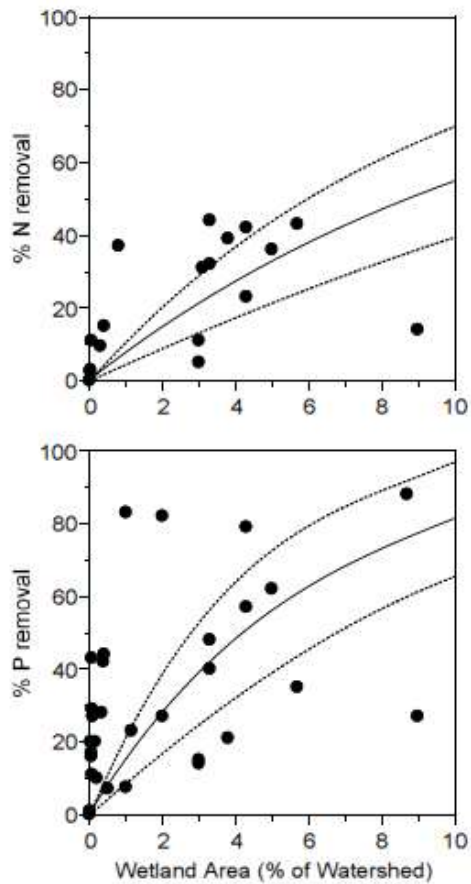
Figure 1. Literature review data points for wetland nutrient removal efficiency based on the wetland area as a proportion of the watershed. Curves indicate non-linear regression fit to data values, with 95% confidence limits. (STAC 2008).

The kinetic equation was developed for wetlands as a BMP (wetlands restoration) in Phase 5.3.2 model scenarios. To use the equation the ratio of wetland area to watershed area must be defined for each BMP reported by a jurisdiction. If this information was not reported by a jurisdiction, alternative calculations for the geomorphic regions were developed (Simpson and Weammert 2009). The alternative calculations assumed wetlands to be 1, 2, and 4 percent of the watersheds in the Appalachian, Piedmont and Valley, and Coastal Plain geomorphic provinces, respectively. The resulting TN and TP removal efficiencies are described in Table 1. If a jurisdiction does not report the geomorphic region of a wetland restoration, a uniform 16.75 percent and 32.18 percent, for TN and TP, respectively are applied.

Table 1. TN and TP removal efficiencies for wetlands by geomorphic province (Simpson and Weammert 2009).

Geomorphic Province	TN Removal Efficiency	TP Removal Efficiency	TSS Removal Efficiency
Appalachian	7%	12%	4%
Piedmont and Valley	14%	26%	8%
Coastal Plain	25%	50%	15%
Default, if HGM unknown	16.75%	32.18%	9.82%

One of the shortcomings of the kinetic equation is that it cannot account for wetlands that are sources of nutrients. Negative removal values (nutrient export) cannot be derived from this equation. During the literature review for development of the equation, any wetlands where only negative removal values were observed were removed from the calculations. When negative removal occurred in particular years, but not on the average, Simpson and Weammert used the average removal percentage in fitting their simple model. In cases where only negative removal was observed the observation was omitted, i.e. for one negative TP removal for one wetland studied by Kovacic et al (2000) and negative TN removal by one of the wetlands studied by Koskiaho et al (2003).

Due to the lack of data, the relationship between total suspended sediment and wetland area was not determined. A uniform 15 percent removal was approved, based on the average annual removal rates that were available in the literature, plus a margin of safety. This 15% removal was then applied to the region with the highest removal rates (Coastal Plain) and adjusted proportionally to the TP removal for the other two HGM regions.

The kinetic equation is unable to account for variations in wetland age, seasonal variation, spatial and temporal variability of flow, landscape position, or type of wetland. These factors will affect the residence time and loadings to a wetland. Craft and Schubauer-Berigan found that floodplain wetlands removed 3 times the nutrients of depressional wetlands on an areal basis (in Simpson and Weammert 2009). Nicholas and Higgins found that phosphorus removal declined significantly after about 4 years (in Simpson and Weammert 2009). Declining phosphorus removal rates over time also are not accounted for in the kinetic equation.

The BMP Assessment recommended future refinements to account for seasonal variability, nutrient discharge, hydraulic loading rate, wetland aging, and potential for dissolved P discharge during anaerobic conditions from wetlands with high phosphorus content (Simpson and Weammert 2009).

Recommended effectiveness estimates for wetland restoration (re-establishment) in Phase 6

Nontidal wetland re-establishment for Phase 6 Watershed Model

The panel evaluated TN, TP, and TSS reduction rates that may be applied for wetland restoration BMPs appropriate to natural wetland functions. Initially, the panel worked with Tetra Tech to develop summary of peer-reviewed reduction rates (Table 2).

Table 2. Summary of wetland TN, TP and Sediment reductions from literature review.

Wetland Type	Vegetation Type	TN % Reduction Mean Range Median (#)	TP % Reduction	TSS % Reduction
Headwater/ Depressional	ALL	33% -8-97 34% (9)	25% -15-94 10% (13)	28% -30-75% 37% (6)
Floodplain	ALL	44% -8-94 38% (24)	37% -41-100 29% (24)	32% -15-95 14% (7)
Constructed	Emergent (plus mixed, other and unknown)	32% 11-52% 33% (12)	38% -54-97% 35% (31)	92% 88-98 91% (4)
All except constructed	Forest, mixed and unknown	47% -8-97 59% (16)	45% -47-100 43% (44)	37% -15-95 32% (8)
All except constructed	Emergent	39% -8-89 36% (20)	31% -15-100 30% (20)	25% -30-75 27% (7)
Chesapeake Bay Only	All	22% -8-89 10% (10)	20% -41-81 17% (10)	24% -15-68 21% (8)
All except constructed	ALL	42% -8-97 39% (36)	40% -47-100 41% (64)	31% -30-95 27% (15)

A more detailed review of the studies summarized in Table 2 is provided in Appendix A. The range of values reflects the unique role of wetlands and current understanding of the importance of landscape, hydrology, soils, vegetation and thus the need to map this land cover class explicitly. The WEP agreed that wetland nutrient reduction function strongly depends upon landscape setting, including physiopgraphic setting and watershed position. The panel concluded that the mean value for all wetlands, exclusive of constructed wetlands, offered the most reasonable values for nitrogen, phosphorus and sediment reductions associated with treatment of upslope acres for re-established wetlands. These are the recommended effectiveness values for wetland re-establishment in the Phase 6 Watershed Model, to replace the current Phase 5.3.2 values described in the previous section.

These efficiency rates represent a greater TN, TP and TSS benefit compared to the Phase 5.3.2 default wetland restoration BMP efficiency summarized in Table 7. However, the panel did

consider how the re-established wetlands may function differently in different contexts throughout the watershed. By taking this approach the panel was able to apply the same general logic provided through Jordan's kinetic equation (e.g., longer retention time suggests greater nutrient and sediment reduction) but apply it in a framework that is based on physiographic sub regions instead of the curves illustrated in Figure 2. The mean efficiency values that the panel obtained through its literature review (42% TN and 40% TP) fall within the range of curves illustrated in Figure 2. These mean efficiencies are based on 36 values for TN and 64 values for TP. The 31% mean value for TSS is based on 15 values in the literature. The panel feels that these mean values from the literature are a reasonable replacement for the kinetic equation used for the Phase 5 wetland restoration BMP and that they can more reasonably be applied in Phase 6 for restoration projects within the Chesapeake Bay region but the panel felt they should be adjusted to reflect wetland characteristics in the different physiographic subregions. Effectiveness values reported in the literature rarely provided information about the physiographic region that could be used to help distinguish the effectiveness values by region. Given this limitation the panel felt it could better distinguish the effectiveness by the default ratio of upland acres treated by the restored wetland. The framework used to guide this decision based on the expected water quality function of wetlands by physiographic region is described in the following section.

Description of wetland water quality function based on form and location
[Editor's note: the remainder of the chapter is still under development; the text and information below will likely change more than sections above]

The panel discussed at length how to distinguish the effectiveness of wetland restoration with available information. The framework that they agreed to was built on their cumulative understanding and best professional judgment of the literature, and wetland restoration within the Chesapeake Bay region. The panel determined that the variability of results reported in the literature did not allow for a reasonable distinction in the efficiency rate itself. However, there was consensus that the current Phase 5.3.2 assumption where one upland acre is treated per acre of restored wetland was extremely conservative and needed to be updated for Phase 6. The panel therefore decided it could distinguish the effects of wetland restoration by assessing the relevant characteristics of physiographic regions in the watershed. This approach would be qualitative and not provide a high level of precision in the resulting recommendations, but it was agreed that the results would be an improvement over the current Phase 5.3.2 methods that apply the same efficiency to the same ratio of upland acres per acre restored (1-to-1), regardless of landscape position or physiographic region. The original intent of the Simpson and Weammert (2009) recommendations and Jordan's kinetic equation was to apply a different efficiency for each of the three hydrogeomorphic regions based on a reported ratio of the restored area and the drainage area, but this approach was not implementable at the time so the default efficiency rate and the 1-to-1 acre assumption were adopted. With this in mind the panel is confident that its recommendations represent a positive step towards a more accurate representation of wetland restoration in the CBP partnership modeling tools.

Chapter 4 describes the panel's understanding of the physiographic regions and the characteristics wetlands in those regions. That overview, when combined with the landscape position of wetlands, provide a framework for understanding the ability of a restored wetland to retain and remove nutrients or sediment that would otherwise continue downstream towards the Bay.

[…]

Table X below summarizes the distribution of natural wetlands by physiographic region and landscape setting, as Chapter 4 describes in more detail…

Table X. Natural Wetland Distributions and characteristics

Physiographic Province	Other Wetlands			Floodplain Wetlands
	Flats	Depressional Wetlands	Sloping Wetlands	
Appalachian Plateau		-moraine depressions	- Aquifer outcrops - Small tributary riparia	-valley floors, above bedrock outcrops
Appalachian Ridge & Valley		- Aquifer outcrops - Fractured rock springs	- Small tributary riparia - Slope breaks	- Medium to large waterways
Blue Ridge		- Ridgetops	- Fractured bedrock outcrops - Riparia	- Tributary confluences - Medium to large waterways
Piedmont			- Fractured bedrock outcrops - riparia	Eroded stream/river terraces
Inner Coastal Plain			- Small streams, floodplain edges	Small to large waterways

Outer Coastal Plain - Poorly drained uplands	Watershed divides	Watershed divides	- Small (natural and artificial) tributary riparia	Small to large waterways
Outer Coastal Plain - - Well drained uplands			- Small tributary riparia	Small to large waterways
Coastal Plain Lowlands	Watershed divides		- Small (natural and artificial) tributary riparia	Bottom lands

Karst terrain
  - Appalachian Plateau
  - Appalachian Ridge & Valley
  - Piedmont
		Tubular springs	Outcrops, slope breaks, springs	

Table X summarizes the key points from Chapter 4, describing the likely potential for a restored wetland to make contact with water flows that are contaminated with nutrients or sediment. As illustrated in the table, a higher likelihood of hydrologic contact increases the opportunity for the wetland to remove nutrients or sediment through the processes explained in Chapter 4. The combination of wetland distribution and likelihood of hydrologic contact provided the panel with a basis for estimating ratios of upland acres that should be treated by the wetland restoration BMP in Phase 6…

Table X. Likelihood of Hydrologic Contact with Non-Point Source Contaminated Waters

Physiographic Province	Other Wetlands			Floodplain Wetlands
	Flats	Depressional Wetlands	Sloping Wetlands	
Appalachian Plateau		L – variability in hydrologic settings & predominant forest cover	L – confined aquifer discharge not likely contaminated	L - predominant forest cover and greater likelihood of hyporheic exchange rather than wetland discharge
Appalachian Ridge & Valley		L – small contributing area; predominant forest cover	L – confined aquifer discharge not likely contaminated; predominant forest cover	L - predominant forest cover and greater likelihood of hyporheic exchange rather than wetland discharge
Blue Ridge		L – small contributing area; predominant forest cover	H - Surficial aquifer and heavy human impacts	M – Incised, more infrequent events; potential deep aquifer by-pass
Piedmont			M - Surficial aquifer and heavy human impacts	M – Incised, more infrequent events; potential deep aquifer by-pass
Inner Coastal Plain			H - Surficial aquifer and heavy human impacts	H – well connected, more frequently flooded

Outer Coastal Plain - Poorly drained uplands	L – small contributing area; flat hydraulic gradient predominant forest cover	L – small contributing area; flat hydraulic gradient predominant forest cover	M – Small contributing area, but surficial aquifer important and heavily influenced by human impacts	M – well connected, frequently flooded but potentially limited exchange due to flat hydraulic gradients
Outer Coastal Plain - - Well drained uplands			L – Deep aquifers with strong potential to bypass contaminated waters	H – well connected, more frequently flooded
Coastal Plain Lowlands	L – small contributing area; flat hydraulic gradient predominant forest cover		H – well connected, more frequently flooded	M – well connected, frequently flooded but potentially limited exchange due to flat hydraulic gradients
Karst terrain* - Appalachian Plateau - Appalachian Ridge & Valley - Blue Ridge & Valley		H – Strong potential for contaminated discharge.	M – Strong potential for contaminated discharge, but potential for rapid flow-through & short contact time	L/M – see floodplain descriptions above, respectively

[…]

Wetland water quality function as basis for estimating effectiveness of nontidal wetland restoration as a BMP in the Phase 6 Watershed Model
[…]

Table 3. Summary of relative retention efficiencies and upland acres treated by each acre of wetland by wetland type and physiographic subregion.

Physiographic Subregion	Retention Efficiency			Upland Acres Treated	
	TN	**TP**	**TSS**	**Floodplain Wetlands**	**Other Wetlands**
Appalachian Plateau	42	40	31	2	1
Appalachian Ridge and Valley	42	40	31	2	1
Blue Ridge	42	40	31	3	2
Piedmont	42	40	31	3	2
Inner Coastal Plain	42	40	31	6	4
Outer Coastal Plain- Poorly Drained	42	40	31	2	1
Outer Coastal Plain- Well Drained	42	40	31	3	2
Coastal Plain Lowland	42	40	31	3	2
Karst Terrain	42	40	31	3	2

*Other wetlands with low treatment potential due to small contributing area predominated by forest and/or strong potential for contaminated water to by-pass the wetlands: 1 ACRE
**Other wetlands with high treatment potential, located in heavily impacted watersheds and having strong likelihood for hydrologic contact: 4 ACRES
***All other wetlands: 2 ACRES
****Floodplain wetlands with additional overbank delivery: 150% of Other

Wetland restoration (re-establishment) in tidal areas

In the Phase 6 model, Tidal wetlands will be simulated in the estuarine model, not the Watershed Model. This means no tidal wetland land use acres to which a tidal wetland restoration BMP can be applied. Given this context and the protocols developed by the Shoreline Management Expert panel already approved, this panel briefly reviewed that effort for relevance to the charge to develop wetland BMPs. Specifically, the panel considered Protocols 2, 3 and 4 as defined by that expert panel.

- Protocol 2: Denitrification
- Protocol 3: Sedimentation
- Protocol 4: Marsh Redfield Ratio

The panel concluded that the Shoreline Management Panel's Protocols 2-4 adequately characterize the relevant nutrient and sediment processes of tidal wetlands. It was noted that no new literature has been published since 2015 that would affect or change the load reductions recommended by the Shoreline Management panel. It is recommended that the sum of these

protocols be used as a load reduction effectiveness estimate for tidal wetland restoration BMP in the Phase 6 modeling tools. The recommended load reduction is summarized in Table 4 below.

Table 4. Summary of Shoreline Management BMP load reductions, Protocols 2-4.

Shoreline Management Protocol		TN	TP	Sediment
Protocol 2 – Denitrification	Acres of re-vegetation	85	NA	NA
Protocol 3 - Sedimentation	Acres of re-vegetation	NA	5.289	6,959
Protocol 4 – Marsh Redfield Ratio	Acres of re-vegetation	6.83	0.3	NA
Tidal wetland restoration		91.83 lbs/ac	5.589 lbs/ac	6,959 lbs/ac

Recommendations for wetland creation (establishment), wetland enhancement and wetland rehabilitation

This panel was unable to determine a recommended benefit for these BMPs in the time available but strongly encourages the partnership to quickly convene another expert panel to evaluate the effectiveness of these categories of wetland BMPs. The suggested definitions and framework for these BMPs are already provided as a starting point for the future expert panel, which should be convened as a high priority under the WQGIT's BMP Protocol. Unlike wetland restoration and wetland creation, the enhancement and rehabilitation BMPs represent gains in function only, not gains in acres. As such, these BMPs would likely be credited as effectiveness estimates applied to nontidal wetland land use acres in the Phase 6 modeling tools and not represented as a land use change. The Wetland Creation BMP, similar to Wetland Restoration, would be expected to be a land use change plus treatment to upland acres. However, the effectiveness estimate applied to the upland acres for Wetland Creation should not be assumed to be equal to the estimate provided by this Panel for Wetland Restoration.

If the future panel is instructed to consider these BMPs for application to tidal areas, the recommended protocols for the tidal BMPs would likely need to reflect the fact that there are no land use acres for tidal wetlands as they are simulated through the Estuarine Model, not the Watershed Model.

Following approval of this report and the wetland restoration BMPs, the Wetland Workgroup and Habitat GIT should work with the Water Quality GIT to promptly form an ad hoc group to craft the charge and scope for a new expert panel to evaluate the effectiveness wetland enhancement and wetland rehabilitation BMPs to reduce nitrogen, phosphorus and sediment loads. The future panel should build and clarify on the recommended definitions of this panel, but is asked to maintain the broader category definitions described in **Error! Reference source not found.**.

The future panel may consider using the same distinction for the BMPs according to physiographic region (Coastal Plain, Piedmont, etc.) and land use (Floodplain and Other), or it

may decide that a condensed or simpler approach is appropriate for the functional gain BMPs or Wetland Creation.

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