Robert P. Brooks
Denice Heller Wardrop *Editors*

Mid-Atlantic Freshwater Wetlands: Advances in Wetlands Science, Management, Policy, and Practice

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With heartfelt love and affection for Becky, my wife, who encouraged me and persevered through the many years devoted to creating this body of work, and to our beloved children, Benjamin and Emily, who contributed in many ways, large and small, along my chosen path studying the natural world.

With love, Rob

If the family were a boat, it would be a canoe that makes no progress unless everyone paddles.

Letty Cottin Pogrebin

For paddling, each in your own wonderfully unique way, and to unknown and fantastic destinations that required trust and affection, I thank my children, Morgan and Tom, my brother Barry, and my Mom and Dad. With my complete devotion and gratitude I thank my husband Rick, who is my North Star and navigator. I also thank Rob, who gave me the opportunity to go on the voyage in the first place.

With love, Denice

Foreword

> … rivers and principal creeks rise in the high country … or in the ridges continuous therewith. Flowing between the groups of hills, and forking at frequent intervals, they run swiftly in a general southeast direction until the wider valleys are reached, and then they stretch more broadly onward to empty into the estuaries … (Scharf 1881, p.14).

The lands and waters of the Mid-Atlantic Region (MAR) have changed significantly since before the sixteenth century when the Susquehannock lived in the area. Much has changed since Captain John Smith penetrated the estuaries and rivers during the early seventeenth century; since the surveying of the Mason–Dixon Line to settle border disputes among Maryland, Pennsylvania, and Delaware during the middle of the eighteenth century; and since J. Thomas Scharf described the physiographic setting of Baltimore County in the late nineteenth century. As early as 1881, Scharf provides us with an assessment of the condition of the aquatic ecosystems of the region, albeit in narrative form, and already changes are taking place—the conversion of forests to fields, the founding of towns and cities, and the depletion of natural resources. We have always conducted our work with the premise that "man" is part of, and not apart from, this ecosystem and landscape. This premise, and the historical changes in our landscape, provides the foundation for our overarching research question: *how do human activities impact the functioning of aquatic ecosystems and the ecosystem services that they provide, and how can we optimize this relationship?*

For nearly two decades, Riparia, a Center at The Pennsylvania State University (originally known as the Penn State Cooperative Wetlands Center), and our collaborators have sought to understand the ecology of wetlands and the stressors that affect them by framing our work in a watershed or landscape context that views wetlands, not as isolated patches, but as part of an integrated system with aquatic and terrestrial components. In other words, we have focused much of our studies on how the surrounding landscape affects a wetland and its adjoining aquatic habitats. Through productive collaborations, we have begun to understand linkages at a number of spatial and temporal scales: between wetlands and their immediate surroundings, among wetland networks and their watersheds, and between freshwater and estuarine systems. This book concentrates on the first two of these linkages, since it

Fig. 1 Wetland depression in northcentral Pennsylvania (Photograph by R. Brooks)

Fig. 2 Wetland depression, beaver impounded in northcentral Pennsylvania (Photograph by R. Brooks)

is, quite literally, our home and what we know best. To leverage the knowledge that we have been fortunate to gain, it is not just about the science of these fascinating and valuable places, but also how to use science to inform policy and practice.

This book focuses on small watersheds and their component parts, and begins by introducing the elements of aquatic landscapes in Chap. 1. Chapter 2 presents a particularly effective way to describe ecological equivalence between various wetland types (hydrogeomorphic (HGM) classification) and documents the establishment of a reference collection of wetlands. These sites are the basis of much of the investigative work presented in other chapters. Chapter 3 discusses the basis of the disturbance gradient (also utilized for many of the investigations in the book) and examines the relationships between land cover changes and structural components of the wetland itself via a case study of eight wetland sites. Chapters 4 and 5 explore two of these physical characteristics, hydrology and soils, in-depth across the HGM classification and reference sites described in Chap. 2, using the disturbance gradient of Chap. 3. Chapters 6, 7, 8, 9, and 10 individually discuss the biological components of hydrophytes, wildlife, birds, amphibians and reptiles, and macroinvertebrates, respectively, across these same gradients of wetland type and anthropogenic disturbance. Chapter 10 is notable in its presentation of conceptual models of riparian structural response to land cover change, with the accompanying shifts in macroinvertebrate community structure, and illustrates the value of monitoring information in formulating management actions. Chapter 11 takes a close look at the origins of monitoring and assessment approaches, the role of regional forums in advancing them, and recounts the stories of their use and application at scales from individual sites to the National Wetland Condition Assessment. The use of monitoring and assessment information specifically in restoration and mitigation is explored in Chap. 12, and Chap. 13 examines the policy and regulatory arena in which monitoring and assessment information is utilized. Chapter 14 carries our understanding of wetlands in a connected aquatic landscape one step further, by discussing concepts of connectivity and implications for conservation and management.

These chapters represent the accumulated knowledge of its authors, coauthors, and the many additional contributors to this body of work. Thus, we need to extend our appreciation to past and current students, Penn State colleagues, collaborators from universities, institutions, agencies, organizations, and companies with whom we have worked, and, certainly, the many landowners who provided access to wetlands and streams found on their lands. Funding from our research, outreach, and education activities that contributed to this book came from many sources—thanks to all of you—but we are especially thankful for continued support and cooperation from the US Environmental Protection Agency, the member states of the Mid-Atlantic Wetlands Work Group (MAWWG), especially Pennsylvania, and our own employer, The Pennsylvania State University.

We need to acknowledge two very special people who left us way too early to see the fruits of their contributions. The first is one whose vision guided and inspired threads of our holistic view, and that is Mark Brinson, formerly of East Carolina University. Mark tragically passed away in January 2011, but his influence remains

with us through his innovative hydrogeomorphic concepts and good nature, and including his contributions to Chaps. 1 and 2. The second is Art Spingarn, of USEPA Region 3, who also left us much too soon. His commitment to monitoring and assessment made the Upper Juniata and Nanticoke watershed studies happen. These first regional watershed assessments of wetland condition demonstrated that the wetland community, writ large, had the technical and programmatic knowledge to conduct assessments of wetland condition at large scales; regional and national scale assessment eventually followed.

Mid-Atlantic Freshwater Wetlands is being published by Springer at a time when publishing practices are changing and diversifying almost daily. To stay current and to provide readers with some value-added information and features, we have established a link on Riparia's website where supplemental information is and will be posted related to this book. We encourage you to visit http://www.riparia.psu.edu/MARbook on occasion where you will find such items as additional color photographs and maps, datasets, and tools that can assist you in learning about and investigating wetlands throughout the MAR region, and beyond.

Greetings from Riparia, Rob Brooks, and Denice Wardrop June 2012

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Chapter 1
Aquatic Landscapes: The Importance of Integrating Waters

Robert P. Brooks, Craig Snyder, and Mark M. Brinson†

Abstract The landscapes of the Mid-Atlantic Region are dissected by networks of rivers, with their associated wetlands and riparian areas. These systems provide important ecosystem services, both ecological functions and societal values, such as floodwater storage, public water supplies, recreational greenbelts, and habitats for a diversity of flora and fauna. Ecologists and hydrologists have increasingly focused on integrating across the four dimensions of these ecosystems: longitudinal, lateral, hyporheic, and time. Here, we use a hydrogeomorphic classification system to describe wetland types, and present a conceptual model of how they connect to other waters through critical components such as hydrologic connectivity, energy flows and sources, and biological integrity. The connectivity of aquatic habitats is described and related in a watershed context. Concepts are supported by a technical review of pertinent literature. We emphasize the flow of water, nutrients, and organisms from headwaters downstream through an interconnected riverine ecosystem.

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1.1 Introduction

The landscapes of the eastern USA, including the Mid-Atlantic Region (MAR), are dissected by networks of rivers, with their associated wetlands and riparian areas. These systems historically have provided important ecosystem services, both ecological functions and societal values, to the inhabitants, and continue to provide for floodwater storage, public water supplies, recreational greenbelts, and habitats for a diversity of flora and fauna. Yet 70–90% of the waterways of the eastern USA have been drastically altered by human activities, and significant amounts of the original wetlands of the eastern states, varying from 30 to 70% among the different states, have been lost (Dahl 1990).

Administratively, the MAR usually includes the states of Pennsylvania, Delaware, Maryland, Virginia, and West Virginia. We define the Mid-Atlantic with somewhat more porous boundaries and an emphasis on the drainage basins of major rivers, ranging in geographic extent east-to-west from the Atlantic beaches and marshes to the top of the Appalachian Mountains, and including the western slopes draining into the Ohio River Basin, approximately to the Pennsylvania–Ohio border. Then, proceeding north-to-south from the most northerly extent of Susquehanna and Delaware river basins in New York and New Jersey, and finally south to the northern portions of North Carolina that drain into the Ablemarle-Pamlico system; this is our region of study (Fig. 1.1).

The importance of the headwater portions of watersheds to the overall health of aquatic ecosystems in this region cannot be over emphasized. In the MAR, headwaters typically comprise about 67–75% of the contributing area of any given watershed. That is, the combined areas of terrestrial habitats, wetlands, floodplains, and headwater streams occupy two-thirds to three-quarters of the total area of the drainage basin for larger rivers (Fig. 1.2). Given this influence on downstream portions of large river watersheds, understanding the impacts of human activities on the ecological structure and function of headwater or tributary watersheds is foundational for optimizing their conservation and management. That said, it is also critically important to understand the linkages and contributions of headwaters to the downstream portions of watersheds where large rivers and broader floodplains dominate.

Increasingly, during the past decade, ecologists and hydrologists have moved toward integrating the traditional studies of the upstream–downstream gradient of rivers (i.e., longitudinal or first dimension, river channel, streambanks) with lateral or second dimension (e.g., floodplains, riparian corridors, wetlands) and vertical or third dimension (e.g., groundwater flows, hyporheic zone) portions to represent a more comprehensive view of all the aquatic components of watersheds, and their interactions with terrestrial areas (e.g., Ward et al. 2002; Naiman et al. 2005) (Fig. 1.3). The separation of constituent aquatic components of watersheds is common in regulatory and management contexts where streams and rivers are treated as separate entities from lakes, wetlands, and estuaries. In fact, such a separation was more for the convenience of defining and managing these units, than it was based on any ecological principles. In this chapter, we discuss the important hydrological and ecological linkages between streams, rivers, and their adjacent

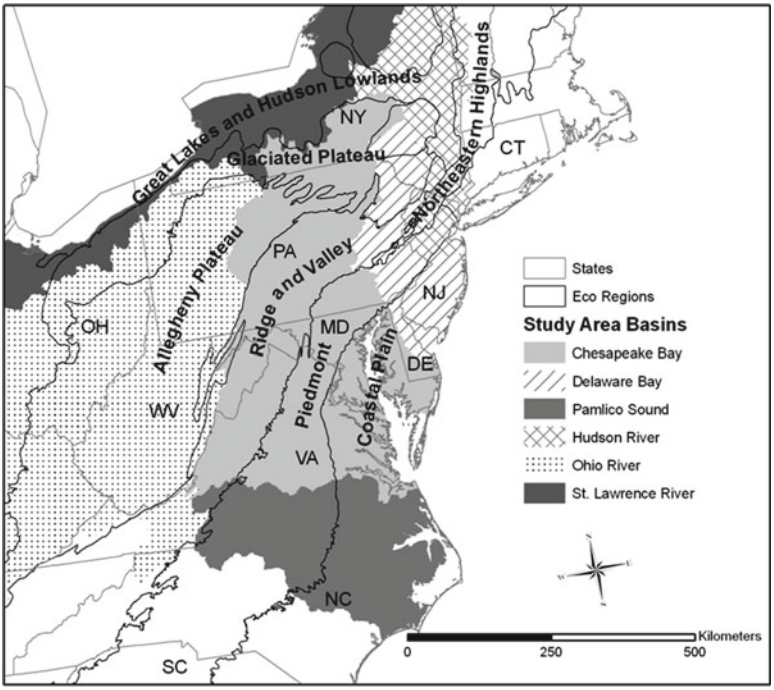

Fig. 1.1 Extended study area of the Mid-Atlantic Region (MAR) showing boundaries of states, ecoregions, and major river basins

floodplains and wetlands, together, comprising a riverine ecosystem. When the interactions with terrestrial components are considered, a holistic aquatic landscape can be visualized.

Historically, rivers have been variously portrayed as continuous linear (upstream–downstream) gradients (e.g., the River Continuum Concept (RCC), Vannote et al. 1980; Minshall et al. 1985), or as a series of distinct, interconnected habitat patches (e.g., Link Discontinuity Concept, Rice et al. 2001). In addition, alternative conceptual models have evolved seeking to classify stream networks (Frissell et al. 1986; Rosgen 1994), explain the physical heterogeneity of rivers (natural flow regimes, Poff et al. 1997; river discontinua, Poole 2002; network dynamics hypothesis, Benda et al. 2004), describe material cycling (riverine productivity model, Thorp and DeLong 1994, 2002; nutrient spiraling, Newbold et al. 1982; process domains, Montgomery 1999), and perturbations (intermediate disturbance hypothesis applied to rivers, Townsend et al. 1997). Of particular value to this discussion focused on wetlands are the ideas that characterize riverine ecosystems as a series of interconnected hydrogeomorphic (HGM) patches (Church 2002; Poole 2002; Thorp et al. 2006) and the relationship of these dynamic patches to aquatic biodiversity (Townsend et al. 1997; Lake 2000; Ward et al. 2002; Thorp et al. 2006).

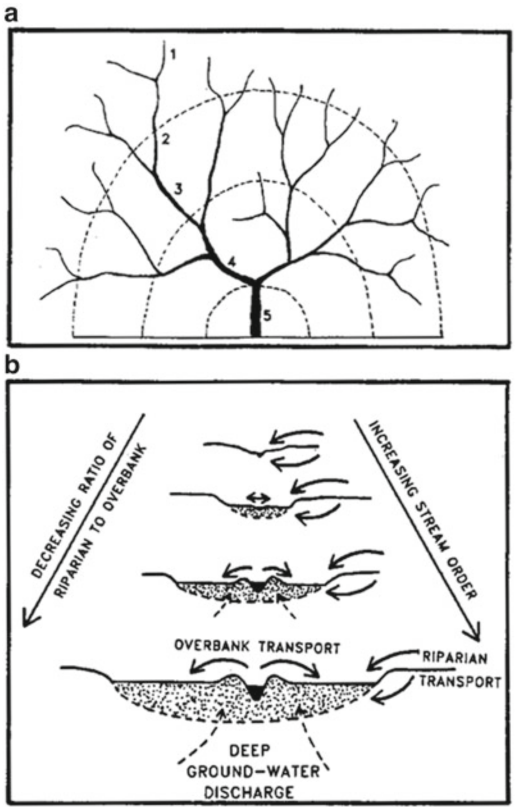

Fig. 1.2 Representations of the relative contributions of stream order to (**a**) watershed area and (**b**) flooding and discharge

Increasingly, these syntheses have begun to move beyond the stream or river channel alone, to incorporating linkages between streams and the landscape in which they flow, thus recognizing longitudinal, lateral, and vertical aspects of the riverine network (e.g., Forman 1995; Ward et al. 2002; Wiens 2002; Naiman et al. 2005). Still missing, however, are attempts to create conceptual models that directly integrate stream, wetland, riparian, and terrestrial components for headwater, tributary, and mainstem portions of watersheds. Within biogeographical constraints, species composition and biological integrity of watersheds are the result of interactions among numerous important instream variables including flow regime, energy source, water quality, instream habitat, and biological interactions. Yet, these variables themselves are largely driven by processes that occur outside of the individual stream channel including weather and climate, geomorphology of the watershed (geology and terrain), and the structure and topology of the surrounding landscape (Karr 1991, 1999). As one moves downstream, instream characteristics are determined by characteristics and processes that occur in upstream areas. Hychka (2010), while exploring the hydrologic responses of wetlands, found that a riverine reach was the spatial unit most appropriate for characterizing the interaction of the stream with the adjacent floodplain and wetlands (Fig. 1.4). It is the magnitude and

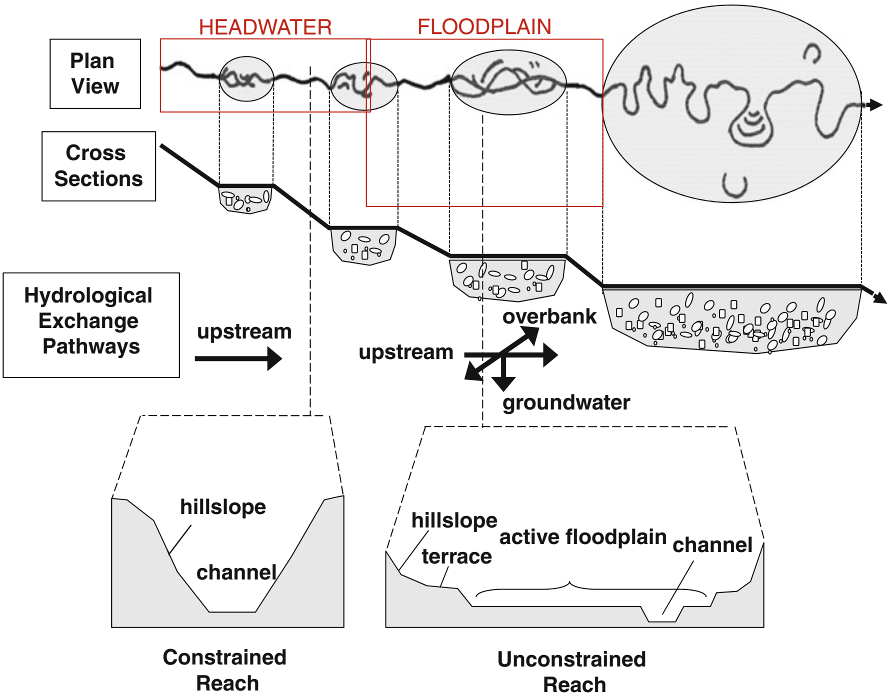

Fig. 1.3 Longitudinal, vertical and lateral dimensions of riverine ecosystems incorporating riverine, wetland, and riparian components (modified from Ward (2002) by S. Yetter)

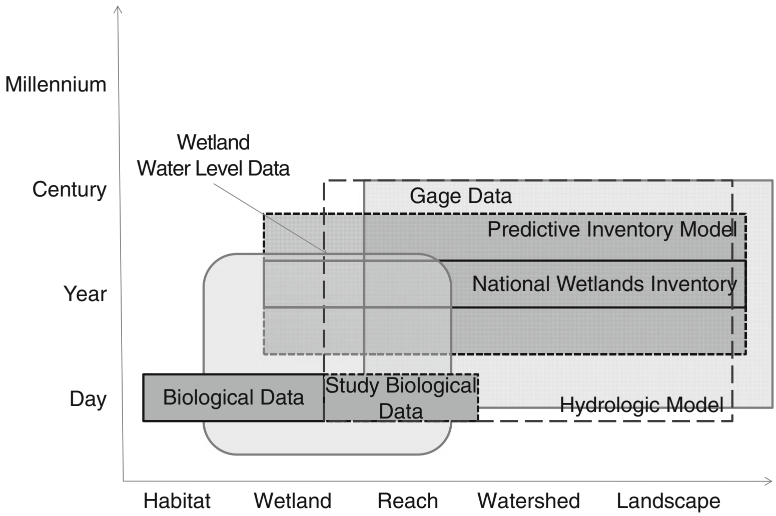

Fig. 1.4 Spatial and temporal scales of relevance for studying wetlands at a reach scale (from Hychka 2010)

interplay among these longitudinal, lateral, and vertical processes that form the basis for most conceptual models integrating riverine ecosystems into aquatic landscapes. In this chapter, we will consider these concepts in light of the characteristics of riverine ecosystems that connect to and support freshwater wetlands found in the MAR.

1.2 Types of Freshwater Wetlands in the Mid-Atlantic Region

During our studies of MAR wetlands, we have found that starting classification with HGM characteristics (i.e., geomorphic setting, water source, hydrodynamics; Brinson 1993a), combined with portions of the hierarchical classification system used for the National Wetlands Inventory (NWI; Cowardin et al. 1979; Tiner 2003) aids in discriminating among wetland types, particularly during field investigations as opposed to using NWI just for mapping. Early attempts to classify wetlands Cowardin et al. (1979) lumped most vegetated wetlands into a single broadly defined Palustrine system. Using an HGM classification approach, we separate additional types based on their hydrologic characteristics and landscape positions. These HGM classes can be further subdivided by their dominant vegetative life forms; aquatic bed, emergent, shrub, and forest. The additional discrimination of wetland types associated with the HGM approach represents a substantial improvement in wetland classification because it incorporates features that influence wetland function and allows assessment of condition. Our approach to classification and inventory is detailed in Chap. 2 of this book, although terms are introduced here for purposes of introducing concepts and terminology.

The majority (>80%) of freshwater wetlands in the MAR are *riverine* types, associated with streams and rivers (Brooks et al. 2011a). Some of these riverine wetlands occur as narrow terraces or vegetated islands within the defined channel banks. Most, however, are found in the adjacent floodplain. If a river is free-flowing and it's channel is not incised, then it interacts with the adjacent floodplain when discharge exceeds channel capacity. Major flood pulses move laterally away from the river channel into the floodplain, and in the process initiate a series of important biogeophysical processes (Junk et al. 1989; Sparks et al. 1990; Bayley 1995).

In headwater portions of a watershed, the most relevant HGM subclasses of wetlands are topographic slopes, depressions, and riverine upper perennial floodplains (which together compose riverine headwater complexes). Perennial and seasonal depressions, some of which are isolated or have subsurface connections, but no surface water connections, and stratigraphic slopes further uphill (e.g., springs and seeps) may contribute to base flow of headwaters. In the lower portions of a watershed, riverine lower perennial types, consisting primarily of the wetland portions of expansive floodplains dominate the system. In some rivers, instream wetlands occur within the defined channel itself, either as low terraces or vegetated islands. Throughout a given watershed, ponds, lakes, and reservoirs (lacustrine types), either naturally formed or impounded, may provide direct hydrologic support or even flow-through connections to streams and rivers.

During the flood pulse as water flows across the floodplain, sediments are deposited and nutrients have an opportunity to be transformed. Pulses of increased flow in the channel that do not overtop most of the riparian banks to inundate the floodplain occur at higher frequencies than overbank flooding events. These are important for replenishing or maintaining sufficient water levels in remnant channels and interconnected depressions. More aptly described as flood flows or minor flood pulses, these events supply water to communities of aquatic macroinvertebrates and wetland plants that occur in the low-lying sections of floodplains (see Chap. 10).

Following the lead of the NWI (Cowardin et al. 1979; Tiner 2003), we divide wetlands associated with riverine reaches of a watershed into upper perennial (i.e., wetlands along smaller tributaries typically with higher gradients; headwaters with or without floodplains) and lower perennial types (i.e., wetlands along rivers with well-developed floodplains, typically with lower gradients) (Brooks et al. 2011a).

Lakes are not common water bodies in the MAR, and, therefore, *lacustrine* (*or fringing*) wetlands are not abundant compared to the Great Lakes or New England regions. The greatest density of fringing wetlands occurs along natural and hydrologically altered lakes and ponds in the glaciated ecoregions of northeastern and northwestern Pennsylvania, northern New Jersey, and southern New York. Reservoirs, impounded for water supplies and/or flood control, increase the area of open water and fringing wetlands throughout the remainder of the MAR. Fringing wetlands are found along the edges of these lakes and reservoirs, usually at depths of <2 m, where light penetration is sufficient to promote the growth of rooted aquatic plants. Although most are comprised of emergent plant communities, shrubs and trees can dominate if suitable conditions prevail.

Flats, by definition, occur where water sources are dominated by precipitation and vertical fluctuations of the water table (Brinson 1993a, b). They typically occur in regions of low topographic relief, such as the coastal plains of Delaware, Maryland, Virginia, and North Carolina. There are other ecoregions of the MAR with landscapes that are relatively flat topographically, including parts of the Allegheny Plateau, and the glaciated regions (Fig. 1.1), but hydrologic regimes in these locales are based on a mix of surface flows and soils saturated with groundwater that are more like other wetland types than characteristic flats. Thus, these wetlands continue to be classified as a mixture of shallow depressions, low gradient slopes, and riverine floodplains, rather than flats.

We have identified two major types of *slope* wetlands (Stein et al. 2004; Brooks et al. 2011a). Topographic slopes, found at the toe of hillslopes, are expressed groundwater discharges. Stratigraphic slopes, located farther upslope, generally represent geologic discontinuities. Both types tend to have unidirectional flows of groundwater from shallow and deep origins.

Wetland *depressions* are quite diverse, being formed through a variety of geophysical processes, and varying in area, depth, and permanence. Isolated depressions, by definition, have no surface water connections to other water bodies. Due to their separation from navigable waters, isolated depressions have been a contentious topic in the wetlands regulatory arena. Depressions can occur anywhere in the landscape where a low-lying area collects and stores water in a shallow or deep, bowl-shaped feature.

1.3 Conceptual Model of the Structure and Functions of Riverine Ecosystems

A primary goal of this chapter is to synthesize the salient aspects of existing ecological theory as it relates to wetlands closely associated with rivers. Our synthesis attempts to move toward a holistic understanding of the ecology and management of what could be termed “riparia,” the areas of transition between water and land. We believe that it is useful to characterize watersheds as a mosaic of interconnected HGM patches (Ward and Stanford 1983; Townsend 1989; Ward 1989; Brinson 1993b) or settings that contain a set of functional process zones (FPZs, Thorp et al. 2006). FPZs consist not only of a distinguishable stream reach, but also include the geologic and topographic aspects of the surrounding terrestrial landscape. This can lead to considering streams, wetlands, and riparian areas as definable landscape units at a reach scale that support characteristic biota and that respond predictably to a set of anthropogenic stressors.

As the various elements of the system are discussed in subsequent sections, information about potential stressors is included to assist the reader in understanding their influence on ecological integrity (Bryce et al. 1999). The elements of the conceptual model, including anthropogenic drivers and stressors, are summarized in Fig. 1.5 (modified from Karr 1991; Karr and Chu 1999; Brooks et al. 2006a, b).

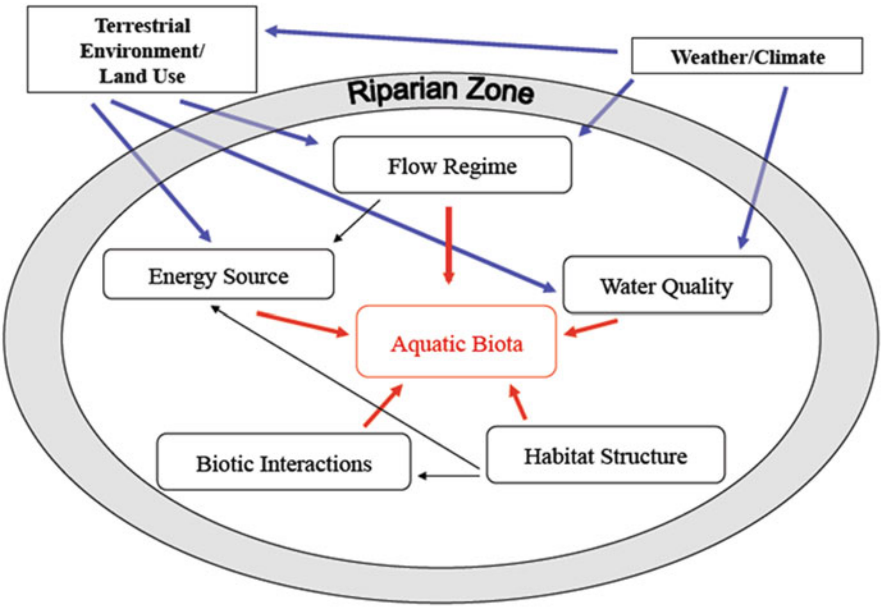

Fig. 1.5 Conceptual model of a riverine ecosystem and its elements (modified from Karr 1991, 1999; from Brooks et al. 2006a, b)

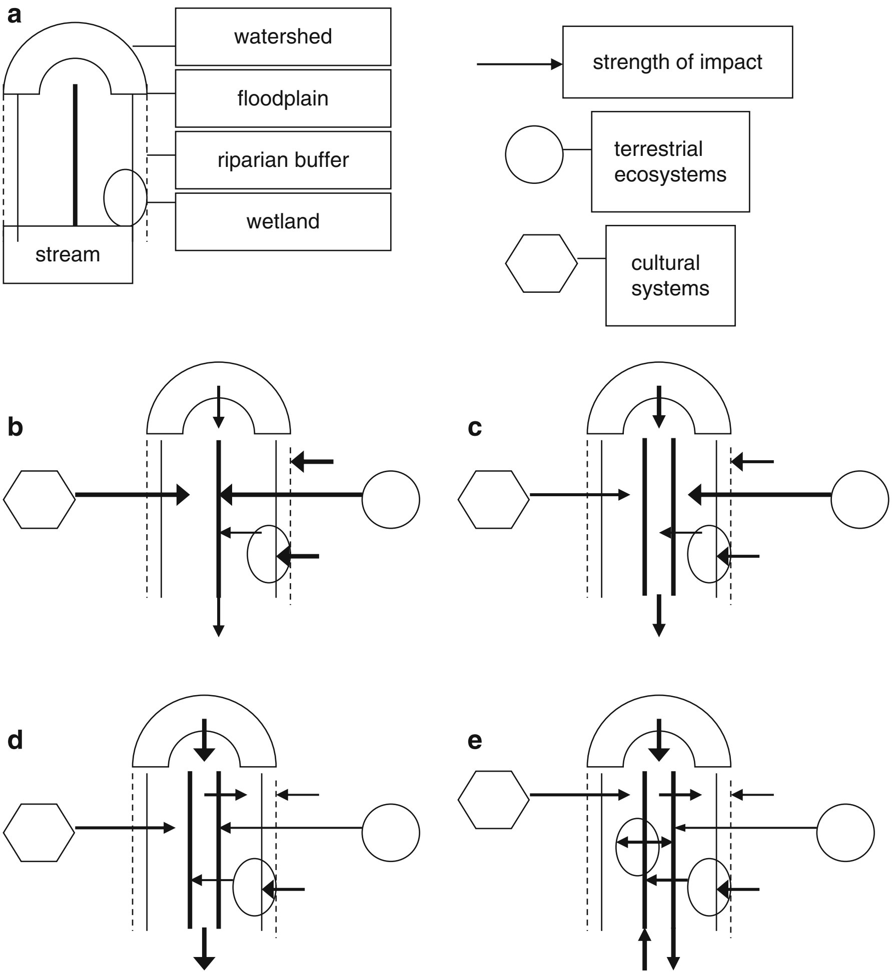

Fig. 1.6 (**a**–**e**) Conceptual model of elements in riverine ecosystems. (**a**) Legend. (**b**) Headwater stream (second order with floodplain). (**c**) Stream (third/fourth order with floodplain). (**d**) River (fifth order with floodplain). (**e**) Tidal River/Subestuary (with floodplain)

The structure and function of these ecosystems is considered under the following headings: 1.3.1, 1.3.2, 1.3.3, 1.3.4 and 1.3.5. For the purposes of this chapter, the hypothesized interactive relationships among the stream, wetland, riparian, and upland components of watersheds for different stream orders are illustrated in Fig. 1.6a–e. Key features of these illustrations are the relative contributions to the functioning of these systems by upstream portions of the watershed vs. immediately adjacent or lateral components.

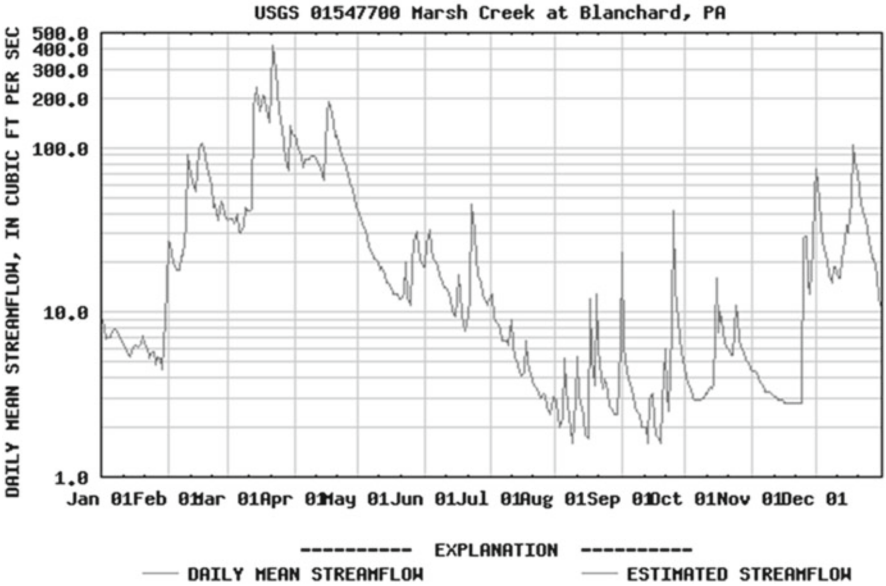

Fig. 1.7 Typical discharge curve for a river in central Pennsylvania showing higher peaks in winter and spring and lower flows in summer and autumn (Data from U.S. Geological Survey Station 01547700, prepared by S. Yetter)

1.3.1 Hydrologic Connectivity in Riverine Ecosystems

In the MAR, there is a surplus of precipitation relative to rates of evapotranspiration (i.e., P>ET, see Chap. 3 of this book), thus, many stream and river channels are perennial, receiving sufficient groundwater discharge to sustain base flow throughout the year. Headwater streams (defined here as Strahler (1952, 1957) first through third orders) often occur in proximity to depressions and slopes such that collectively they encompass the type we call a riverine headwater complex. The same phenomenon can contribute to the hydrologic regime of floodplains of larger rivers, referred to as lower perennial wetlands, or a riverine floodplain complex. In these HGM settings, the discharge of groundwater or underlying shallow water table may form a broader band of hydrologically similar conditions such that larger areas of forested wetlands or emergent marshes are formed.

Typically, sustained high flows occur in the late winter and early spring due to either snowmelt or spring rains that often come when water levels are already near peak (Fig. 1.7). There are, of course, exceptions, with singular events like severe thunderstorms or hurricanes precipitating torrential rains over short periods of hours or days. In either case, when flood levels are high, overbank flooding can occur rapidly or over sustained periods of time, and the kinetic energy of these lateral flows is correspondingly strong. This provides the power to erode floodplain soils, alter existing remnant oxbows, cut through sharp meanders of the current river channel, carve new channels, and create vertical banks. The resulting changes in surface flow patterns are influenced by existing floodplain topography (e.g., past channels, areas of dense vegetation, erosion-resistant rock or compact soils) and those interactions modify the location, shape, depth, and size of wetlands within the floodplain. Dams or bank burrows created by beaver, debris piles, or large tree windrows can all influence the directional pathways and force of water flows across the floodplain.

Higher elevation areas, such as portions of the natural levee, distal upland edges, and upland inclusions within the floodplain tend to divert the flows and resist erosive forces. Low-lying depressions, narrow berms, and old and new channels are more likely to shift their shapes and locations dynamically. Through the combination of erosive and depositional forces, existing wetlands may be deepened, filled, or otherwise altered, and new wetlands and channels may be formed. Shallow depressions are scoured out, with and without inflow or outflow channels to the river. Low elevation areas are flooded for extended periods of time, which may shift plant communities toward more hydrophytic species. When the floodwaters recede, infiltrate, or evaporate, then the duration of inundation or saturation, coupled with the interplay of these surface waters with groundwater, will determine whether a wetland is formed, and if so, of what type.

Flow resistance on a floodplain, and hence, its ability to withstand the high kinetic energy of flood pulses and flood flows, is based on the presence and amount of coarse woody debris (CWD), microtopography, and vegetation. The size and density of plant stems (e.g., tree trunks vs. dense grasses) introduces impediments to surface water flow and reduces the energy of storm runoff. Roughness created by vegetation and other physical features (as represented in roughness coefficients such as those in Arcement and Schneider 1989) slows current velocities, causing water to deposit sediment and debris. High vegetation density corresponds to higher effective roughness, flow resistance, and erosion protection of the system. Areas of dense vegetation, including herbaceous and shrub species, and especially mixed age classes of trees, slow flow and create slack water, allowing the water retained by the floodplain to be available for infiltration and potential groundwater recharge. Microtopographic complexity increases the tortuosity of flow pathways, reduces average velocity and increases the variety of moisture conditions present in a site. This increases the diversity of biogeochemical processes occurring in the wetland, especially nitrogen cycling (Brinson et al. 1995), and the presence of abundant and varied microhabitats (Brooks et al. 2004). CWD in the channel, derived from large trees and snags, blocks flows and modifies flow patterns, accelerating the lateral migration of streams and rivers as is expected in a naturally functioning floodplain.

1.3.2 Hydrology/Floodplain and Channel Morphometry

Patterns of discharge and current velocity are the primary hydrologic determinants of species composition in streams and rivers through their influence on carbon and nutrient transport (Newbold et al. 1982), habitat formation and stability (Giberson and Caissie 1998), and direct effects on species mortality patterns resulting from extreme flow events including floods and droughts (Reice 1984). Similarly, wetlands depend on specific hydrologic flows patterns, energies, and duration to maintain their characteristic flora and fauna. While precipitation is the driving force in initiating a flooding event, the physical characteristics of the drainage basin, hydrology, and geomorphology of the stream–floodplain ecosystem are the primary factors controlling the concentration, spatial distribution, and dispersal rate of floodwaters (Staubitz and Sobashinski 1983). Thus, the amount of flooding in a wetland or floodplain is dependent on climate, topography, channel capacity and slope, soil, and lithology (Novitzki 1989; Brinson 1990; Thorne 1998).

Small streams are more directly influenced by precipitation events and thus are "flashier" than larger rivers (Junk and Welcomme 1990; Benke et al. 2000). Although climate and geology are important, they are generally considered to be similar within a given region and wetland type, which holds true for much of the MAR. Differences in landscape-level characteristics, such as upland land uses and stream size are important characteristics to consider. Site-level indicators, however, can be useful for describing these systems with regard to measures of ecological integrity and for functional assessments (Brinson et al. 1995). Riparian–wetland areas function properly when site-level indicators such as adequate vegetation, landforms, or large woody debris are present to dissipate stream energy and improve floodwater retention and groundwater recharge.

The physical characteristics of floodplain wetlands determine the potential of an area to store and manage floodwaters. Wetlands, particularly various types of depressions adjacent to streams, reduce the amount of runoff that reaches the streams by storing runoff from adjoining areas (Demissie and Khan 1993). This desynchronizes water delivery to streams, which decreases the frequency and magnitude of flooding downstream (McAllister et al. 2000). Floodplains provide a broad area for floodwaters to dissipate energy through the reductions of water velocities, flood peaks, and erosion. Floodplain vegetation retards water flow through surface roughness (Arcement and Schneider 1989), although slope remains an overarching variable in determining discharge. Topographic depressions, backwater swamp areas, and low elevation areas behind natural levees trap floodwaters as long-term storage, only to be depleted by subsurface seepage to the channel and evapotranspiration (Owen and Wall 1989). In a recent white paper, the Association of State Floodplain Managers (2008) reversed past guidance by stating that conserving the natural hydrodynamics and vegetative structure of floodplains would do more for protecting these critically important systems than imposing engineered solutions.

It is useful to consider the flow of water and materials from the upper reaches of the watershed to lower reaches. Initially, waters at the watershed boundary begin to accumulate in surface and near-surface areas. Precipitation, surface runoff, and near-surface runoff (i.e., interflow) accumulate in narrow, ephemeral, or intermittent channels (sometimes referred to as zero-order streams), or in headwater depressions. A portion of the precipitation component infiltrates into shallow and deep aquifers. The amount is dependent on the areal extent, vertical structure, and composition of vegetation, soil type, topographic gradients, surficial geology, and coverage of human-built structures. Discharges of shallow and deep groundwater may be expressed at the surface as springs, seeps, and slope wetlands, or below the surface entering directly into streams and wetlands, forming a hyporheic zone. Such discharges generally constitute the base flow to these aquatic systems.

As water tables rise to the surface near headwater stream channels, especially in areas of steep topography, the surface runoff of precipitation is nearly instantaneous and contributes to the flashiness of these streams. This “variable source area” expands and contracts depending on antecedent position of the water table as well as duration and intensity of a storm (Hewlett and Nutter 1970). Downstream from the headwater region, eventually, and somewhat dependent on season and the accumulation of base flow, sufficient water accumulates to sustain the flow as a perennial stream. Whereas the zero-order channel tends to dry out seasonally, first-order streams tend to have a persistent base flow in all but the driest of years, usually in a relatively linear channel with little or no floodplain. These relatively small elements are strongly influenced by the characteristics of the adjacent riparian corridor, including the amount of tree cover, type of soil, or range of stressors present. These influences, separated from inputs originating upstream, can be referred to as lateral effects, keeping in mind that these effects include flood flows moving in the opposite direction, connecting the stream channel back to the floodplain. The correspondence between stream order and perennial flow differs in the coastal plain ecoregion. Because of the flat topography, water tables frequently fall below the elevation of low order stream channels during periods of high evapotranspiration during the growing season. For this reason, channelized stream channels, created through deepening and widening actually have more perennial flow than their natural counterparts because the channel is incised deeper into the water table (Hardison et al. 2009).

As discharge increases, energy also increases to the point where physical modifications to the channel can occur. Pool–riffle complexes develop in the widening channels of tributary streams (second to fourth order) (Forman 1995; Naiman et al. 2005). Floodplains continue to widen as the flow transitions from tributary streams to larger rivers (Fig. 1.3). In these stages, the river itself, and to some extent the adjoining floodplain, are tied more closely to the characteristics and periodicity of the flows that have accumulated from upstream reaches, and less by the activities in the riparian corridor and adjacent contributing watershed. Thus, mid-reach and mainstem portions of the river network become uncoupled from upland hill slopes and the sediments eroded from uplands. Sediments, first deposited and then resuspended and redeposited during flood events across the alluvial floodplain, define the channel and its flow path, and determine where and what type of wetland might be formed. This dependency is represented conceptually in Fig. 1.6a–e by the size of the arrows, which represent the strength of influence.

Long-term surface water storage, for weeks or more, helps to maintain the characteristic hydroperiod of wetlands and streams. Hydroperiod affects just about all components of aquatic ecosystems; plant communities, soil processes, nutrient cycling, and faunal communities are all influenced by the duration and frequency of inundation (Gosselink and Turner 1978, Carter 1986; Tiner 1988). Standard gauging stations have long been used to plot the expected hydrographs for streams and rivers throughout the USA. These data are readily available digitally online, although not all streams are gauged. On a smaller scale, Riparia and others have prepared typical hydrographs of the expected hydrologic regime for making comparisons among wetland subclasses (Fig. 1.8) (e.g., Cole and Brooks 2000; Brooks 2004; Cole unpublished, see Chap. 3 of this book). Deviations from this expected pattern can be used to suggest the presence of watershed stressors.

When one incorporates components outside the stream channel proper into the riverine model, complexity of the ecosystem increases. The accumulation and flow of water across the landscape coupled with the varied microtopography of these areas results in a *river mosaic* of hydrologically derived gradients and discontinuities across the surface (Forman 1995; Ward et al. 2002). The wetland components of these mosaics can be referred to as *headwater and floodplain complexes* (see Chap. 2 of this book for details on classification), whereas previously, wetlands were classified primarily on the dominant vegetation and hydrology (Cowardin et al. 1979). Alternatively, the HGM approach (Brinson 1993a; Smith et al. 1995; Cole et al. 2008; Brooks et al. 2011a) emphasizes physical elements for classifying wetlands and for comparing functions and condition across reference sites.

1.3.2.1 Impacts of Human and Beaver Activities and Alterations on Riverine Ecosystem Hydrology

Human activities upstream influence flood frequency and intensity (McAllister et al. 2000). Urbanization creates impervious surfaces and underground sewers, which accelerate the delivery rate of surface water to the stream (Pennsylvania Environmental Council 1973; Paul and Meyer 2001). As little as 3% impervious cover in a contributing area has been shown to negatively impact the ecological integrity of aquatic ecosystems (e.g., May et al. 1997). Serious declines in biological integrity have been observed when urban land exceeds 7% of total watershed area (Snyder et al. 2003). Channelization, levees, and floodwalls, both on-site and upstream, disconnect or destroy wetland and riparian habitat, restrict river flows, decrease water elevations at low flows, and increase water levels at the same locations during floods (Scientific Assessment and Strategy Team 1994). Channelization restricts flow within the stream channel, rather than allowing overbank flow to spread water across wetlands and decrease velocity (Brown 1988). This results in decreases in the ability of wetlands to perform other functions, such as removing sediment and nutrients, and long-term surface water storage (Johnston et al. 1984; Brown 1988; Rheinhardt et al. 1999). Channelization also alters stream morphometry, which leads to scouring and incision. Highway embankments remove vegetation, eliminate natural storage areas, and reduce space available for floodwater storage (Owen and Wall 1989). These and other activities often result in channel degradation, which lessens the depth, frequency, duration, and predictability of flooding. The floodplain frequently becomes increasingly isolated from the stream channel through incision and no longer has the opportunity to perform this function. These activities not only impair performance on-site, but they also increase the flood

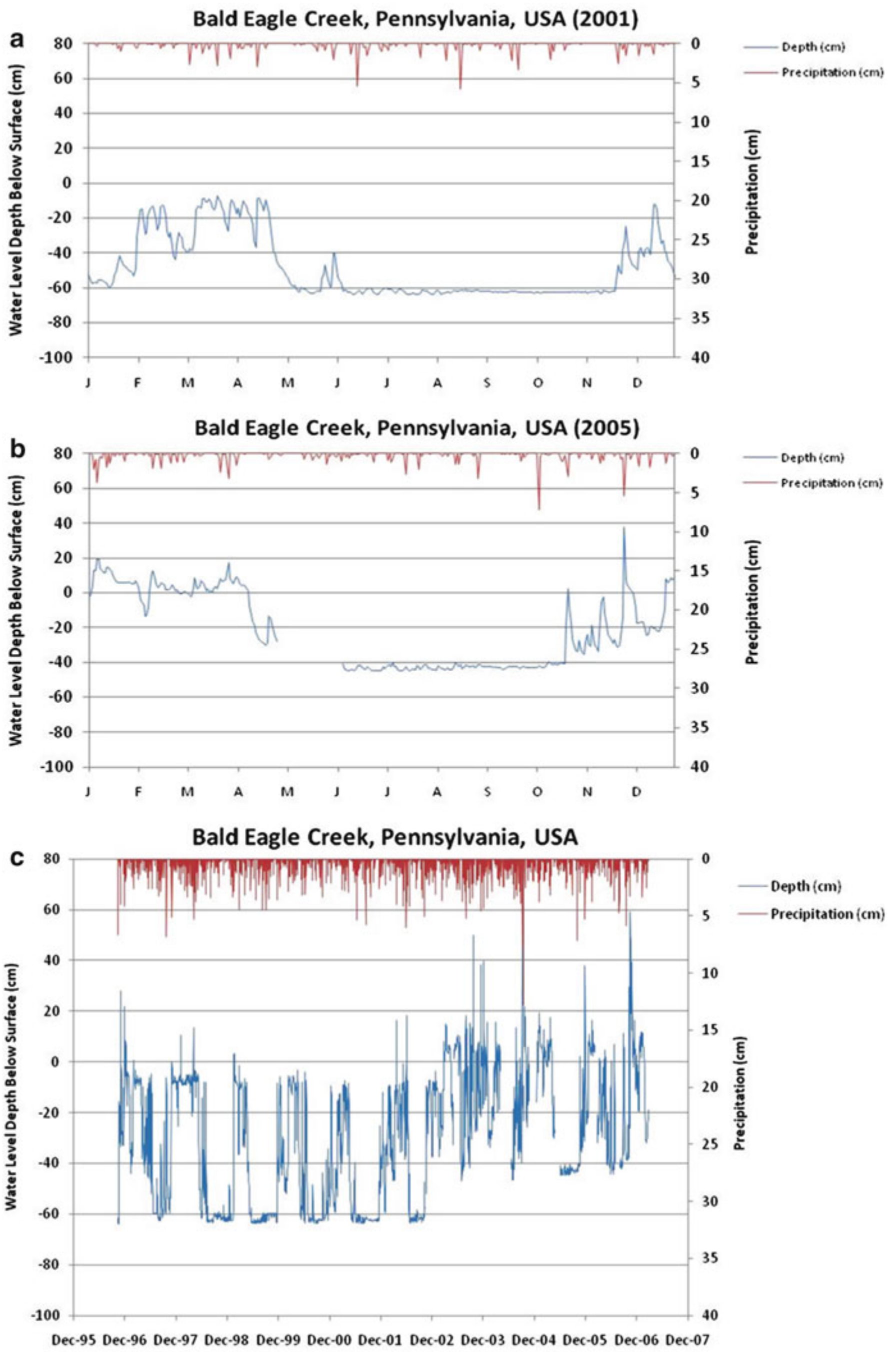

Fig. 1.8 Typical hydrograph for HGM Wetland Subclass, riverine lower perennial wetland in the floodplain of a fifth order river in central Pennsylvania, showing depth to water table in slotted well (0 cm on left side represents ground level) vs. precipitation (along top of graphs, scale in cm on right side) for: (**a**) drought year, (**b**) wet year, and (**c**) across a 12-year period (Data collected by C.A. Cole, graphs prepared by K.C. Hychka)

pulse downstream, a process that places additional pressure on downstream wetlands to dissipate energy and temporarily detain floodwaters. Even in urban areas where intensified peak flows lead to overbank events in incised channels, flooding is too brief to effectively saturate floodplain sediments and substantially raise water tables to permit wetland biogeochemical functioning (Hardison et al. 2009).

Changes in the structure and composition of surrounding landscape, particularly forests, can also have large effects on stream flow. For example, complete removal of forest vegetation associated with logging dramatically increases annual water yields and bank flow flood frequencies (Swank et al. 1988). In addition, human or pest-induced changes to the composition of surrounding forests can alter stream flow. For example, the hemlock woolly adelgid (HWA), an exotic insect forest pest that kills eastern hemlock trees, has been identified in numerous Mid-Atlantic watersheds. The pest is expected to cause significant and perhaps complete hemlock mortality. In a study designed to determine potential effects of HWA-induced hemlock decline on headwater streams in Delaware Water Gap National Recreation Area (DWG), Snyder et al. (2002) found that headwater streams draining hemlock forests were less likely to dry up completely during drought years than similar streams draining mixed hardwood forests. Based on their findings, they predict that HWA will have dramatic effects on headwater stream hydrology, and thus, biodiversity.

Channelization increases the rate of runoff, which increases peak flow, and decreases water storage and the residence time of water (Brown 1988). Studies show that increases in water level fluctuation relate directly to increases in runoff from adjacent uplands (Euliss and Mushet 1996). Human alterations also cause an increase in the amount of sediment transported in a stream and ultimately across a floodplain. Excess sediment can fill critically important interstitial spaces in the substrate of streams, reducing or eliminating aquatic biota (e.g., larvae of aquatic insects, salamanders, and fishes). This same source of sediment may result in the filling of depressions or reduce plant germination rates (Mahaney et al. 2004), and hence, cause a reduction in the storage capacity and topographic complexity of wetlands and on the floodplain in general.

A discussion of headwater wetlands would not be complete without describing the role of beaver. The primary effect of beaver ponds is to expand the area of wetlands in headwater streams, often by several-fold. Several studies in the MAR have demonstrated both the habitat and the water quality consequences of beaver pond establishment in the Adirondack Mountains of New York (Cirmo and Driscoll 1993), the Appalachian Plateau of Pennsylvania and Maryland (Margolis et al. 2001) and the coastal plain of Maryland (Correll et al. 2000) and North Carolina (Bason 2004; Bason and Brinson in preparation). These and studies in other geographical regions demonstrate an expansion of aquatic habitat, conversion of riparian forest and adjacent upland forest to open water and herbaceous wetland vegetation, and the concomitant shift in species composition toward more abundant waterfowl, wading birds, and a host of amphibians and reptiles. Effects on water quality are profound and represent an amplification of the biogeochemical processes discussed below in the section on *Water Quality and Biogeochemistry.* Principal among these effects is the removal of nitrate, presumably by denitrification, due to the detention of water in the ponds (relative to a stream channel) and the expansion of surface area of organic-rich sediments (Bason and Brinson in preparation).

1.3.3 Energy Flow and Sources

The changes in the relative importance of energy sources and the associated changes in plant and animal species structure and composition is the basis of the RCC. Essentially, the RCC proposes that, in unperturbed watersheds, stream communities change in predictable ways as we move from headwaters to large rivers and these changes are mediated by a continuum of physical gradients that control the amount and sources of energy (Vannote et al. 1980). It is important to understand these aspects of headwater stream to understand how they potentially interact with adjacent wetlands.

Within the headwater stream component of tributary watersheds (zero through second order), the source of detrital energy is mainly from outside the stream channel (i.e., allochthonous inputs), largely in the form of leaf litter, or coarse particulate organic matter (CPOM) (Cummins et al. 1973). The quantity, quality, and timing of leaf litter inputs vary depending on regional climate and the structure and composition of the surrounding forests. In watersheds draining deciduous forests (which predominate in Mid-Atlantic watersheds), the majority of leaf litter inputs to streams occur in autumn when deciduous trees naturally lose their leaves. However, in watersheds comprising mainly coniferous forests, inputs may be more consistent throughout the year. The source of leaves can originate from either upland forests or forested wetlands. Once leaves or conifer needles fall into headwater streams, a large fraction of the associated carbon is rapidly dissolved or leached directly into the water as dissolved organic matter (DOC) and transported to downstream reaches with flow. The CPOM tends to accumulate in pools and stream margins into leaf packs where they are colonized and undergo decomposition by bacteria and aquatic fungi, a process termed conditioning (Cummins and Klug 1979). These incompletely decomposed but conditioned leaves are then available as food for aquatic macroinvertebrates, which in turn supports the production of fish and other secondary consumers. In addition to being used directly by microbial and macroinvertebrate assemblages, a significant fraction of the CPOM component is broken down into smaller particles or fine particulate organic matter (FPOM), by the abrasive forces of stream flow and by the feeding activity of leaf-shredding macroinvertebrates (Boling et al. 1975; Iversen et al. 1982). Subsequently, FPOM is suspended into the water column and exported to stream reaches downstream (e.g., Neatrour et al. 2004).

In forested watersheds, the quantity of leaf litter that enters streams is not limiting. However, the extent to which litter inputs are available to stream communities depends on two factors. The first is the extent to which headwater streams can retain CPOM within headwater reaches in the face of downstream flow. Although increases in flow associated with storms are responsible for most export of CPOM from headwater streams on an annual basis (Schlesinger and Melack 1981), correlations between discharge and transport of organic carbon are weak in headwater streams, especially during non-storm periods indicating the importance of retention. There are many factors that influence organic matter retention including biological uptake. However, the physical structure and complexity of stream channels have been implicated as primary determinants. Specifically, physical features of streams such as boulders and a stream channel that allows floodwaters to overflow their banks, slow the transport of water and materials downstream. Of particular note is the role that CWD plays in retaining particulate organic matter within headwater stream reaches (e.g., Bilby and Likens 1979; Wallace et al. 1995; Brookshire and Dwire 2003). Recent research indicates that undisturbed watersheds contain more CWD and are more retentive of carbon than disturbed watersheds, thus enhancing the availability of organic material to benthic consumers (Wallace et al. 2001; Scott et al. 2002).

The second factor that affects organic matter availability in headwater streams is the species composition of the leaf litter itself. Specifically, leaves of different plant species break down at different rates due in large measure to the chemical characteristics, especially nitrogen and fiber content, of the leaves (Webster and Benfield 1986). In an extensive study of leaf breakdown in streams, Peterson and Cummins (1974) found wide variation in leaf breakdown rates among species and suggested that this variation ensured that carbon was available to secondary consumers throughout the year. Conifer needles break down much slower than deciduous species and the leaves of herbaceous plant break down faster than those of woody plants. Thus, disturbances that change the composition of the surrounding riparian area would be expected to result in changes in the amount and timing of organic carbon available to stream communities. Forest pests such as HWA and gypsy moth are both common in some Mid-Atlantic watersheds and can have dramatic effects on the species composition of riparian forests. In addition, numerous abiotic factors have been shown to affect litter decomposition rates within plant species. For instance, litter breakdown rates are positively correlated with temperature and dissolved nutrients, and negatively correlated with acidity and various toxic effluents. Consequently, factors that reduce water quality are also expected to significantly alter the energy pathway in headwater streams and lead to disruptions in ecological integrity. Water quality of stream and wetland resources is a major concern throughout the various ecoregions of the MAR.

Further downstream in the mid-reaches (third and fourth order), stream channels begin to widen which allows more light to penetrate the forest canopy and reach the stream bottom. At this point, the RCC predicts instream primary production (i.e., autochthonous inputs) becomes an important energy source mainly in the form of benthic diatoms (Molloy 1992). In addition, FPOM derived and exported from upstream reaches also represents a significant energy source to stream biota. Thus, in unperturbed mid-reaches, direct litter inputs from the riparian zone diminish in importance and instream primary production and carbon inputs derived from upstream reaches become more important carbon sources to fuel secondary production. In addition to increased production and diversity of benthic algae in mid-reach streams, the composition of macroinvertebrate assemblages also change in response to changing sources of energy. Therefore, from an energy perspective, the ecological integrity of stream communities in mid-reach streams is determined mostly by factors that affect retention, transport, and the quality of organic matter from headwater areas upstream, and by factors that influence instream primary production within mid-reach areas. In particular, the effects of nonpoint source pollutants associated with agriculture and urban land use in upstream or adjacent landscapes, of significant concern in the MAR, have been shown to affect energy pathways in mid-reach areas. The colonization and movement of macroinvertebrates between streams and the adjoining floodplain and wetlands is discussed further in Chapter 10.

Herbicides and increased sediment inputs have been shown to reduce overall instream primary production with subsequent changes in macroinvertebrate diversity and production (Georgian and Wallace 1983; Guasch et al. 1998). Also, nutrient enrichment from agriculture has been shown to cause a shift in benthic algal composition from an assemblage dominated by diatoms, a preferred food source of many macroinvertebrate species, to an assemblage dominated by filamentous green and blue-green algae that detritivores mostly avoid (Hart and Robinson 1990; Jacoby et al. 1990). Acidification of stream habitats has also been shown to alter primary production in streams (e.g., Planas and Moreau 1986). Brooks et al. (2009) showed that for watershed throughout the MAR, the aforementioned set of stressors occurring collectively in wetlands, the floodplains, and in the stream channel, negatively impact both benthic macroinvertebrate and fish communities.

The RCC also makes specific predictions regarding energy sources in larger rivers and the associated responses of biological communities. Essentially, the RCC predicts that in larger rivers (>4th order) the primary energy source fueling secondary production is derived from terrestrial inputs that were inefficiently processed by consumers in headwaters and mid-reach sections of the drainage network and exported downstream. However, the data from larger rivers show less agreement with the predictions of the RCC than do headwater and mid-reach streams (Thorp et al. 2006). Consequently, other models have been proposed to explain energy pathways and food webs in larger rivers. The two that have received considerable attention are the Flood Pulse Concept (FPC, Junk et al. 1989) and the Riverine Productivity Model (RPM, Thorp and Delong 2002). The FPC emphasizes linkages between the river channel and the floodplain arguing that most secondary production is attributed to organic matter directly or indirectly derived from the periodic flooding of floodplain vegetation and aquatic macrophytes. Thus, the FPC explicitly singles out riparian wetlands as important drivers of large river food webs.

Like the FPC, the RPM recognizes that by the time organic matter, leaked from upstream areas, made it to large rivers (as proposed by the RCC) it is highly refractory and thus probably not sufficient to support secondary production in large rivers. Rather, Thorp and Delong (2002) postulated in the RPM that most large river consumers preferentially assimilate autochthonously derived carbon (i.e., algae and phytoplankton), and to a lesser extent, direct inputs from riparian areas. The RPM also explicitly recognizes the importance of impoundments and reservoirs to large river food webs. There is empirical support for both the FPC (e.g., Bayley 1989; Junk et al. 1989) and the RPM (Bunn et al. 2003; Delong and Thorp 2006), and it appears that the most appropriate of the two models for a given river depends on landscape-scale hydrologic characteristics (Hoeinghaus et al. 2007). Specifically, landscapes that result in low-gradient floodplain rivers are fueled primarily by aquatic macrophyte production associated with wetlands and riparian areas which supports the FPC, whereas landscapes that result in relatively shallow, high-gradient floodplain rivers are disproportionately driven by algae and phytoplankton which conforms to the RPM. Thus, although neither of these models has been tested sufficiently in MAR rivers, we would expect a higher fraction of the food web to be fueled by algae and phytoplankton in those systems located in mountainous regions such as the Ridge and Valley or Allegheny Highlands physiographic provinces, and by macrophytes and riparian vegetation in low-gradient regions such as the Piedmont and Coastal Plain physiographic provinces.

To summarize, recent evidence indicates that the ultimate carbon sources driving large river food webs primarily originate within the river reach or the adjacent floodplain, and not from POM exported from upstream reaches as postulated in the RCC. This does not imply, however, that large river food webs are not influenced by patterns and processes that occur within upstream portions of the watershed. In fact, disruptions to headwater and mid-reaches can have dramatic effects on riverine hydrology and water quality that would ultimately affect large river food webs. For example, urbanization in upstream portions of watersheds increases the frequency of major floods in large rivers (reviewed in Praskievicz and Heejun 2009) that could destroy or degrade riparian wetlands. Likewise, nonpoint source pollution from upstream has been shown to affect instream primary production in large rivers (e.g., Van Nieuwenhuyse and Jones 1996). Rather, predictions from the FPC and the RPM suggest that management and regulatory programs that rely solely on protecting upstream components of watersheds will, by themselves, not be sufficient to maintain biological integrity of large rivers. The protection of riparian areas (especially wetlands) and preserving floodplain–river channel linkages are also required. This new paradigm has significant implications for the conservation of large river biota and further complicates management activities that seek to balance environmental protection with human development. This is because many human activities that routinely take place within or adjacent to large rivers and once thought to be relatively benign to aquatic communities are now believed too highly disruptive. Moreover, many of these activities such as channelization projects to enhance navigation, dikes and levees for flood control, and development within the floodplain are either irreversible or restoration would be prohibitively expensive.

Tributary components of headwater systems are mainly forested under relatively unaltered conditions throughout the MAR. Not only do trees contribute leaf litter and downed wood described below, but forests provide shade, root structure for stream bank stability, and other physical and microclimatic controls to the stream. Wetlands in the riparian zone maintain the organic-rich conditions in the soil that is critical for denitrification in groundwater flowing to stream channels from nitrate sources such as agriculture (Peterjohn and Correll 1984). Forested wetlands support both aquatic and terrestrial food webs; aquatic food webs when they are flooded and terrestrial food webs when they are seasonally dry. As in nearly all terrestrial and aquatic food webs, the detrital food web dominates over grazing pathways (Brinson et al. 1981).

What sets apart headwater streams is the extent to which they are hydrologically connected to riparian wetlands (Fig. 1.6a). In contrast to larger rivers, where large volumes of water flow past floodplain wetlands, both headwater streams and their associate wetlands are minute by comparison, but they form dense dentritic or trellis networks intersecting the terrestrial areas throughout the MAR, often at a density of 1 km of stream per 1 km^2 of land surface. The consequence of this proximity and abundance is that they are most exposed to human activities that modify wetland condition. The most pernicious are channelization and ditching (straightening, deepening, and widening), processes that remove most hydrological and biological connections between stream channels and floodplain. The conversion of riparian forest to agriculture, pasture, residential areas, and urban land uses fundamentally changes ecological processes. To the extent that overbank flow during floods connected stream habitat with floodplain habitat, channelization and incision totally disrupts this connection and simplifies the complexity of food webs and the complex pathways of energy. Detrital biomass is an important component of headwater ecosystems and plays a role in nutrient cycling and habitat for plant and animal communities in tributary watersheds. Detrital biomass is represented by snags, down and dead woody debris, organic debris on the forest floor, and organic components of mineral soil. This has been described for wetlands in the national riverine HGM model (Brinson et al. 1995) and regional HGM models (Brooks 2004), and for Mid-Atlantic streams by Barbour et al. (1999) and Boward et al. (1999). Detritus is considered an indicator of the potential decomposition and nutrient cycling rates at a site. Decomposition is generally faster in aquatic than in terrestrial landscapes due to increased leaching, fragmentation, and microbial activity (Shure et al. 1986). Large pieces of CWD derived from adjacent or upstream forests are processed into FPOM and then further processed and incorporated into organic matter (Bilby and Likens 1979; Jones and Smock 1991). Organic material may be transported to channels or respired as CO_2 at any stage of the decomposition process (Bilby and Likens 1979; Jones and Smock 1991).

Riverine wetlands are a major source of particulate organic carbon (POC) entering streams. Woody debris is a nutritional substrate, provides habitat for microbes, invertebrates, and vertebrates, is a substrate for seedling growth, and serves as a long-term nutrient reservoir; a consistent source of organic material (Harmon et al. 1986; Brown 1990). POC is a small fraction of total organic carbon (TOC), but ranks disproportionately higher as a food source for fish and invertebrates (Taylor et al. 1990). POC from wetlands contributes substantial amounts of organic matter to stream channels (Mulholland and Kuenzler 1979, Dosskey and Bertsch 1994). In fact, POC comprises between 24 and 46% of the TOC in streams (Dosskey and Bertsch 1994). Detrital inputs to the stream during peak inundation periods support microbial and macroinvertebrate communities in the stream channel (Smock 1990). Although we tend to view movement of materials from uplands and wetlands into headwater streams, there are a significant number of invertebrate and vertebrate taxa moving laterally between streams and their associated floodplains and wetlands. There are macroinvertebrate taxa that are obligate to floodplain wetlands, relying on sufficient flood pulses to create habitats, periodic flood flows to maintain sufficient hydrology, and retention of POC within the wetland itself as a food source (see Chap. 9 of this book, Yetter).

The rate of particulate matter degradation depends on many factors, including soil moisture levels. According to Bilby et al. (1999), when compared to either fully submerged or terrestrial conditions, wood decays at a much faster rate when periodically wetted and dried, conditions typical of many wetlands and floodplains. Floodplains have higher decomposition rates for wood than streams (Cuffney 1988). Forested riparian corridors maintain more benthic habitat, increase channel and bank stability, and provide additional contact area for transforming both nutrients and pesticides than non-forested reaches (Sweeney et al. 2004).

1.3.4 Water Quality and Biogeochemistry

Headwater stream and wetland communities are strongly influenced by the chemistry of the surface and ground waters than those associated with larger rivers (Jones and Mulholland 2000). Natural variation in water hardness, specific conductance, acidity, and dissolved oxygen are all major determinant of species composition, and consequently must be considered when seeking to understand reference conditions and when designing a sampling program to monitoring aquatic resources. This is of interest because human sources of pollution reduce water quality and alter aquatic communities directly by killing or weakening individuals, or by altering energy pathways.

In the headwater portions of watersheds, measures of water chemistry are more reflective of the geologic and topographic characteristics of the landscape than for the lower reaches of larger rivers. The complex geology of the Appalachians, running through several Mid-Atlantic ecoregions, can create circumstances where relatively short stream reaches and individual wetlands can have different water chemistry than their neighbors (USEPA 2000; Snyder et al. 2006). Such variability produces extraordinary biodiversity at a regional scale.

Numerous human sources of water quality degradation have been identified within the region, including urbanization, failing septic systems, agriculture, acid mine drainage (AMD), and acid precipitation. Wetlands and riparian corridors often act as buffers to these water sources due to their ability to filter out and transform contaminants (e.g., Fig.1.6a–e). Of particular importance are nonpoint pollutants, including nutrients such as nitrogen, phosphorus, pesticides, herbicides, and sediments that enter stream and wetland habitats through shallow groundwater and surface runoff.

Eutrophication from excess nutrients (e.g., nitrogen and phosphorus) can be a significant stressor in aquatic ecosystems. Over time, eutrophication typically alters energy pathways by increasing primary production (see section on Energy flow and

Sources above), which often results in lower dissolved oxygen concentrations due to excessive organic matter decomposition. These changes usually lead to highly productive, but taxonomically and trophically simple biological communities in both streams and wetlands (Reddy et al. 1999; Sandin and Johnson 2000; Brinson and Malvarez 2002). Herbicides also disrupt energy pathways, but they cause reductions in instream primary production, and pesticides directly affect survival and reproduction of populations. Excess turbidity caused by high levels of suspended sediment decreases oxygen levels and photosynthesis rates, impairs the respiration and feeding of aquatic organisms, destroys fish habitat, and kills benthic organisms (Johnston 1993b). In wetlands, high sedimentation rates decrease the germination of many wetland plant species by eliminating light penetration to seeds, lowering plant productivity by creating stressful conditions, and slowing decomposition rates by burying plant material (Jurik et al. 1994; Vargo et al. 1998; Wardrop and Brooks 1998; Mahaney et al. 2004).

In some instances wetland and riparian habitats can be effective mitigators of nonpoint source pollutants, especially nutrients and sediments, due to their ability to filter and transform contaminants, if their capacities are not exceeded. Because sediments and phosphorus are transported from uplands to streams and wetlands through surface flow, the primary removal mechanisms for phosphorus and metals are the settling of particles out of the water column and adsorption to organic matter and clay. Long-term removal can occur through roots, buried leaves, and sediment deposition (Richardson and Craft 1993). As long as there is sufficient time for transported material to come in contact with surface litter, riparian vegetation can be effective in retaining sediments and nutrients. For example, in a floodplain wetland in Sweden, 95% of phosphorus entering the wetland in surface runoff was removed within 16 m (Vought et al. 1994). In North Carolina, approximately 50% of the phosphorus leaving agricultural fields in runoff was removed in riparian areas (Cooper an. Gilliam 1987). However, during storms and in high-gradient watersheds, sediment retention by riparian zones is less effective (Jordon 1986). Phosphorus is even more sensitive to flow rates because it tends to bind to smaller particles that are less efficiently trapped by surface litter. In contrast, nitrogen moves primarily through ground water as dissolved nitrate, ammonia, or organic nitrogen (Peterjohn and Correll 1984), and thus, its removal is less tied to sediment dynamics than phosphorus.

Most nitrogen is removed from subsurface water through denitrification by soil microbes within wetlands and riparian soils (Davidsson and Stahl 2000). Research has shown that riparian forests are capable of retaining up to 89% as compared to 8% for cropland, and the nitrogen loss from the forest was primarily via groundwater (Peterjohn and Correll 1984; Gilliam 1994; Jordan et al. 1997). But as with sediments and phosphorus, retention of nitrogen is also more efficient at low discharge. During high discharge, relatively more water moves from upland and riparian areas to streams and lowland wetlands through surface flow vs. shallow groundwater flows. Thus, there is less time for vegetative uptake and microbial transformation of nutrients (Pionke et al. 2000). Research has shown a 90% or more reduction in NO_3^- concentrations in water as it flows through riparian areas (Groffman et al. 1992;

Gilliam 1994; Vidon et al. 2010). Labile organic matter is fundamental to denitrification as it provides the substrate necessary to drive this process for microbes to perform the process of denitrification. Plant uptake is an additional means of nitrogen removal from the system. Channelization and channel incision interferes with both of these processes by driving groundwater flowpaths deeper below the organic-rich alluvium (Phillips et al. 1993).

Sediment retention in wetlands and riparian corridors not only removes phosphorus, but has the additional function of reducing turbidity and contaminants sorbed to sediments, thus benefiting neighboring streams, rivers, and lakes (Oschwald 1972; Boto and Patrick 1978; Cooper an. Gilliam 1987; Hemond and Benoit 1988; Johnston 1991). While wetlands and floodplains have been shown to trap sediment in relatively unaltered settings, accelerated sedimentation can quickly overwhelm the capacity of these habitats to store and process sediments (Jurik et al. 1994; Wardrop and Brooks 1998; Freeland et al. 1999). High sedimentation rates decrease germination of many wetland and riparian plant species by eliminating light penetration to seeds, lower plant productivity by creating stressful conditions, and slow decomposition rates by burying plant material (Jurik et al. 1994; Vargo et al. 1998; Wardrop and Brooks 1998; Mahaney et al. 2004).

Landscape disturbances impact sediment loading and retention within the aquatic components of watersheds, and for the MAR, anthropogenic disturbances have been occurring for several centuries. Walter and Merritts (2008) reported on the sediments stored behind 10,000 s of mill dams in the MAR that as they become derelict are releasing “legacy” sediments to downstream areas. The influence of these sediment releases, mostly unpredictable, on aquatic ecosystems downstream is difficult to discern. In addition, former mill dams and beaver dams may, in fact, be the origins of some wetlands along headwater streams. Hupp et al. (1993) found sedimentation rates to be highest in wetlands located downstream from agricultural and urban areas. Phillips (1989) found that between 14 and 58% of eroded upland sediment is stored in alluvial wetlands and other aquatic environments, and as much as 90% of eroded agricultural soil was retained in a forested floodplain in North Carolina (Gilliam 1994). Eighty-eight percent of the sediment leaving agricultural fields over the last 20 years was retained in the watershed of a North Carolina swamp (Cooper et al. 1986). Approximately 80% of this was retained in riparian areas above the swamp and 22% was retained in the wetland itself. Carline and Walsh (2007) found that streambank fencing restricting livestock access to narrow riparian corridors (3–4 m) along pastures in a central Pennsylvania watershed reduced suspended solids by >50% resulting in an increase in abundance of benthic macroinvertebrates. These studies and others demonstrate the critical need to consider the aquatic and terrestrial mosaic of riverine ecosystems.

Another major threat to water quality in streams and wetlands of higher elevations is increased acidity associated with AMD and acid deposition (AD). As a region, the pH of rainfall in the MAR is among the lowest nationwide (NADP 2003), and, although long-term monitoring have shown wide-spread improvements in air quality and a reduction in acid deposition (Stoddard et al. 1999), aquatic biota in MAR streams and lakes have not shown evidence of recovery to AD (Stoddard et al. 1999). Low pH has a negative impact on the presence and breeding success of pond-breeding amphibians in wetlands, even more so when in combination with high concentrations of metals like aluminum and zinc (Clark 1985; Sadinski and Dunson 1992; Rowe and Dunson 1993; Horne and Dunson 1994). Snyder et al. (2005) found that over one quarter of wetlands in the Delaware Water Gap National Recreation Area along the Pennsylvania–New Jersey border had pH levels <5. In this study area, the lowest pH levels are among depression wetlands that lack permanent stream connections to adjacent water bodies (Julian 2009). This trend threatens amphibian species that breed early in the year because wetlands that lack permanent stream connections are used most frequently by early-breeding species (Julian 2009). Early-breeding species, like ambystomatid salamanders (Family Ambystomatidae), prefer these habitats because the drying regimes of these wetlands reduce aquatic predator populations. In DWG, ambystomatid salamanders bred in half of all wetlands whose maximum flooded areas were more than halved by the start of summer (Julian et al. 2006), yet Snyder et al. (2005) found they were excluded from these wetlands if the pH ≤4.6.

Increased acidity can have dramatic effects on stream and wetland communities, particularly in headwaters that are poorly buffered, chemically speaking. Increased H^+ ions directly disrupt ion regulation in most animal species causing death or compromising fitness depending on the level (Gerhardt 1993). Certain metals such as aluminum, which are prevalent but relatively inert in streamside soils and stream sediments, become dissolved, mobilized, and toxic to aquatic species at low pH (Nelson and Campbell 1991). In addition, when acidic waters merge with pH neutral or basic waters at stream junctures, certain metal complexes such as iron hydroxide precipitate out of solution and coat stream substrates thus smothering benthic algae and macroinvertebrates (DeNicola and Stapleton 2002). Consequently, acid effects can extend downstream even in areas where stream pH is relatively high. Finally, leaf litter decomposition rates in headwater streams and wetlands are significantly reduced as streams become acidified (Kittle et al. 1995; Niyogi et al. 2001). The lower reaches of most rivers flow primarily through valleys of the various ecoregions in the MAR. In the Ridge and Valley ecoregion, calcium-rich limestone comprises the underlying bedrock, thus neutralizing the detrimental effects of acidification that arises in the highlands. In other ecoregions, the larger discharges of some rivers tend to override the influence of underlying geologic strata, or alluvial fill for coastal plains.

In the case of AMD, acidity and metal concentrations are frequently so high that the affected stream or wetland may be devoid of all life. In less extreme cases, AMD and AD has been shown to adversely affect the species diversity and productivity of benthic algae (e.g., Verb and Vis 2000), macroinvertebrates (e.g., Rosemund et al. 1992), amphibians (see Freda 1991 and Freda et al. 1991, for reviews mostly concerning wetland and vernal pool-breeding amphibians), and fish (Carline et al. 1992). After completing three separate investigations, which included field sampling as well as in situ and laboratory bioassays, reduced abundance and distribution of most lungless stream salamanders (Family Plethodontidae) was attributed to stream acidification (Rocco 2007).

The pH of water in streams receiving AMD or AD is often poorly correlated with the pH of the sources indicating that some systems are more vulnerable than others to acidification. As mentioned above, the water chemistry of headwater streams are more strongly related to the geology and terrain of the surrounding watershed than for larger rivers (Babb et al. 1997). One important characteristic of headwater stream water chemistry that is an important determinant of sensitivity to AMD and AD is the concentration of base cations (e.g., calcium and magnesium). Streams that have high base cation concentrations typically have high acid neutralizing capacities (ANC), and are, therefore, more able to maintain a stable pH despite AD (Faust 1983). In headwater streams, base cation concentrations are largely a function of the underlying surface geology. Streams underlain by carbonate geologies such as limestone supply considerable ANC to streams compared to geologies with little or no base cations like sandstone. However, the ability of carbonate geologies to buffer acidity associated with AD also depends on the amount of time that streams are exposed to AD. Specifically, in streams exposed to AD, the production of base cations through mineral weathering is slower than the rate they are leached into the stream. Thus, over time, the pool of available cations may become depleted causing a threshold effect whereby the ability of carbonate geologies to buffer AD is compromised (Kirchner 1992).

Wetlands offer some mitigation potential for acidified streams. For example, comparative research studies have shown that beaver ponds generate significant ANC to associated streams resulting in more stable pH (Cirmo et al. 2000; Margolis et al. 2001). Moreover, laboratory experiments using simulated wetlands have demonstrated that wetland soils act as sinks for strong acid anions (nitrates and sulfates), and wetland microbial communities transform toxic metals to less toxic or available forms (Tarutis et al. 1992; Williams et al. 1994). Constructed wetlands have been shown to be an effective mitigation tool for restoring streams affected by AMD by removing up to 99% of the iron and aluminum and up to 30% of the nitrogen loading (Brenner 2000). Although the ability of wetlands to ameliorate AMD was first observed in natural systems, constructed wetlands treatment systems are more likely to be used today, with design and size tailored to match the constituents of specific discharges.

1.3.5 Biological Integrity

The biological diversity of aquatic ecosystems in the MAR has been documented reasonably well. Some taxa pertinent to the region are particularly diverse, notably salamanders (Rocco et al. 2004), freshwater mussels, aquatic insects (Klemm et al. 2003), and breeding neotropical migrant songbirds (e.g., Stein et al. 2000; O'Connell et al. 2003; Tiner 2005). Various investigations have tallied the species and communities that are prevalent in the region (e.g., Majumdar et al. 1989; Croonquist and Brooks 1993; Brooks et al. 1998; Myers et al. 2000; Snyder et al. 2002; Ross et al. 2003) (see Chaps. 6, 7, 8, and 9 of this book for additional information about fauna).

The maintenance of a characteristic plant community is a fundamental property of ecosystems. It is a designated HGM function for wetlands that also relates to a variety of ecological functions in watersheds such as energy dissipation via roughness, stabilization of sediments and soils, detrital production and nutrient cycling, and biodiversity and habitat functions. The composition of vascular plant communities has long been used to characterize and classify wetlands (Cowardin et al. 1979; Tiner 1988; Mitsch and Gosselink 2000; Miller and Wardrop 2006). Plant community composition influences many ecosystem properties, such as primary productivity and nutrient cycling (Hobbie 1992). Plant species composition plays an important role in determining soil fertility through feedbacks attributable to the original potential for site productivity (Wedin and Tilman 1990; Hobbie 1992). Plant community composition also influences the habitat quality for invertebrate, vertebrate, and microbial communities in both wetlands and streams (Gregory et al. 1991; Andreas and Lichvar 1995; Norokorpi 1997; Ainslie et al. 1999).

Plant communities may be highly modified by human alterations that facilitate colonization by invasive and aggressive species. Invasive species change competitive interactions, which result in changes in species composition (Walker and Smith 1997; Woods 1997). A checklist, which includes provisions for invasive plants, has been developed to record any observed stressors on streams, wetlands, and riparian areas in the region (Brooks et al. 2006; Brooks et al. 2009). Streams and riparian systems are particularly vulnerable to exotics because their linear nature exposes them to invasions (Simberloff et al. 2005).

Land use can be considered a major driver of the characteristics and conditions of aquatic landscapes, and activities tied to changing land use can be the source of many stressors. Stream biological integrity is strongly correlated with the extent of agriculture, wetlands, and forests in the surrounding landscape (Roth et al. 1996; Snyder et al. 2003). Of particular importance to aquatic ecosystems are the patterns that arise along riparian corridors (Jordan et al. 1993; Castelle et al. 1994; Sweeney et al. 2004). In the MAR, stream reaches with wider forested riparian corridors, composed of uplands or wetlands, support higher abundance of macroinvertebrates, and process more carbon, nitrogen, and pesticides than narrower reaches. Because of these relationships, attributes of both landscape patterns and riparian corridors can be used to assess condition (King et al. 2005; Brooks et al. 2009).

When considering how various stressors influence aquatic landscapes, it is instructive to consider deviations from reference standard conditions that support the highest levels of biological integrity. In the eastern USA, the best attainable conditions for aquatic systems are usually derived from a landscape dominated by mature forests (or emergent marshes in the portions of some ecoregion such as the Coastal Plain), which produce characteristic inputs of organic matter, shade over wetlands and narrow stream corridors, and habitat for an expected set of species. In the floodplains of larger rivers, microtopographic heterogeneity arises from the interplay of hydrologic forces, vegetative structure, and underlying soil characteristics. The resultant mosaic of wet and dry patches found in natural floodplains and along the interfaces between aquatic and terrestrial systems support a diversity of biological communities adapted to wetting and drying cycles. These physical and biological complexities interact with and upon the materials present through biogeochemical processes to produce the ecological functions and services recognized from these systems.

Reference domains can exist for all major types of land use: forested, natural herbaceous, mixed, agricultural, and urban (Brooks et al. 2006). Most human-caused disturbances set back ecological succession to early stages. That is, for varying lengths of time, mature forests and large trees along riparian corridors will be absent, soil formation may be retarded, and the composition of floral and faunal communities will be different. Although natural processes also retard succession (e.g., severe floods, fire, disease, and insect epidemics), in the MAR these typically create a quilt-like mosaic of recovering habitat patches. As humans continue to transform the landscape of the MAR and elsewhere, forest cover is generally reduced, replaced by agricultural, suburban, and urban land uses linked through transportation and utility corridors, although in some areas forests are stable or increasing, and urbanization replaces agricultural lands (e.g., Brooks et al. 2011b). The spatial extent and pattern of these changes determine the degree of alteration and degradation observed in aquatic landscapes. Additionally, point sources of urban stormwater, agricultural runoff, and other pollutants can severely degrade these systems. Degrees of change can be detected through monitoring if selected attributes are used as indicators or vital signs. If the desired spatial mix and connectivity of natural habitats and human-influenced land uses can be determined, then land use policies and management practices can be focused on achieving those goals. The common thread to consider when planning land use policies and practices is to treat aquatic landscapes holistically rather than as a set of separate, disconnected components. This theme courses through the remaining chapters.

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Chapter 2
Hydrogeomorphic (HGM) Classification, Inventory, and Reference Wetlands

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Abstract Classifying wetlands is useful for describing and managing their natural variability. The hydrogeomorphic (HGM) approach, which covers classification, reference, and functional assessment aspects, has proven to be helpful in classifying wetlands as to their position in the landscape, their source of water, and the flow of that water. In this chapter, we review the origins and characteristics of freshwater wetlands for ecoregions of the Mid-Atlantic region (MAR), which are dominated by riverine types. Inventories of wetlands in the MAR are dated, so we discuss what is known with regard to status and trends, and potential solutions. We discuss the value of establishing a reference set to assist with classification, assessment, and mitigation of wetlands, and describe the set of reference wetlands compiled for Pennsylvania by Riparia. Preliminary results from a regional condition assessment of wetlands in the MAR are provided.

2.1 Classification

We classify things, habitats included, because we need a way to systematically organize the data or information we have collected into a conceptual framework that is useful to us. A natural resource inventory can be described as a list of observable

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or measurable physical, chemical, and biological features. These inventory "snapshots" from a narrow window of space and time were used to group types of streams into categories defined by specific objectives (Naiman et al. 2005). Similarly, if we wish to develop a representation of the land cover and land use for a geographic area, spectral signatures from satellite imagery can be grouped to represent forest, field, or urban lands. We can group wetland plants into families and genera to populate a botanical classification system, or into functional groups, or guilds if we prefer to focus on their ecological traits. In each case, classification provides a way to organize an array of things into logical groupings.

2.1.1 Classification in the Hydrogeomorphic Approach

Recent evidence indicates utility in classifying wetlands as to their position in the landscape, their source of water, and the flow of that water. These concepts are the basis of the hydrogeomorphic (HGM) approach, and were adopted for national implementation in permit evaluations and watershed planning by the U.S. Army Corps of Engineers (Brinson 1993; Smith et al. 1995). The HGM classification system recognizes seven major classes, which can be further divided into subclasses. From a national perspective, HGM classes consist of **Riverine**, **Depression**, **Slope**, **Fringe** (**Lacustrine and Estuarine**), **and Flats** (**Organic and Mineral**). *Bolded* terms in this chapter refer to HGM classes and subclasses in Table 2.1, a wetland classification system for the Mid-Atlantic region (MAR) (Brooks et al. 2011).

The same inherent variability in ecological characteristics that defines ecological function and instills societal value for wetlands has hindered their classification and protection. The classification system of Cowardin et al. (1979) that describes vegetation and hydroperiod, neglects differences in morphometry and landscape position. Classification by HGM harnesses this additional wetland variability, and when integrated with Cowardin et al. (1979), provides a framework to characterize observed differences in wetland structure and function. Wetland functions are closely tied to HGM class, and thus, wetlands in the same HGM class should have similar structure and functions. We have found this to be true for most measured variables (Brooks 2004).

Through an understanding of the distribution of HGM subclasses within a watershed and their relative condition, one can begin to assess how wetlands potentially contribute to watershed health. A change in HGM subclass distribution should signal an alteration of function within the watershed (Bedford 1996; Cole et al. 1997; Wardrop et al. 2007). Because of the tight coupling of function to wetland type, a change in the distribution of wetland type may be the first sign of a significant loss of function. Thus, the distribution of wetland type provides a logical and scientifically based first step of wetland and watershed protection.

Table 2.1 Key to tidal and nontidal hydrogeomorphic (HGM) wetland types in the Mid-Atlantic region of the US classes and subclasses are in *bold*. Please read footnote before using this wetland classification system[a]

1. Wetland found along tidal fringe of a marine ecosystem (ocean, beach, rocky shore)	2
1. Wetland not associated with marine ecosystem	3
2. Continuously submerged littoral zone	**Marine subtidal** (MF1)
2. Alternately flooded and exposed to air	**Marine intertidal** (MF2)
3. Wetland associated with shallow estuarine ecosystem (mixture of saline and freshwater)	4
3. Wetland not associated with shallow estuarine ecosystem	7
4. Wetland not impounded	5
4. Wetland impounded	**Estuarine impounded** (EFh)
5. Wetland continuously submerged	**Estuarine subtidal** (EF1)
5. Wetland alternately flooded and exposed to air	6
6. Wetland regularly or irregularly flooded by semidiurnal, storm, or spring tides	**Estuarine lunar intertidal** (EF2l)
6. Wetland flooding induced by wind	**Estuarine wind intertidal** (EF2w)
7. Wetland associated with freshwater stream or river	8
7. Wetland not associated with freshwater stream or river	11
8. Wetland associated with permanent flowing water from surface sources	9
8. Wetland dominated by ground water or intermittent flows	10
9. Wetland associated with low gradient tidal creek (see Estuarine types 3)	
9. Wetland associated with low gradient and low velocities, within a well-developed floodplain (typically >3rd order)	**Riverine lower perennial** (R2)
9. Wetland part of a mosaic dominated by floodplain features (former channels, depressions) that may include slope wetlands supported by ground water (see Slope 17)	**Riverine floodplain complex** (R2c)
9. Wetland associated with high gradient and high velocities with relatively straight channel, with or without a floodplain (typically 1st - 3rd order)	**Riverine upper perennial** (R3)
10. Wetland part of a mosaic of small streams, depressions, and slope wetlands generally supported by ground water	**Riverine headwater complex** (R3c)
10. Wetland associated with intermittent hydroperiod	**Riverine intermittent** (R4)
Note: For any riverine type that is impounded, distinguish between:	
Wetland impounded by beaver activity	**Riverine...beaver impounded** (R...b)
Wetland impounded by human activity	**Riverine...human impounded** (R...h)
11. Wetland fringing on a lake or reservoir	12
11. Wetland not fringing on lake or reservoir	14
12. Wetland inundation controlled by relatively natural hydroperiod	13
13. Wetland inundation is permanent with minor fluctuations (year round)	**Lacustrine permanently flooded** (LFH)
13. Wetland inundation is semipermanent (growing season)	**Lacustrine semipermanently flooded** (LFF)
13. Wetland inundation is intermittent (substrate exposed often)	**Lacustrine intermittently flooded** (LFJ)
12. Wetland inundation controlled by dam releases	**Lacustrine artificially flooded** (LFK)
14. Wetland water source dominated by precipitation and vertical fluctuations of the water table due to low topographic relief	15
14. Wetland differs from above	16
15. Wetland substrate is primarily of mineral origin	**Flat mineral soil** (FLn)
15. Wetland substrate is primarily of organic origin	**Flat organic soil** (FLg)

(continued)

Table 2.1 (continued)

16. Wetland water source is primarily ground water and has unidirectional and horizontal flows — 17
16. Wetland forms a depression — 18
    17. Water source for wetland derived from structural geologic discontinuities resulting in discharge of groundwater from distinct point(s) on slope — **Stratigraphic slope**(SLs)
    17. Water source for wetland accumulates at toe-of-slope before discharging — **Topographic slope** (SLt)
        Note: For any slope type, distinguish between:
        Wetland substrate is primarily of mineral origin — **...slope mineral soil** (SL...n)
        Wetland substrate is primarily of organic origin — ...**slope organic soil** (SL...g)
    18. Wetland with frequent surface connections conveying channelized flow — **Depression perennial** (DFH)
    18. Wetland with infrequent surface water connections conveying channelized flow — **Depression seasonal** (DFC)
    18. Wetland with no surface outlet, often perched above water table — **Depression temporary** (DFA)
        Note: For any depression type that is impounded or excavated distinguish between:
        Wetland is impounded by human activities — **Depression...human impounded**(DPh)
        Wetland is excavated by human activities — **Depression...human excavated** (DPx)
        Wetland is impounded by beaver activities — Depression…beaver impounded (DPb)

Modified from Brooks et al. (2011)

[a]No classification system can capture effectively all of the inherent variability in natural systems, nor can it provide a foolproof determination given the different experiences of users. This wetland classification system for the Mid-Atlantic region is designed to distinguish among major wetland types with recognizable differences. It also purports to serve both the needs of the regulatory community where certainty is preferred, and the science community that grapples with variability in ecological systems. Given that dual function, it is critical that users consider the landscape and hydrologic contexts of each wetland. How large an area is being classified? A river channel and the associated floodplain on both sides of the channel, or just the wetland associated with a property on the upland edge of a floodplain. Context really matters, and should be carefully and succinctly documented

When seeking to classify a particular wetland, the most fundamental question the user must ask is, “How was the wetland formed?,” which can be stated as, “What is the origin of the wetland?.” If this question is thoughtfully answered and described in a brief narrative, then the actual label assigned to the wetland matters less, because the user will have considered where and how the wetland fits in a given landscape and hydrologic setting. Obviously, this is more relevant for regions where wetlands do not form the dominant matrix of a landscape (e.g., coastal salt marshes, bottomland hardwood forests)

For example, is it a depression that is isolated during drier times of the year, but located in a floodplain setting? Or is it isolated from all riverine influences, and receiving a combination of groundwater and precipitation? Clearly, these wetlands are distinctively different in many of their attributes and functions, but they could have the same morphometric dimensions. Either wetland also could have some characteristics of yet another type, warranting a dual label (e.g., depression/slope) just as NWI mapping recognizes mixed vegetation classes (e.g., forested/scrub-shrub, FO/SS). Thus, it is important to recognize these distinctive elements and document the reasons for labeling the wetland as a specific type. This is especially important when addressing wetlands that occur along a broad hydrologic gradient and when a group of microhabitats occur in a cluster. Thoughtful selection of classes supported by careful documentation will make any classification system more consistent among users

2.1.2 Wetland Hydrogeomorphic Syntypes and Holotypes

Borrowing nomenclature from biological systematics, we suggest that investigators proposing local or regional classification systems provide locations of representative examples of wetlands that typify each major HGM subclass (i.e., syntypes). In addition, a single site that is the archetypical member of the subclass should also be designated as a HGM holotype, displaying characteristics that best define a specific type of wetland for that region. Given that these are not species, but habitats, and that they are subject to fairly rapid changes from climatic and disturbance forces, having a single representative may not be desirable. Thus, we suggest listing several examples—syntypes—that are relatively homogeneous in structure and function when compared to other wetland subclasses, but that display the inherent, natural variability within the designated subclass.

Toward that goal, we recommend listing locations for sites on publicly accessible lands and providing representative photographs for a few examples of wetland subclasses. On the following website, there is an accumulation of images and associated data to be archived over time (http://www.riparia.psu.edu/MARbook). We envision that this growing database of wetland syntypes will become a useful service primarily for educational and training purposes for those seeking to learn and recognize the diversity of wetland types they occur in the MAR. Procedures for submitting exemplar wetlands are being considered at this time by the Society of Wetland Scientists (http://www.sws.org; Brooks and Tyrna 2012).

2.1.3 Origins and Landscape Settings of Freshwater Wetlands in the Mid-Atlantic Region

In this section, the influence of the regions' physiographic provinces or ecoregions (Fig. 2.1) on the origins and locations of wetland types and their abundance are described. Riverine types are described first, followed by the other major wetland types, contrasting their occurrence across an ecoregional gradient from the Atlantic coast to the Ohio River valley.

2.1.3.1 Riverine Wetlands

As stated in Chap. 1 of this book, the majority of freshwater wetlands in the MAR are **riverine** types, associated with streams and their floodplains. These include wetlands in-stream, occurring as narrow terraces or vegetated islands within the banks of the defined channel. Most, however, are found in the adjacent floodplain along with other features derived from the dynamics of the stream over time (Fig. 2.2). Riverine wetlands are described for each of the major physiographic regions in the MAR.

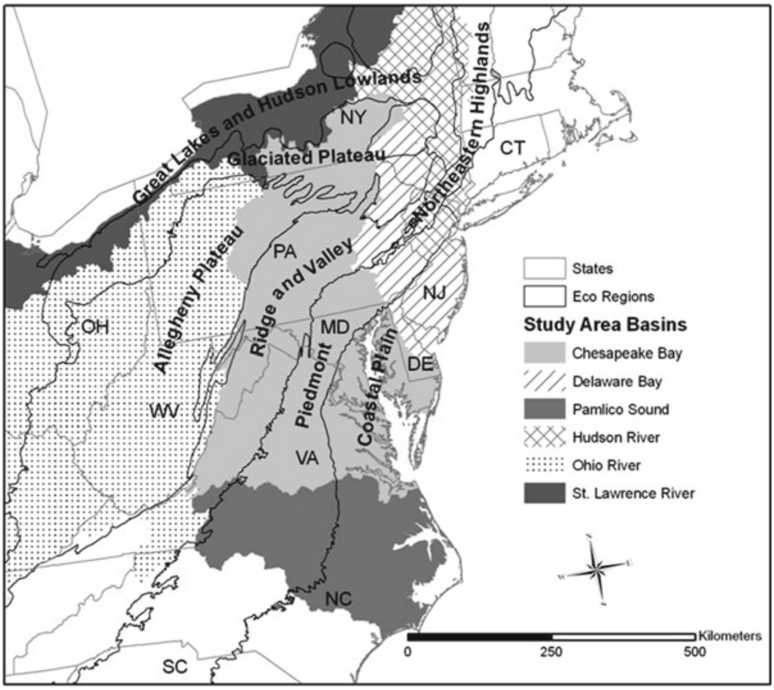

Fig. 2.1 Extended study area of the Mid-Atlantic region (MAR) showing boundaries of states, ecoregions, and major river basins

Coastal Plain

The Appalachian Mountains, the dominant geophysical features of the MAR, began rising to their peak elevations about 250 million years ago when the ancestral African continental plate collided with the eastern edge of the North American plate during the Alleghenian Orogeny (Slatick 2003), and have been eroding ever since. Rivers, at times interacting with glaciers, have carried these sediments southeastward to form the current Atlantic coastal plains, and westward to the wide floodplains of the Ohio River's valleys.

The coastlines that exist today along the Atlantic Slope are a product of the interplay of those southeast-flowing rivers and their bedload, changing sea elevations of the Atlantic Ocean, and variable rates of land subsidence and rebound. Along the coast, barrier islands contain coastal bays and drowned river mouths become estuaries, both rimmed with salt marshes and other wetland types. One does not generally encounter freshwater wetlands until locating reaches where the seaward freshwater flows of the rivers confront the incoming marine tides. At these zones of hydrologic tension, freshwater tidal and brackish marshes occur, referred to as **estuarine lunar or wind intertidal wetlands** in Brooks et al. (2011). Refer to overviews such as

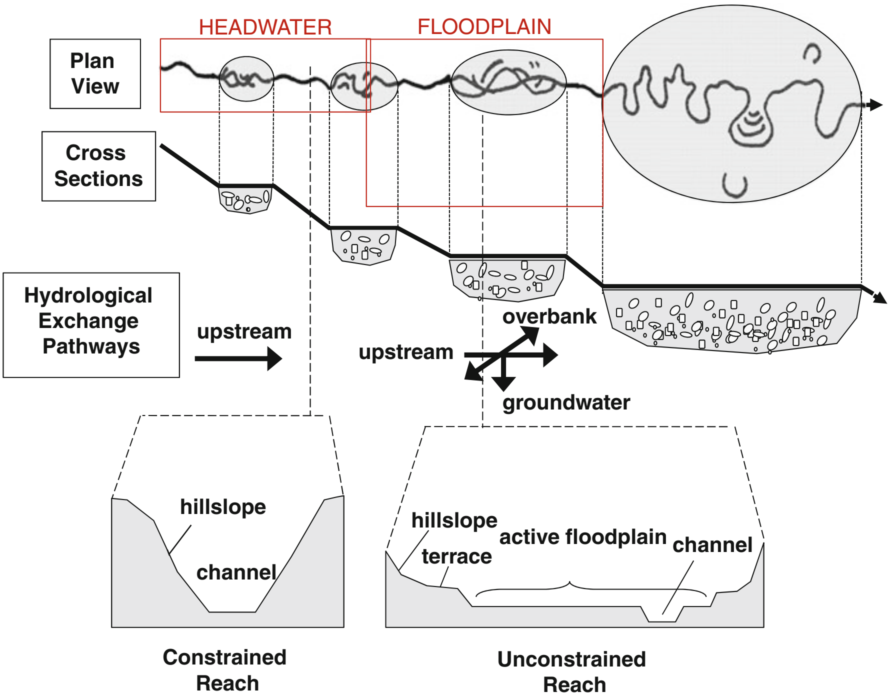

Fig. 2.2 Conceptualization of a riverine corridor derived over time by an active river channel interacting with the adjacent floodplain and uplands. Shown from headwater to mouth are alternating sequences of constrained and unconstrained floodplain reaches with predominant hydrological exchange pathways indicated for longitudinal (*horizontal arrows*), lateral (*oblique arrow*), and vertical (*vertical arrow*) dimensions (graphic by S. Yetter, modified from Ward et al. 2002)

Barendregt et al. 2009 or Perillo et al. 2009 for descriptions of these wetlands, as they are not discussed in this book.

The streams of the MAR's Coastal Plain flow across relatively flat landscapes formed of unconsolidated sands, silts, and clays. Elevations average 200 m above sea level (asl), but are much lower along the rivers and their floodplains. The sinuosity of these mature sections of rivers is high because of the numerous meanders formed over time across these weakly dissected alluvial soils (U.S. Forest Service 2010). Remnant channels become crescent-shaped oxbows that often support hydrophytic vegetation. Springs and other groundwater discharges erupt laterally along the edge of the floodplain and then mingle with surface water flows in a variety of ways, to produce **riverine floodplain complexes** (Brooks et al. 2011).

Floodwater dynamics, overbank flooding and subsequent floodplain deposition, and erosion from surface flow patterns, along with remnant meander scars and levees, produce distinct surface topographic and soil variations that then affect conditions for wetland formation. Biotic factors, such as beavers excavating bank dens or toppled trees, further modify the hydrologic conditions by affecting the conveyance of flood flows between the river and the bordering floodplain.

In areas where regional water tables are at or near the surface for extended periods of time, soils are saturated sufficiently to support vegetation indicative of **flats**.

Piedmont

A distinct and historically relevant line of cataracts or areas of steep gradient divide the more inland Piedmont ecoregion from the Coastal Plain. Many of the major cities of the region, such as Philadelphia, Baltimore, and Richmond, began where mills powered by hydraulic energy were established along this geologic feature, the "fall line."

Before European colonization, communities of indigenous peoples took advantage of the abundant resources found in these zones. These river rapids initially formed impediments to upstream navigation by colonizing Europeans, and thus, served as settlement and economic kernels from which these cities grew.

The riverine ecosystems of the MAR Piedmont exist on a mature, dissected peneplain interspersed with hilly and rolling terrain. Dendritic patterns are more common here than in the Ridge and Valley, where the predictable northeast-to-southwest orientation of both valleys and ridges tend to produce trellis drainage patterns. Bedrock is overlain by residuum on the ridges and hills, colluvium on the slopes, and alluvial materials in the valleys. Thus, the rivers wind through areas of soils and bedrock of varying resistances to erosion producing elevational ranges from 25 to 500 m (U.S. Forest Service 2010).

Wetlands of the Piedmont occur primarily along the riparian corridors. Headwater streams receive base flow from springs occurring at the break in slopes where hillsides meet flood plains. If the floodplains are wide enough, **topographic slope** and **riverine headwater complexes** are formed. In more narrow stream valleys, wetlands form linear strips within the channel or along the riparian banks creating a **riverine upper perennial subclass**. In active beaver habitats, dams maintained in place for years or decades can greatly enlarge these riverine wetlands. In the later successional stages of beaver habitat development, large patches of emergent hydrophytes form marshes or shrub wetlands on the fine sediments accumulating behind the dams; these are designated as **riverine beaver-impounded**.

Ridge and Valley

The Ridge and Valley Ecoregion is where the tortuous geology of the Appalachians is most apparent. The orogenic folding rearranged the horizontal bedding planes of sedimentary sandstone, siltstone, shale, carbonate rock, and coal into a myriad of angles from the original horizontal planes to completely vertical and even overturned strata. When coupled with fracturing and solution openings formed in carbonate rock the effects on groundwater storage and pathways, and on surface water drainage patterns are astonishing. Predicting the occurrences of wetlands on ridge tops, along hillslopes, and within the floodplains is challenging. Many of the headwater wetlands are small, and can be quite cryptic beneath the dominant forest canopies. Elevation ranges from 200 to 600 m (U.S. Forest Service 2010). Approaches for inventorying these types of wetlands are detailed later in this chapter in Sect. 2.

Wetland formation in the Ridge and Valley is similar to that in the Piedmont, and likewise, open-water or deep-water lakes are uncommon unless excavated ponds or dams have been built. These man-made impoundments often alter the dynamics of existing streams or rivers. Some headwater streams have steep gradients if they descend directly from the ridges to the valley bottoms, allowing few wetlands to form. However, tributary confluences often are sufficiently broad to form **riverine upper perennial** or **headwater complexes**. **Stratigraphic slopes** frequently form a distinct band along a topographic contour where the underlying geologic strata are less permeable than the strata above, leading to an expression of groundwater as a seep or spring. For example, a sandstone layer atop one of siltstone or shale may force groundwater to discharge at the surface where those two layers meet, forming a spring or slope wetland. Such contact zones can serve as indicators of wetland occurrence (e.g., McLaughlin 1999; Herz 2005).

Some ridge tops may have a saddle, providing landscape locations where wetlands can form. If flows are slow, and soils remain saturated, moss or sedge peat can accumulate and bogs or fens may develop (e.g., **depression perennial**, or **…beaver-impounded**; **riverine headwater complex**). Beaver can more easily dam the streams in these locations, and further enlarge wetland extent.

As tributary streams reach the broad valleys of this ecoregion, floodplains can be much wider, spanning hundreds of meters, providing larger areas for wetland formation; **riverine lower perennial**, **depression perennial or seasonal**. Rivers form meanders in the flatter valleys, but are often constrained on one or both sides by a ridge. The linearity of the ridges produces the trellis patterns common to the lower reaches of these watersheds. Steep riparian banks can form either where the river erodes into the base of ridges, or on the outer curve of meanders. These habitats are used extensively by nesting belted kingfishers and bank swallows, and by resting wood turtles. Point bars, or shallow, gently sloping areas of sediment deposition, form on the inner curve of meanders, where the river's kinetic energy is low. These flatter areas are more conducive for germinating annual hydrophytes and aquatic shrubs. Shorebirds, such as the spotted sandpiper, will frequent bare soils, whereas basking amphibians and turtles benefit from vegetated bars.

Two major factors can alter the typical pattern of heterogeneous wetland formation on broad floodplains. Karst landscapes form in carbonate valleys, where streams alternately flow aboveground, then belowground (i.e., sinking), entering solution channels, or sinkholes. The other factor is the centuries of human activities, including farming, transportation corridors, and urban development, that have altered natural river flow patterns through channelization, ditching, riparian bank hardening, and changes in soil infiltration rates. These valleys have been farmed extensively, removing most of the woody vegetation and influence hydrologic patterns. Today, either agriculture continues to dominate the valley bottoms, or urban development is expanding, with subsequent increases in impervious surfaces and stormwater runoff. In combination, these factors reduce the presence and extent of all types of wetlands in this region.

Allegheny Plateau: Unglaciated

The unglaciated Allegheny Plateau consists of a mature, dissected plateau, producing relatively narrow valleys, and hence, narrow floodplains. Elevation generally ranges from 200 to 400 m asl, with a few areas exceeding 600 m. As in much of the Appalachians, bedrock composed primarily of sandstone, siltstone, and shale is overlain by residuum on the ridges and hilltops, colluvium on the slopes, and either or both alluvium and Pleistocene lacustrine materials in the valleys (U.S. Forest Service 2010). The geologic strata, however, tend to remain more horizontal and much less folded than similar beds in the Ridge and Valley. Limestone and dolomite are less prominent than in the Ridge and Valley, whereas coal beds are more common, but vary in depth from the surface. Soils tend to be fine-grained, and are commonly acidic. The high acidity is a result of the parent bedrock material, high rates of wet and dry acid deposition from coal-fired power plants, and in isolated pockets, acid mine drainage produced during past coal mining activities.

As the region's name suggests, the upper elevations form broad, level, or gently undulating terrain. Where local water tables occur near to the surface, extensive wetlands, often forested, can form. Headwater streams intermingle with small, seasonal depressions and topographic slopes, which can produce a highly interspersed mixture of wetland and upland patches (e.g., **riverine headwater complexes**; **depressions seasonal and temporary**).

Dendritic stream networks have eroded the plateau producing, in some areas, steep, narrow valleys as can be found in central and western West Virginia and western Pennsylvania. Wetlands are linear in these valleys (e.g., **riverine upper perennial**), often further pinched by roads built to gain access into and through this steep, dissected terrain. Lower-lying areas are typically flooded more frequently than in the Ridge and Valley, creating a diversity of habitats with different hydrologic regimes, soils conditions, and plant communities (e.g., riverine lower perennial and floodplain complex). These floodplain forests are temporarily flooded during seasonal high water and periodic flood events, but during much of the growing season the groundwater may be well below the surface.

Allegheny Plateau: Glaciated

The glaciated portion of the Allegheny Plateau also consists of a mature, dissected plateau, but with rounded hills, ridges, and broad valleys. Elevation ranges from 200 to 300 m asl. Glacial features include valley scour, ground moraines, kames, eskers, and kettled outwash plains. Thin Pleistocene till and stratified drift cover many upland bedrock surfaces. Lower slopes are covered by colluvium. Glacial outwash, recent alluvium, and glacial lacustrine materials cover valley floors. Bedrock beneath the drift consists of shale, siltstone, sandstone, conglomerate, and coal (U.S. Forest Service 2010).

Not all soils are of glacial origin, with residual materials occurring in place upon weathered bedrock on the hills, and colluvium transported by water and ice to the lower slopes. Soils are highly variable, but tend to be more coarse-grained.

The stream networks display more amorphous patterns, having had much less time to develop (thousands vs. millions of years). Streams are underlain primarily by thick coarse sand and gravel in glacial outwash. Small natural lakes and wetlands (either bogs or marshes) are features of this glaciated landscape. Those derived from orphaned ice blocks of the receding glacier tend to form deeper kettle holes, whereas shallower depressions, typically forming emergent marshes or fens, have varied origins. The density of wetlands is much higher in both the northwestern and northeastern glaciated portions of the MAR (i.e., Pennsylvania) than any of the other ecoregions; 19% and 26% of land area, respectively (Tiner 1987). Whereas wetland patches of any type exceeding 10–20 ha are quite uncommon in the Piedmont and Ridge and Valley ecoregions, it is not uncommon to find wetlands >50–100 ha in area in the glaciated regions.

2.1.3.2 Other Wetland Types

The greatest density of **lacustrine** (**or fringing**) wetlands occurs along the natural and hydrologically altered lakes and ponds in the glaciated ecoregions of northeastern and northwestern Pennsylvania, northern New Jersey, and southern New York. The more acidic soils of the Pocono Plateau support classic bogs, dominated by *Sphagum* mosses, ericaceous shrubs (e.g., leather leaf, bog rosemary), and black and red spruce, that one finds in New England, the Adirondacks, or the boreal regions of Canada. In the northwestern corner of the MAR, calcareous soils support sedges. The wetland-dependent evergreen species of the Poconos are less common in this ecoregion, where alders, dogwoods, and a diversity of grass-likes and forbs that favor alkaline soils predominate.

Flats occur where water sources are dominated by precipitation and vertical fluctuations of the water table (Brinson 1993). They typically occur in regions of low topographic relief, such as the Coastal Plain, particularly on the outer coastal plain of Delaware, Maryland, Virginia, and North Carolina, and North Carolina. They are represented by pocosins, which tend to have soils with high amounts of organic matter. Other types of flats may have soils containing predominately mineral sediments derived from extensive outwash plains of rivers. There are other ecoregions of the MAR with landscapes that are relatively flat topographically, including parts of the Allegheny Plateau, and the glaciated regions (Fig. 2.1), but hydrologic regimes in these locales are based on a mix of surface flows and soils saturated with groundwater that are more like other wetland types than the characteristic flats. Thus, these wetlands have been classified as a mixture of shallow depressions, low gradient slopes, and riverine floodplains, rather than flats.

We have identified two major types of **slope** wetlands (Stein et al. 2004; Brooks et al. 2011). **Topographic slopes** are those located at the toe of a hillslope where the volume of groundwater discharge is sufficient to support a wetland. **Stratigraphic slopes** are those typically located farther upslope, but they are also found on valley floors, where bedrock contacts form a permeability contrast which allows a discharge of sufficient groundwater to support a wetland. Slopes have unidirectional flow occurring either in braided channels, or across a broader surface. These flows often contain a mix of deeper groundwater, interflow, and surface runoff from precipitation events. They tend to be smaller in area than most other wetlands found in the MAR. Similar to flats, slopes can be further differentiated by dominant soil type, mineral vs. organic.

Wetland **depressions** can vary in area, depth, and permanence of water. They are formed through a variety of geophysical processes. Isolated depressions, by definition, have no surface water connections to other waterbodies. Small (typically <1 ha), isolated depressions are often referred to as temporary, seasonal, or vernal pools. This type is noted for its importance as habitat for breeding amphibians (Calhoun and deMaynadier 2007). Larger depressions can occur anywhere in the landscape where a low-lying area collects and stores water in a shallow or deep, bowl-shaped feature. Those found in the uppermost reaches of watersheds can collect water and release it to headwater streams whenever the outlet elevation is exceeded. Bogs and fens, many of which formed when ice blocks cleaved from retreating glaciers melted in place, occur predominantly in the northern and mountainous portions of the region. Other depressions form in floodplain or valley bottom settings where they may have both inlets and outlets, allowing greater connectivity with the stream or river network. Water levels are maintained by layers of impervious soils that perch the water or by a water table that is high enough to keep the soil saturated. Areas scoured by past flowing water (e.g., oxbows) or wind (e.g., Carolina Bays) forces also form wetland depressions. Dams or excavations, whether created by beaver or humans, can produce either open-water or vegetated depressions, familiarly known as farm ponds, reservoirs, and lakes.

2.2 Inventory

2.2.1 Innovative Approaches to Wetlands Inventory

The MAR was one of the first geographic areas of the United States to produce National Wetlands Inventory quadrangle maps, status and trends reports (e.g., Tiner 1987). Consequently, the aerial photographs on which those original NWI data were based are now over 30 years old. Western Pennsylvania's NWI maps and the subsequent digital data are based on black and white imagery at 1:80,000 scale from the late 1970s and early 1980s. High priority areas of Pennsylvania have received more recent attention. The Delaware River and Lake Erie coastal zones were recently updated with high-resolution imagery (i.e., 2004 NAIP CIR 1 m resolution, 2003–2006 PAMAP True Color 0.3 m resolution, 2005 DVRPC 0.3 m resolution; Pennsylvania Department of Environmental Protection 2011). Despite the recent availability of statewide lidar and digital orthophotographs (<1 m resolution; http://www.pasda.psu.edu), there is no definitive plan to produce a new wetlands inventory for Pennsylvania wetlands away from the two coasts.

Each state has approached their inventories of wetlands in different ways, and independent efforts by agencies, research scientists, conservation organizations, and consultants have produced a set of fragments for which no central repository exists. Inventory efforts around the United States are working on the leading edge of advancing technology to "find" wetlands that may be small in size, hidden under forest canopies, or seasonally wet (e.g., Maxa and Bolstad 2009). So, the wetlands inventory collective for the region consists of mixed media and varied chronology sources that are not universally compatible or accessible. Based on Tiner's (1987) report of total wetland acreage for both inland and coastal wetlands in the region, based on sampling the original NWI aerial photography, the proportion of wetlands by state was Virginia (46%), Pennsylvania (22%), Maryland (19%), Delaware (9%), and West Virginia (4%), for a total of about 800,000 ha.

As Wardrop et al. (2007) and others have shown, the NWI data for the MAR can underestimate the abundance of inland wetlands by almost 50%. In part, this is because many wetlands in the region are small in size, and others are obscured by forest canopies. Thus, finer resolution aerial imagery is not necessarily the only solution. A variety of predictive techniques have been explored to further identify and delineate wetlands. For example, McLaughlin (1999) combined aspects of geomorphology (e.g., faults, contacts) and topography (e.g., changes in slope) to predict likelihood of wetland occurrence in the Ridge and Valley.

Herz (2005) combined GIS-based spatial modeling with field validation to predict locations of groundwater discharges along streams in central Pennsylvania as a means to locate small wetlands dependent upon springs and seeps. She found that three factors, concave curvature of the landscape, underlying geologic structure and composition, and hydric soils, used individually or collectively could predict discharge locations for 60–70% of the sampled field sites. Use of higher resolution topographic data, from lidar could enhance the predictions.

Both of these studies demonstrated the importance of understanding landform and landscape setting to enhance inventories for small, groundwater-supported wetlands.

2.2.2 The Future of Wetlands Inventory in the Mid-Atlantic Region

Given this patchwork quilt of available imagery and interpreted geospatial data for wetlands in the MAR, it is doubtful that a consistent, region-wide wetlands inventory will be sustainably produced. Funding for a comprehensive inventory, by the NWI or other entity, is probably prohibitively expensive. The likelihood that a region-wide product could be produced at predictable intervals (e.g., 5, 10, or 20 years) is low. Yet, the MAR urgently needs to develop an approach that efficiently and periodically produces wetland inventories. We suggest, therefore, that a wetlands inventory for the MAR be built around acceptance of a continuous process, where verified changes to the NWI base layer (or whatever layer is deemed to be the best for each state) be made as they become available. Each state would most likely maintain its own database. For example, wetlands delineated for permit submittals could be routinely provided in digital form to the designated office. Similarly, intensive inventories created for a single watershed or river basin using advanced technologies (e.g., lidar coupled with low-altitude, multispectral photographs) could be submitted. Whenever possible, these individual efforts should follow FGDC Wetlands Mapping Standard (FGDC 2009) (http://www.fgdc.gov/standards/projects/FGDC-standards-projects/wetlands-mapping). Whatever is used, however, should be properly documented with appropriate metadata and be served from publicly accessible, web-based databases. At a minimum, a technically proficient, two-person team could provide this service for a state for a reasonable investment. This inventory team could be based within an existing resource agency, an organization, a university, or a private firm, as long as the funding sources and mechanisms were sustainable, and access was assured.

With this approach, we might not gain a uniform dataset acquired during a narrow window of time that would reflect the entire region, but we would have a continuously updated database that could provide the best available inventory data for a geographic area of interest. Since most decision-making involving changes in land use takes place at spatial scales that are relevant to a small watershed, municipality, or county, we would be assured that the most recent wetlands inventory data would be available for those users. Areas of high priority due to intense development pressures or identified as desirable for protection might receive more frequent attention, but all regions would likely be updated more often than if we wait for a single, region-wide effort.

2.2.3 Status and Trends

Overall, in 2009, there were an estimated 44.6 million ha of wetlands in the conterminous United States, with 95% being freshwater types (Dahl 2011). The average year for imagery used was 2009. During the study period, 2004–2009, there was slight decline in area overall, in contrast to the previous report which showed a slight gain (Dahl 2006). Most of these losses and gains can be attributed to land use trends, successional changes, and variations in effectiveness of regulatory and non-regulatory programs. Regardless of the causes of these variations, it appears that during the past decade, we are beginning to meet the goals emerging from the National Wetlands Policy Forum of no net losses and long-term gains (National Wetlands Policy Forum 1988). This seminal meeting held in 1987 was convened by the Conservation Foundation at USEPA's request, and set the stage for concerted

Table 2.2 Wetlands losses by state for the Mid-Atlantic region (includes inland and coastal types; data from Dahl 1990; modified from Mitsch and Gosselink 2007)

	Original estimated area (ha)	National wetlands inventory (ha)	Change
State	Circa 1780	Mid-1980s	(%)
Delaware	194,000	90,000	−54
Maryland	668,000	178,000	−73
New Jersey	607,000	370,000	−39
New York	1,037,000	415,000	−60
North Carolina	4,488,000	2,300,000	−44
Pennsylvania	456,000	202,000	−56
Virginia	748,000	435,000	−42
West Virginia	54,000	380,000	−31
Mid-Atlantic region	8,252,000	4,370,000	−47

political, scientific, and grassroots efforts to stem the losses of wetlands that occurred since European settlement, and to begin to offset those losses with gains.

Because NWI's most recent analyses of losses and gains were conducted for the entire nation, the trends for individual states or regions are not available. The most recent report for the MAR providing state data comes from Dahl's (1990) estimates of losses from the colonial period of the 1780s through the mid-1980s (Table 2.2). Since that time, losses in area have declined nationally and regionally, primarily due to strong regulatory programs at both federal and state levels, and significant non-regulatory programs that provide incentives to private landowners to protect and conserve their wetlands.

2.3 Reference Wetlands

2.3.1 Concepts of Reference Wetlands

The use of reference sites has become increasingly more common as scientists and resource managers search for reasonable and scientifically based methods to measure and describe the inherent variability in natural aquatic systems (e.g., Hughes et al. 1986; Kentula et al. 1992). We use the term reference wetlands to connote naturally occurring sites composed of wetland, stream, and riparian components that span a gradient of anthropogenic/human disturbance. Although reference sites often represent areas of minimal human disturbance (i.e., reference standards in HGM parlance; Smith et al. 1995), in many instances it is more useful to represent a range of environmental conditions across a landscape (Karr and Chu 1999; Brooks et al. 2006).

The primary reasons to include reference sites in regional assessments and restoration efforts are the need to compare impacted or degraded sites to a least-impaired set of attributes or benchmarks. These benchmarks can represent a starting point in time for trend analyses (e.g., long-term successional studies or impact analysis on a group of wetlands). Reference sites can also serve as alternatives to standard experimental controls, which are seldom available in large-scale field studies. The primary criterion for selecting reference sites involves choosing sites that represent ideal, relatively natural conditions represented by the least disturbed sites available, which is common for stream assessments (Karr and Chu 1999). Sites can be chosen to represent the best attainable conditions for a particular region even though they may not be pristine (Smith et al. 1995). This approach has been adopted in the MAR by several states, in part, because there has been an intentional process to use common approaches and methods.

Sites within the reference set can span several gradients. They should include, at a minimum, the common types of wetlands found within a region, and the range of conditions from relatively pristine (ecologically intact) to severely disturbed sites (degraded ecological integrity and functions). This will provide the data necessary to assess and rank the condition for the full range of sites that are being assessed (Brooks et al. 2004).

Given limited human and financial resources, creating a pool of reference wetlands that satisfies multiple objectives is desirable. Investigators and managers must decide jointly upon the acceptable level of analytical compromise they can tolerate vs. the advantages of shared data and resources. Most studies will benefit from some overlap among sets of reference sites. Using reference wetlands from a wide variety of vegetation types, disturbance regimes, and landscape positions allows for characterization of this variability. Once established, reference wetlands can be used to set the standard by which mitigation and management projects (restoration, creation, or enhancement) can be judged.

Since the early 1990s, universities, agencies, and organizations throughout the MAR have assembled a growing set of reference wetlands. Other wetlands have been studied by numerous investigators, but here we distinguish wetlands purposefully established as a set and monitored to provide condition assessments and/or comparative data for mitigation projects. From 1993 to 2003, the Penn State Cooperative Wetlands Center (now Riparia) established a set of 222 reference wetlands for Pennsylvania (Table 2.3, Fig. 2.3). They were chosen based on three criteria; accessibility for multiple years, commonly impacted types, and random selections. The following protocol was used to standardize the selection procedure. Candidate sites were selected at random from a regional pool of sites developed from US Geological Survey topographic maps, NWI maps, and Natural Resource Conservation Survey soil surveys. Potential wetlands of the desired type (e.g., public land, vegetation type, size class, degree of disturbance) occurring in 1 km × 1 km blocks (UTM grid) on NWI maps in the vicinity of the target population were numbered and placed in a pool of potential sites. To obtain an adequate distribution geographically, several potential sites from each of 5 to 10 topographic maps within an ecoregion were chosen. Each was checked during a site visit, and if permanent access from landowners was available, the site was selected for the reference set. Wetlands in the reference set used for the Upper Juniata Watershed study (about one-third of the total) were selected using USEPA's generalize random-tesselation

Table 2.3 Summary of reference wetlands sampled in Pennsylvania, 1993–2003 ($n=222$) by ecoregion and HGM subclass

	Glaciated plateau	Piedmont	Glaciated poconos	Allegheny plateau	Ridge and valley	Total
Lacustrine permanently flooded (fringing)	0	1	1	5	10	*17*
Riverine upper perennial	2	2	2	7	52	*65*
Riverine beaver-impounded	0	0	1	3	7	*11*
Depression, seasonal or temporary (isolated)	0	0	2	0	15	*17*
Riverine lower perennial and floodplain complex	3	3	4	9	24	*43*
Riverine headwater complex or depression perennial	0	0	8	2	16	*26*
Slope	3	3	6	12	19	*43*
Total	*8*	*9*	*24*	*38*	*143*	*222*

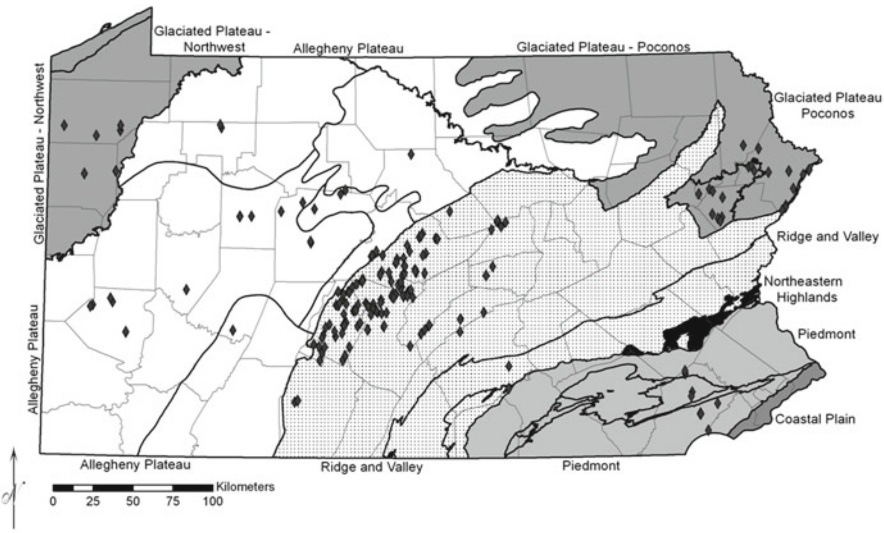

Fig. 2.3 Distribution of reference wetlands by ecoregion in Pennsylvania, as established from 1993 to 2003 by Riparia at the Pennsylvania State University

stratified (GRTS) design (Stevens and Olsen 2004), which assures a spatially well-balanced sample. A full listing of sites, their location and characteristics appears in Brooks 2004, Table 2, and at http://www.riparia.psu.edu/MARbook.

When bounding reference wetlands, it will be necessary to truncate wetland complexes and mix types, particularly in long, linear, riverine systems. Riparia selected sites in the range of about 0.25–3.0 ha, with most being about 0.4 ha.

Sampling is typically conducted along the hydrologic gradient. This approach has been used elsewhere in the MAR, but is not universally applied.

Reference collections need not be restricted by state borders. For example, to increase the number of reference wetlands in the Glaciated Plateau of Pennsylvania, we arranged to share data with the Ohio Environmental Protection Agency for adjoining ecoregions along our common border with Ohio (J. Mack, pers. comm.). These types of collaborative arrangements among states can help efficiently generate regional sets of reference wetlands for use by multiple groups.

2.3.2 Recommended Steps for Establishing a Regional Set of Reference Wetlands

Based on our experience, the following steps for establishing a regional set of reference wetlands are recommended (Brooks et al. 1996, 2002). It is assumed that one of the primary uses of the reference set will be to classify wetlands and develop functional models using the HGM approach, but that other needs will be met by the same set.

1. Identify the need and goals for establishing reference wetlands in a specific ecoregion or set of ecoregions that are similar.
2. Choose a multi-organizational regional assessment team with the necessary expertise to assess the types of wetlands in the given region.
3. Assessment team core members must commit to repeated meetings and field visits to establish the reference set. Auxiliary team members can come and go as needed and as available to expand the realm of expertise.
4. Ideally, the assessment team should range from 5 to 10 members (minimum of 3, maximum of 12). This will provide sufficient expertise while still allowing the group to develop as a cohesive unit. Presumably, all or a portion of the assessment team will be involved in aspects of characterizing (modeling subclasses for HGM approach) the reference set.
5. The assessment team should be provided the Corps HGM documents as a starting point (e.g., Brinson 1993, Smith et al. 1995, regional HGM models).
6. The assessment team members should conduct a series of 1-day seminars on HGM concepts, classification, and functions for potential stakeholders in the region. This will explain the rationale and methodology for establishing reference wetlands, as well as introduce potential users to the HGM approach.
7. It is useful to discuss potential regional changes in the national HGM classification system for the region of concern and conduct several field visits to multiple types of wetlands until the assessment team consistently recognizes and agrees upon classification of most sites.
8. At some point, it will be necessary to determine whether all or only some HGM subclasses will be considered. Wetland types to be investigated can be prioritized by potential threats, relative abundance, or available expertise.

9. We recommend that the assessment team identify a pool of wetlands at least 2–3 times the desired number of reference sites targeted for detailed characterization to account for access problems. To ensure adequate spatial coverage to facilitate post-monitoring analyses, we suggest organizers review and consider use of the GRTS approach to survey design, as utilized in USEPA's National Aquatic Resource Surveys (Stevens and Olsen 2004) (http://www.epa.gov/nheerl/arm/designing/design_intro.htm).
10. Further cautionary notes regarding selection of reference wetlands:
    (a) Consider all needs for reference sites, not just for HGM functional assessment.
    (b) One cannot always examine a statistically valid sample for each wetland type or HGM subclass; our rule of thumb is to use three wetlands as the absolute minimum per subclass, 30–50 is probably a maximum, and 8–12 begins to cover the variability in a subclass; Smith et al. (1995) suggest a minimum of 20.
    (c) Sites can be chosen based on proportions of NWI types, or types of special concern.
    (d) Sites should have long-term accessibility, which suggests public ownership, yet the reference set must cover subclass variability, including disturbance, which probably will require that some sites be on private lands subject to typical land use and management activities.
    (e) A subset of the total reference set should meet the requirement of long-term accessibility; this subset should consist of representative/typical wetlands.
    (f) Once selected from the pool, secure written permission that acknowledges the probable sampling protocol and access procedures.

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Chapter 3
Linking Landscapes to Wetland Condition: A Case Study of Eight Headwater Complexes in Pennsylvania

J.B. Moon and Denice Heller Wardrop

Abstract A major focus of wetland management is on documenting condition, identifying stressors, and determining the relationship between the two. One challenge of this work is to relate and understand the pathways between stressors that are being controlled at the landscape scale to microbially mediated ecosystem processes (e.g., nitrification, denitrification, decomposition, methanogenesis) happening at much smaller spatial extents. In this chapter, we use a preexisting multiple-stress function (alternately termed an anthropogenic or human disturbance indicator) to select study sites for comparison of condition metrics. More specifically, this is a case study of eight headwater complexes, four reference standard sites, and four "stressed" sites, where landscape and site level structural components are compared and contrasted. The site level components that were selected for this analysis were those known to influence microbially mediated ecosystem processes. We reveal many significant differences in these groups for both landscape parameters across space and time, and in site level components. In addition to collecting process-based data, we suggest that connecting structural baseline data such as those parameters described herein with process-based models, as a way to begin to hypothesize what the collective effects of wetland components are on microbially mediated ecosystem processes, as well as to understand which components are most influential. With this understanding it becomes easier to link processes to landscape-driven stressors, through preexisting knowledge about the links between stressors and wetland structure. We also suggest shifting some attention to spatiotemporal dynamics of the stressor(s), in order to determine the feasibility in managing, restoring, and/or protecting wetland ecosystems.

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3.1 Introduction

The relationship between anthropogenic stressors and ecological condition forms the platform of most current day environmental management programs, and can be viewed by stress response curves. Ecological condition is expressed on the *y*-axis. It describes the extent to which a given site departs from the full measure of ecological integrity that is possible in a region, and is defined by the least-impacted or reference condition. It can be measured in terms of structure (e.g., community composition and biomass), form (i.e., the arrangement of ecosystem components, which helps define how they interact), or process (e.g., primary productivity) (see Chap. 11).

The *x*-axis is expected to be an appropriate expression of anthropogenic stress that is uniquely tied to the structure, form, or process of interest. Historically, these conceptual models have been most rigorously tested and refined in aquatic ecosystems, most notably wadeable streams. Here there has been abundant attention paid to various measures of ecological condition (Davies and Jackson 2006) and the relationship to a common measure of stress. While finding a specific stressor (e.g., sedimentation, nutrient enrichments) that relates to an ecological condition might be the desired approach, because of the possibility of direct correlations to Best Management Practices (BMPs), the complexities of wetland systems (e.g., feedbacks between vegetation, hydrology, and the physiochemical environment) hinderbacks this overly simplistic approach. In addition, biological assemblages are generally subjected to a variety of stressors simultaneously, and thus, an integrated approach might be more representative of real-world conditions. Therefore, a viable alternative is the development of an indicator of multiple sources of stress, termed herein as a multiple-stress function. Multiple-stress functions have been used in survey designs for the development of ecological indicators (e.g., Danz et al. 2007), and are generally defined by land-based measures (e.g., land cover types, occurrence of point sources of pollution).

Once the ecological condition measure and anthropogenic stressor(s) are chosen, the focus turns towards the shape of the relationship, since the path the condition measure takes along a gradient of increasing stress has meaning to management approaches (Fig. 3.1a). For example, the Tiered Aquatic Life Use approach utilized in many state stream programs (e.g., Maine and Ohio) assumes a linear decline in condition with increasing stress; restoration approaches seek to move systems "up" the curve by decreasing stress. However, the appearance of threshold relationships between stressors and condition measures calls for an alternative approach. Under this scenario, we expect to maintain the stress level to the left of the threshold, as the resources required to restore the system are substantial once the threshold is crossed. Additional relationships have been theorized, such as alternative stable states that are based on hysteretic effects, but have not been widely proven for wetland ecosystems to date (Beisner et al. 2003).

For over a decade Riparia, at the Pennsylvania State University, has focused its attention on developing a multiple-stress function and assessing its relation to an array of ecological condition measures for wetlands within the Mid-Atlantic region

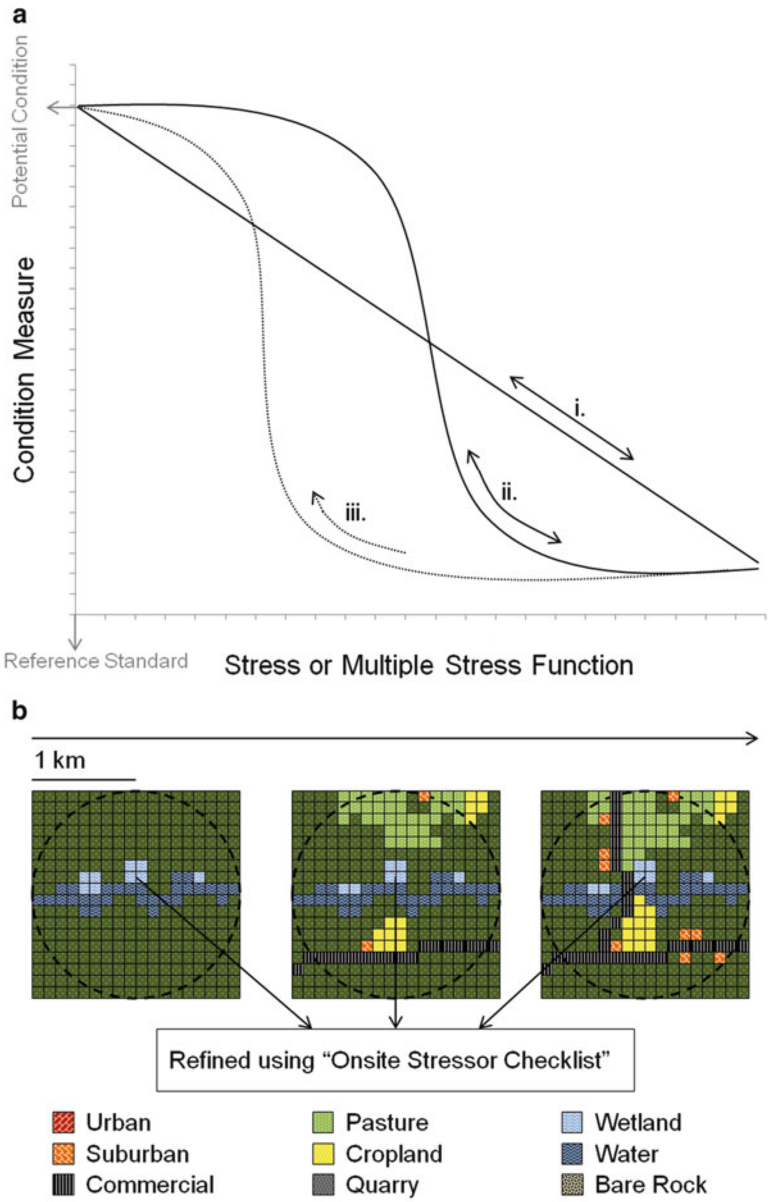

Fig. 3.1 A conceptual model of the stress–response relationship used in management practices. (**a**) Depicts examples of (i) linear and (ii) threshold condition responses to a stress or multiple stress function with either linear or (iii) hysteretic restoration condition responses. (**b**) Depicts a multiple stress function created for wetlands in the Mid-Atlantic region. This function is based on measures of percent forest in a 1-km circle centered on the site, breaks or punctures in the forest buffer immediately surrounding the site, and onsite stressors

(MAR) (Fig. 3.1b). In these studies, the *y*-axis has been commonly defined in terms of structural components of biological communities, such as Indices of Biological Integrity (IBI) applied to birds, plants, amphibians, wildlife, and macroinvertebrates, or in terms of functional components, such as Hydrogeomorphic (HGM) Functional Assessment models. IBI and HGM approaches are described in detail in Chaps. 2 and 11 of this book and elsewhere in the literature.

Similar to others, Riparia researchers first looked to broad land-based indicators of stress (i.e., readily available land cover data) to develop their multiple-stress function (Fig. 3.1b). They asked the question, "In this context, what is reference standard?" (Brinson and Rheinhardt 1996; Rheinhardt et al. 1997, 1999). Forested land cover was accepted as reference standard, as it was the predominant land cover class in the MAR during precolonial times. More specifically, in Pennsylvania only 2–3% of land cover was of non-forested land cover classes (Schein and Miller 1995). Today, mature forests are considered the least altered landscapes in Pennsylvania, while agricultural and urban landscapes are considered the most significantly altered by human activity.

While the amount of forested land cover surrounding sites had demonstrated utility as an indicator of good condition in early studies (Chap. 11), later field visits to forested sites revealed the occurrence of a large range of stressors, as defined by Adamus and Brandt (1990) and Adamus et al. (2001), which were "under the trees." These stressors appeared to be related to decreases in various measures of wetland condition. What Riparia researchers desired, then, was a multiple-stress function that took into account both the context of the site (i.e., utilizing land cover as a proxy for a family of potential stressors), as well as what was occurring at a site-specific scale that was unseen by land cover data.

A conceptual model was constructed, viewing wetlands as cells, wherein flows of energy and materials—stressors—from the surrounding landscape could potentially enter the cells through their membranes (i.e., buffers surrounding the wetlands). The impact that the surrounding landscape could exert on a wetland was a function of the number of different stressors and the nature of the membrane/buffer. This conceptual model was represented by the following equation, where disturbance is calculated as:

$$\text{Disturbance Score} =$$
$$100 - \text{CF}\left\{\%\text{FLC}\left[10 - \frac{\#\text{Stressors}}{10}\right] + [\text{BufferScore} - \text{BufferHits}]\right\}$$

where CF is a calibration factor (100/114) that standardizes the score on a scale of 0–100, %FLC is the percent of forested land cover in a 1-km radius circle centered on the wetland, #Stressors is the number of stressor categories present on the wetland,[1]

[1]The Penn State stressor checklist can be found in Wardrop et al. (2007a, b) and Brooks et al. (2006). The Disturbance Score, when reversed, can also provide a measure of condition (Chap. 11, and Wardrop et al. (2007a, b)).

BufferScore is a value from 0 to 14 assigned to the buffer given its type and width, and BufferHits are the number of stressors that "puncture" this buffer. The disturbance score ranges from 0 to 100. Zero predicts no anthropogenic disturbance, while 100 predicts a completely impacted site.

This multiple-stress function, the disturbance score, has proven useful in identifying a gradient of inadvertent effects from the surrounding landscape on the condition of an array of biological structural components in headwater complexes (O'Connell et al. 2000; Laubscher 2005; Miller et al. 2006). However, we seek to go a few steps further in its use as a precursor to management and policy utility. First, while a multiple-stress function can be used to describe a general correlation between human activity and ecological condition, it "black boxes" the complicated and interrelated stressor–response relationships and restricts our landscape scale of analysis and management to 1-km aerial extents. In other words, it does not describe the precise linkages between land cover actions, individual stressors, and ultimate impacts on ecological condition. Thus, it gives us little direction in knowing what can be done to improve condition(s) and at what scale the management must be performed.

Second, while relationships between the multiple-stress function and biological structural components have been well-established, fewer direct measures of ecosystem processes, particularly those mediated by microbial communities (e.g., nitrification, denitrification, carbon mineralization), have been studied across this gradient of disturbance. The sequestration and release of carbon and nitrogen in soils, through both natural and anthropogenic pathways, has taken on increased significance in recent years due to the impact these processes have on the global climate (IPCC 2001) and water quality. Wetlands provided humans with an array of ecosystem services, including the regulation of both climate and water quality through supporting services such as microbial biochemical cycling, which is "the storage, recycling, processing and acquisition of nutrients" (Millennium Ecosystem Assessment 2005). Thus, potential alterations to these microbial processes in wetlands, through the effects of anthropogenic stress, create cause for concern and warrant further investigation.

In this chapter our aim was to begin to open the "black box," using a case study to describe both landscape structural components, past and present, at multiple scales and an array of onsite condition measures. The condition measures were selected to specifically represent structural components of wetlands (i.e., microtopography, temperature, hydrology, vegetation, soils) linked to biogeochemical cycling, which in turn can help focus future studies of microbial processes. Unfortunately, this case study does not allow for evaluations of the condition measures across the full range of the multiple-stress function, but rather compares and contrasts landscape and site level structural components of the lowest (n=4) and the highest (n=4) levels of the disturbance score multiple-stress function. For this purpose we selected eight sites from the Riparia wetland reference site collection (n = 222). Hereafter these disturbance score groups will be referred as reference standard or stressed. We compiled landscape data for these study sites from the late 1800s through 2005 and we collected site level data between 2006 and 2011.

3.2 Case Study

3.2.1 Study Sites

During the site selection process, we controlled five variables (i.e., wetland type, physiographic province, geology, soils, hydrologic unit) (Table 3.1). First, we chose to focus our analysis on headwater riverine wetlands (classification as per Brooks et al. 2011, see Chap. 2), as they are the predominant wetland type within the region. Headwater riverine wetlands are also one of the least well-studied freshwater forested wetland types in the USA, in terms of both losses and current conditions (in contrast to bottomland hardwoods, cypress swamps, etc.). As small forested systems, they can sometimes be overlooked during National Wetland Inventory (NWI) assessments, which use satellite imagery to identify wetlands in the landscape (Dahl 2006).

However, their importance to the connected landscape cannot be ignored. Although small, they function to buffer a number of earth system processes (e.g., climate change and water quality—Millennium Ecosystem Assessment 2005) that are close to or already have passed thresholds of unacceptable environmental change (Rockstrom et al. 2009). For example, they lie adjacent to low order headwater streams, which account for 60–75% of the total stream and river miles in the USA (Leopold et al. 1964) and tend to set the biogeochemical state of the downstream networks. Through biogeochemical cycling of large portions of the global carbon and nitrogen pools, they act as sinks for and transformers of inorganic nutrients and as sources of organic material (sometimes inorganic, see Noe and Hupp 2007) to aquatic systems during flooding events. Due to their geographic position and periodic fluvial disturbances, these ecosystems can be "hotspots" of habitat and biotic diversity and productivity.

We used a regional HGM classification system developed for the MAR (Brooks et al. 2011) to select sites. The HGM system is based on geomorphic setting, source of water, and hydrodynamics (Brinson 1993). The overall HGM system, modified from Brinson (1993), recognizes seven major classes: Mineral Soil Flat, Organic Soil Flat, Slope, Depression, Lacustrine Fringe, Riverine, and Tidal Fringe (Marine and Estuarine) (Smith et al. 1995). These can be further divided into regional and local subclasses. We were interested in subclasses that described headwater wetlands. The classification system for the MAR has three such subclasses under the Riverine class: intermittent, headwater complex, and upper perennial.

The headwater complex subclass represented an addition to previous HGM classifications for the region, as a response to field conditions encountered during an assessment of wetlands in the Upper Juniata (Wardrop et al. 2007a, b). It represents an attempt to capture a frequently occurring mosaic of jurisdictional[2] wetland patches, generally supported by groundwater, that may fill the entire bottom of a small valley or occur as a substantial ribbon-like area along first, second, and third order streams. A common scenario is one in which groundwater may emanate from the toe of a slope, fill in depressions in the riparian zone, and provide water to

[2] Jurisdictional wetlands are those wetlands that are regulated by the U.S. Army Corp of Engineers under Section 404 of the Clear Water Act.

Table 3.1 Characteristics of the eight headwater complex study sites, including: geographic location, hydrogeomorphic (HGM) classification, National Wetland Inventory (NWI) classification, geology, soil order, soil series, and disturbance scores

			Classification					
Site[a]	Latitude	Longitude	HGM[b]	NWI[c]	Geology[d]	Soil order[e]	Soil series	Disturbance score[f]
83	40.3592	−77.7933	UP	PFO1/4A	Clinton Group	Inceptisol	Atkins silt loam	76
188	40.6633	−78.3339	UP	L1UBHh	Catskill Family Undivided	Inceptisol/entisol	Udifluvents dystrochrepts complex	82
60	40.7028	−77.8486	UP	PEM1A	Bloomsburg Fm/ Mifflintown Fm	Inceptisol/ultisol	Atkins silt loam/buchanan, extremely stony loam	94
13	41.0317	−77.1092	RD	BUBHh	Reedsville Fm	Inceptisol/entisol	Udifluvents and fluvaquents, gravelly	98
140	40.4061	−78.4445	UP	POWZ	Hamilton Group	Inceptisol/entisol	Udifluvents dystrochrepts complex/ monogahela silt loam	33
158	40.4781	−78.2895	UP	PSS1/EMY	Hamilton Group	Inceptisol/ultisol	Holy silt loam/ernest silt loam	33
151	40.6069	−78.0137	UP	PEM1A	Bloomsburg Fm/ Mifflintown Fm	Inceptisol	Philo and Basher silt loam/Basher silt loam	30
124	40.4539	−78.1104	RD/UP	PSS1/EMY	Bloomsburg Fm/ Mifflintown Fm	Inceptisol	Atkins silt loam	25

[a]Site numbers are identification numbers from the Riparia reference collection: http://wetlands.psu.edu/projects/reference_wetlands.asp
[b]HGM classifications include: *UP* upper perennial and *RD* riparian depression
[c]Descriptions of NWI Classifications can be found in Cowardin et al. 1979
[d]Geology was taken from PA DCNR, http://www.dcnr.state.pa.us/topogeo/map1/bedmap.aspx and described in http://www.dcnr.state.pa.us/topogeo/map1/polyattrib.pdf
[e]Soil orders and series were taken from USDA NRCS, http://soildatamart.nrcs.usda.gov/Survey.aspx?State=PA
[f]Disturbance scores were calculated based on Brooks et al. (2006)

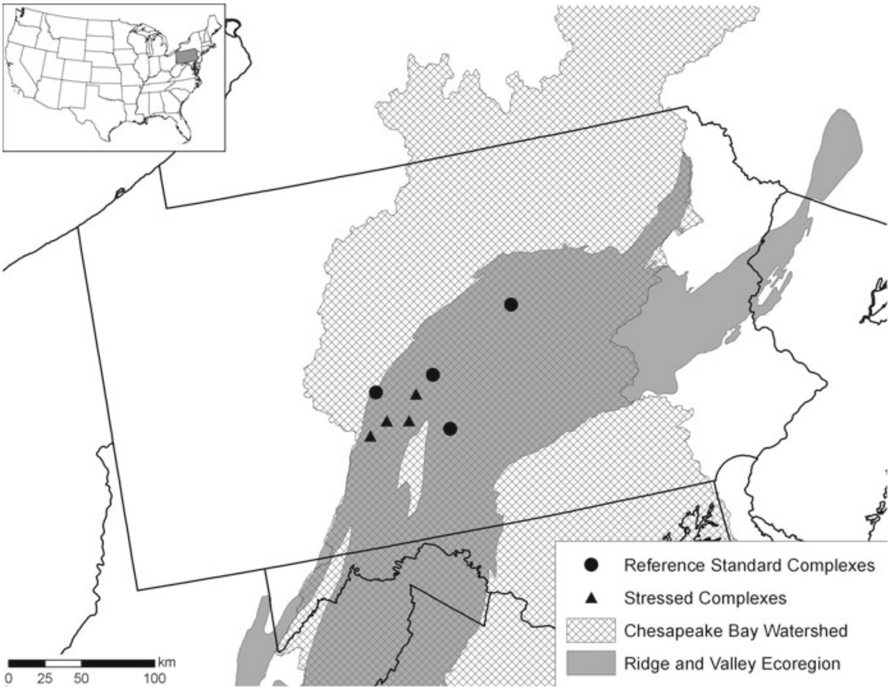

Fig. 3.2 Geographic locations of the headwater complex study sites. Study sites were located within the Chesapeake Bay Watershed portion of the Ridge and Valley ecoregion of central Pennsylvania

a low-gradient stream, often within a relatively small area (approximately <1 ha in size). At the wetland patch scale these three areas could be separately identified as slope, depression, and upper perennial wetlands, respectively (Brooks et al. 2011). However, this mosaic of wetland patches, along with the interspersed patches of floodplain (i.e., upland) function as an integrated wetland ecosystem, and are best viewed as a unique subclass. It should be noted that, by the very definition of the headwater complex subclass as a mosaic, there can be variation in which wetland patch types play a more dominant role. The final eight sites were classified as headwater complexes, with six dominated by upper perennial elements, one dominated by depression elements, and one containing characteristic traits of both.

All selected sites were located within the Appalachian Mountain Section of the Ridge and Valley ecoregion of central Pennsylvania (Fig. 3.2). Over half of the wetlands in Pennsylvania are located within the northeastern and northwestern portions of the Commonwealth (Bushell 1989). Wetlands are concentrated in these regions due to Pleistocene glaciations (Bushell 1989). Although smaller in extent, wetlands do exist in the unglaciated regions of the Commonwealth, such as in the Ridge and Valley physiographic province. This physiographic province consists of folded and faulted Paleozoic rocks, consisting of sandstone ridges with a predominance of shale and siltstone valleys. However, some valleys are underlain by limestone and dolomite (PA DCNR, http://www.dcnr.state.pa.us/topogeo/map13/13ams.aspx).

Headwater complexes chosen for this study had shale and siltstone primary and secondary lithologies, respectively (Berg et al. 1980, PA DCNR, http://www.dcnr.state.pa.us/topogeo/map1/polyattrib.pdf). One exception was Site # 83, which had a limestone secondary lithology.

Within the Ridge and Valley ecoregion, elevation ranges between 134 and 846 m above sea level (PA DCNR, http://www.dcnr.state.pa.us/topogeo/map13/13ams.aspx). Most wetlands are located within the valleys as part of the Susquehanna River network in the Chesapeake Bay Watershed (Bushell 1989). This network lacks large floodplain areal extents, and as such wetlands tend to be predominately within the headwater portion of the valleys (Bushell 1989). The headwater complexes used in this study were no exception, located in the valleys and spanning a range of elevations between 213 and 395 m above sea level.

All headwater complexes also had soils with hydric ratings based on the Natural Resources Conservation Service (NRCS) descriptions (USDA NRCS, http://soildatamart.nrcs.usda.gov/Survey.aspx?State=PA). These soils were alluvial in origin and documented to be of a silt loam texture class. A field survey was conducted in July of 2011 to verify NRCS soil series maps. Finally, all complexes were situated in distinct hydrological landscape units (HUC 14 coordinates).

3.2.2 Methodology

3.2.2.1 Landscape Structure

Topography

We obtained topographic relief and mean slope in 100-m, 1-, 2-, and 5-km radius circles around each study site. The analysis was conducted in ArcMap Version 10.1 (Environmental Systems Research and Inc 2010) with digital elevation models (DEMs) from the National Elevation Dataset (NED) (USGS Seamless DataServer, http://seamless.usgs.gov/). This dataset has a resolution of 1 arc s (~30 m) and a vertical accuracy of 7–15 m.

Historical Land Cover

We obtained historical land cover information, at and around study sites through county atlases and historical aerial photographs. Atlas surveys from the late 1800s (Beers 1868; Nichols 1873) provided information on topography, and the presence of roads, residences, mills, furnaces, and mines upstream of the study sites. Aerial photographs between 1937 and 1942, between 1957 and 1962, and between 1967 and 1972 provided information on agricultural practices and other land cover classes through the past century. We obtained these aerial photographs from Penn Pilot Historical Photographs of Pennsylvania, sponsored by the Pennsylvania Geological Survey (PA Geological Survey, http://www.pennpilot.psu.edu/).

Additional orthoimages were captured for 2005 (PA DCNR, http://www.pasda.psu.edu). We used these surveys and photographs to observationally identify major changes to each study site and its surrounding land cover over time.

Current Land Cover

We conducted an analysis of the current land cover surrounding each study site using 2005 land cover data (Warner et al. 2005). Land cover data were analyzed (Bishop and Lehning 2007) in ArcView Version 3.3 with Spatial Analyst Extension (Environmental Systems Research and Inc 2002). The following land cover classes were identified: suburban, urban, commercial, quarries, pastures, row crops, bare rock, forest, water, and NWI wetlands.[3] We calculated the percentage of each land cover class in 100-m and 1-km radius circles and for successive buffer strips (i.e., 100–200 m, 200–400 m, 400–600 m, 600–800 m, and 800–1,000 m) surrounding each study site.

Additional land cover metrics were calculated, including the percent of core forest (i.e., forest cover >100 m from alternative land cover classes), the percent of impervious surface (including roads and set proportions for land cover classes with impervious surfaces), and the land development index (LDI). The LDI is a land use-based index used to estimate potential human impact by weighting the potential use of each land cover class (Brown and Vivas 2005). Although wetlands are generally classified as having no potential human impact, a large NWI wetland existed at Site # 124, as part of a pasture. Therefore, the LDI score for the 100-m radius circle surrounding the site was adjusted to account for this land cover. The LDIs calculated for buffer strips did not take into account any NWI wetlands on alternative land covers.

3.2.2.2 Headwater Complex Structure

Study Area

To perform their important biogeochemical processes, headwater complexes act in concert and are highly dependent on their wetland and floodplain patches, the adjacent upland, and the adjacent and upstream channels (Brooks et al. 2006, 2009). Thus, measurements were made on the entire headwater complex, rather than solely on the wetland patches within the complex. In addition, because headwater complexes can run continuously along extended lengths of a stream, we established an arbitrary 3,994-m^2 study area for sampling at each study site (Fig. 3.3). The areal extent of the sampling plot was selected by averaging the sites' maximum distances perpendicular to the stream that could be achieved before shifting into surrounding land cover classes with no wetland patches. Site # 140 was sampled at the same grain size, but within a smaller areal extent (1,815 m^2), due to large tree debris that

[3]For more information on mapped NWI wetlands see U.S. Fish and Wildlife Service, http://www.fws.gov/wetlands/Data/Mapper.html.

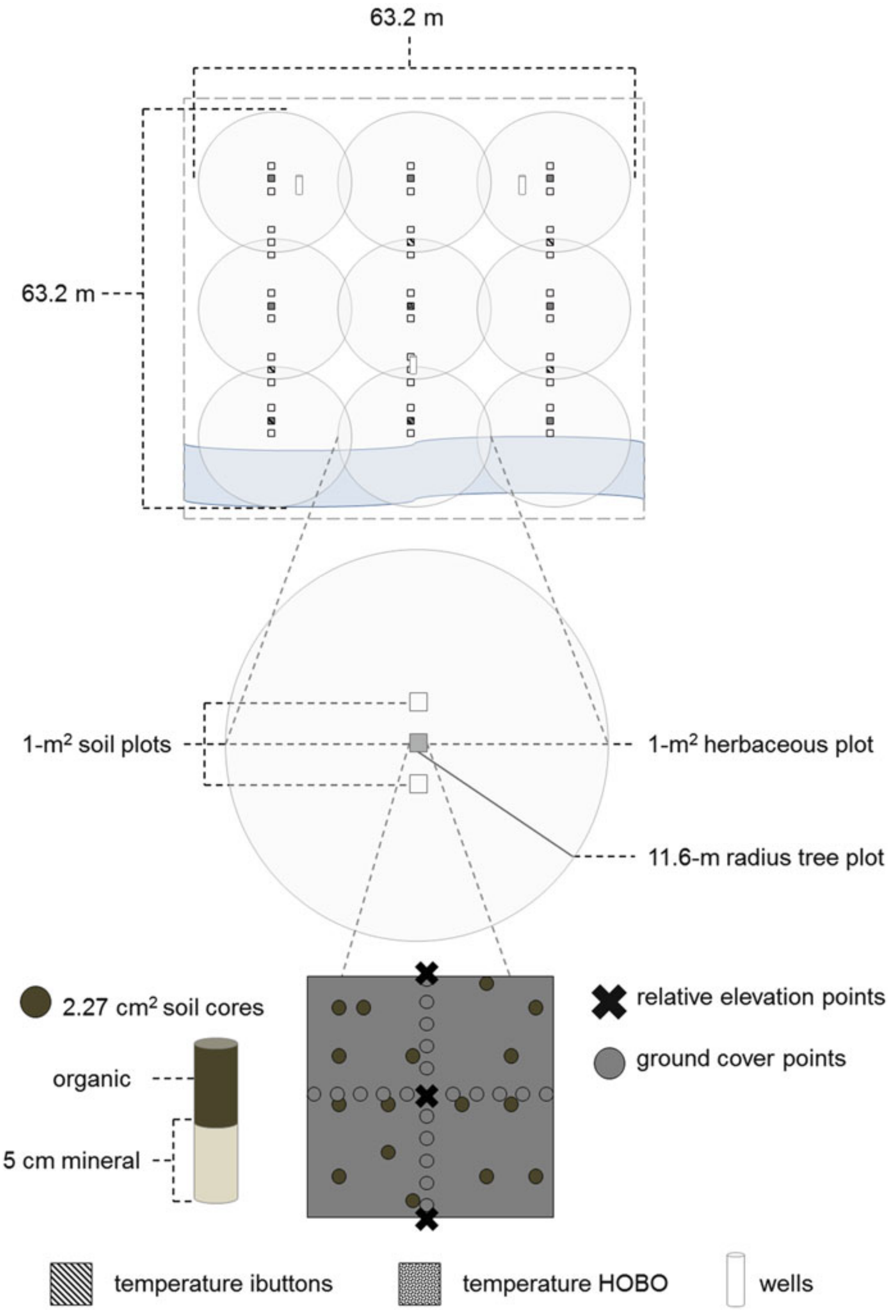

Fig. 3.3 Sampling scheme for measurements collected at headwater complex study sites

prevented access to sampling points. At each study site, we established three transects to collect measurements. Transects were 20 m apart and ran perpendicular to the main stream channel.

Hydrology

To assess hydrology, we compiled datasets that were collected from these study sites on automated Ecotone™ and Ecotone™ WM well level recorders (Remote Data Systems Inc. Wilmington, NC) between 2005 and 2011. Although we made attempts to collect data from all study sites within this period, bear activity and issues with well battery casings prevent us from collected complete datasets. Only the most complete datasets are discussed herein. Water level data from one well, located on the center transect ~10 m from the stream, was available for 6 month periods (i.e., August–January) during 3 years (i.e., 2005, 2006, 2007) at Site # 60 and Site # 83, during 2 years (i.e., 2006 and 2007) at Site # 158, and during 1 year (i.e., 2006) at Site # 151. Additional water level data was available for two or three wells installed in an equilateral triangle (~30 m apart) at Site # 60, Site # 13, Site # 158, and Site # 151 between July 2010 and August 2011.

We summarized water level data for these five study sites using three hydrologic moisture regimes (inundation = >0 cm, saturation = 0 to −30 cm, dry = < −30 cm) described by Cole et al. (2006). However, caution should be taken when interpreting these results. Wardrop et al. (Unpublished Data) recorded variability in water table depths using multiple wells within headwater complexes. A single well does not take into account spatial variation in soil properties (e.g., soil bulk density), microtopography, and/or water sources that could lead to differences in water table summary variables across a study site. The lack of water table depth information across complete annual cycles for data collected between 2005 and 2007 was also a concern.

Due to the potential for high spatial variability in water table depth, we also measured the occurrence rate of different intensities of water table fluctuations. These measures were assumed to be more stable across a study site. We calculated these parameters using the percent of time during the 6 or 12 month period that the water table had fluctuated by 10, 20, and 30 cm across one time step (1 day).

Vegetation

To describe the vegetation at each study site, a combination of datasets were used from past Riparia assessments and from data collected during the summer of 2006. Prior to this study all headwater complexes were assessed (in 1994, 1997, 1999 or 2000) for use in the development of various indices of biotic integrity (e.g., Miller and Wardrop 2006; Miller et al. 2006; Wardrop et al. 2007a). During these assessments, site level adjusted floristic quality index (FQAI) scores (I') were determined. The FQAI is a measure of ecological conservatism of the site's native plant species. It was originally developed by Swink and Wilhelm (1994) and modified to reduce sensitivity to sites with higher numbers of native species by Miller and Wardrop (2006). I' is calculated using the following formula:

$$I' = \left(\frac{\bar{C}}{10} \frac{\sqrt{N}}{\sqrt{N + A}} \right) \times 100$$

where $\bar{C}$ is the mean of the coefficients of conservatism (i.e., COC values = measures an individual species fidelity to a specific habitat) for the plant species at the site, N is the number of native species, and A is the number of non-native species. The I' score can range from 0 to 100. An I' of 0 assumes a site is dominated by habitat generalist species and a score of 100 assumes a site is dominated by habitat specialist species.

We recollected vegetation data from the study sites in June and July of 2006. As prescribed by a species–area curve analysis (Miller and Wardrop 2004), a 20 m by 20 m sampling grid of 9 sampling plots was assessed. These plots were centered along the three study transects. Only 6 plots were assessed at Site # 140 and 8 plots at Site # 188, due to the intersection of a large debris pile and a gravel road, respectively. Site # 124 was excluded from this analysis because high grazing activity prevented the identification of ground cover species. At each plot we sampled vegetation following a modified version of the RAP designed by Riparia (Miller and Wardrop 2004).

Percent ground cover was measured using a point sampling method (originally developed by Levy 1927) in 1-m^2 blocks, centered on the plots. A 10-point frame (Heady and Rader 1958) was used in two directions (parallel and perpendicular to the stream) to collect a total of 20 measurements within each sampling block. Percent ground cover was calculated for five cover categories at each site, based on the 180 points assessed. We broke native species into three categories across a gradient of coefficients of conservatism (COC) (Chamberlain and Ingram 2012), and non-native species into introduced and invasive species categories. Bare ground was included as a sixth category. The basal area of the upper story canopy was also estimated by obtaining diameter at breast height measurements of each tree within 11.6-m radius circles around plot centers. This data was collected in 2006, with the exception of Site # 151, where previous assessment (2000) data was used.

Physiochemical Environment

Microtopography

At each study site we collected relative elevation measurements in November 2009 using conventional survey equipment (David White Instruments, Model LT8-300). Measurements were taking along the three study transects. We recorded measurements at 0.5 m intervals on the horizontal center of 15 1-m^2 sampling blocks (i.e., three measurements per sampling block), which were used for soil sampling. We calculated limiting elevation differences (LD) and limiting slopes (LS) (Linden and Van Doren 1986) for each study site. LD has been described as a measure of relief because it reflects measures of elevation differences at the largest distances considered. In contrast, LS has been described as a measure of roughness because it reflects a measure of the surface slope at the smallest distances considered (Moser et al. 2007).

Temperature

We retrieved climate information for State College, Pennsylvania (40.793 °N 77.867 °W, 356.62 m above sea level) to evaluate general weather patterns in the region over the course of the sample collection period. Data has been collected here since 1893 by the U.S. Historical Climate Network (HCN, http://climate.met.psu.edu/gmaps/PASC_DEVELOPMENT/#). All study sites are located within a 69-km radius of this location. Temperature (i.e., daily minimum and daily maximum temperatures) and precipitation data (i.e., rainfall and snowfall) were summarized by year between 2000 and 2010.

We installed seven temperature loggers at each study site from mid-December 2009 through mid-August 2010. Two ibutton temperature loggers (DS1921H/DS1921Z, High-Resolution Thermochron® iButton®, Range *H*: +15 to +46°C, *Z*: −5 to +26°C) were placed randomly along each of the three study transects. One HOBO® temperature logger was placed at the center of the site (HOBO® Temperature Data Loggers). Loggers recorded temperature measurements every 3 h. We were unable to relocate and/or download all loggers during collection, but a minimum of two loggers were found and used to calculate the mean site temperature for each 3-h interval.

Soils

We collected soils from 15 1-m^2 sampling blocks (9 sampling blocks at Site # 140) along each study transect on four sampling dates: August 2006, November 2006, May 2007, and August 2007. All samples were collected from a site on the same day, except in August 2007 when each sites was sampled on two consecutive days. To minimize sampling block destruction, a total of four randomly sampled soil cores were pooled from each 1-m^2 block in August 2006, November 2006, and May 2007. However, during the final August 2007 sampling period, a more complete but more destructive approach was taken. A total of 16 randomly sampled soil cores were taken and were pooled from each 1-m^2 block. Before cores were taken we measured volumetric water content (Θv) at each core point, using an EC-5 soil moisture probe (0–100% range, 0.003 m^3 m^{-3} resolution and 0.001 m^3 m^{-3} VWC accuracy in mineral soils, Decagon Devices, Inc. Pullman, WA, USA). An auger was then used to collect the 2.27 cm^2 soil cores. Before soil cores were pooled and/or stored, we first separated them into the complete organic horizon (if present) and the top 5 cm of the mineral horizon. We treated these two samples separately during laboratory analyzes.

All soil samples were placed in a cooler on ice while being transported back to The Pennsylvania State University, University Park, PA. Upon arrival at the laboratory, we weighed soils and stored them in a −10.7°C freezer until further processing. Soils were subsequently freeze dried in a pilot lyophilizer (Virtis Company, Gardiner, NY). Soil masses before and after freeze-drying were used to determine gravimetric water content ($\Theta g \pm 0.01$). This analysis was performed for all samples on all dates except on August of 2006. Dried soils were subsequently ground and passed through a 2 mm sieve to remove non-soil fractions (e.g., rocks, organic matter debris).

For all study sites and dates, we subsampled soil for pHw (pH in water) measurements on an Orion Ross Sure-Flow combination semi-micro pH electrode (±0.03) (Thermo Fisher Scientific, Inc. Product # 8175BNWP) in conjunction with an automatic temperature compensation probe (VWR® sympHony® Product # 11388-378). A 2:1 ratio of deionized water to soil was used (scaled from Thomas 1996) for most samples. However, a few organic horizon samples required a 3:1 ratio for the soil to be immersed in solution. This change in ratio could have had an effect on the pH readings (+0.4) (Davis 1943, as shown in Thomas 1996). If affected, the readings were altered in the conservative direction, making significant differences between disturbance groups and soil strata harder to find.

For August 2007, additional subsamples were used for nitrate–nitrite ($N\text{-}NO_3^- + NO_2$) and ammonium ($N\text{-}NH_4^+$) levels, and for SOM content. $N\text{-}NO_3^- + NO_2$ and $N\text{-}NH_4^+$ levels were analyzed using 2 M KCl extractions in a 1:10 ratio of soil to extractant (Keeney and Bremner 1966) with colorimetric measurements on a Lachat QuikChem® 8500 Series 2 Flow Injection Analysis System ($N\text{-}NO_3^- + NO_2^-$ method 12-107-04-1-F by Prokopy 2003 and $N\text{-}NH_4^+$ method 12-107-06-3-B by Diamond 2003). Corrections were made for low levels of $N\text{-}NO_3^- + NO_2^-$ (range: 0.00–0.04 mg N L^{-1}) and $N\text{-}NH_4^+$ (0.00–0.26 mg N L^{-1}) found on filters. We measured SOM content using loss on ignition (LOI) following Nelson and Sommers (1996) with modification by Ben-Dor and Bannin (1989). In addition to the use of blanks and controls, 10% of samples were analyzed in replicate for pHw (SD ± 0.03), $N\text{-}NO_3^- + NO_2^-$ (SD ± 0.14), $N\text{-}NH_4^+$ (SD ± 0.60), and SOM (SD ± 0.57) to calculate laboratory precision. We calculated summary statistics (i.e., means, SDs, CoVs) for all parameters at each study site on all dates measured.

Statistical Analyses

Comparisons were made between disturbance score groups for landscape and headwater complex parameters using two-way analysis of variance (ANOVA), one-way ANOVAs, and *t*-tests. When required, data were log transformed to conform to assumptions of normality and homogeneity of variance for ANOVAs and Tukey post hoc tests. Unequal variance tests (Welch's *T*-Test) or nonparametric tests (i.e., Mann–Whitney *U* Test) were used in place of *t*-tests (Two sample *T*-Test) when assumptions were not satisfied. These statistical analyses were conducted in the base package of R version 2.10.1. (R Development Core Team (2008)).

3.2.3 Results and Discussion

3.2.3.1 Landscape Structure

Topography

Topographic features, such as elevation, relief, and slope can influence the spatial configuration of land use across a landscape. Thus, they not only define the potential for a wetland to exist in a given location (Mitsch and Gosselink 2000), but in part, can determine the level of anthropogenic disturbance a headwater complex receives. Average elevation was not significantly different between disturbance score groups (Table 3.2). Although there was overlap in elevations between these groups, this was only true for the two most moderately stressed sites (i.e., Site # 140 and Site # 83); with three of the reference standard complexes situated at slightly higher elevations than three of the stressed complexes. Topographic relief and average slope within the 1- and 2-km radius circles were significantly different between disturbance score groups. Reference standard complexes had higher relief and mean slopes within these areal extents. Within the 100-m radius circle, relief was not significantly different, but mean slope followed the significant pattern found in the 1- and 2-km radius circles. Differences in relief and mean slope were not statistically significant within the 5-km radius circle areal extent.

The lower elevation, relief, and mean slopes of stressed complexes could reflect why these sites were more prone to exist in surrounding anthropogenic land cover classes, compared to reference standard complexes, situated within steeper, sloped, forested valleys. Areas of lower elevation and more gentle terrain are more accessible and economically viable for land uses, such as agriculture, which accounts for 27% of land use in Pennsylvania (USDA 2009). These findings are consistent with other regions. For example, in the southern Appalachian region lower elevation and gentler terrain were used to explain why areas were more likely to remain in non-forest land cover (Turner et al. 1996; Wear and Bolstad 1998) after extensive forest clearing at the turn of the twentieth century (Williams 1989). In regions of Western Europe abandonment of agricultural practices has been found to exist in remote, steeply slope areas (MacDonald et al. 2000). However, under favorable economic conditions and/or with temporal depletion of land resources encroachment in these more remote, rugged regions is possible (Turner et al. 1996, 2003).

Historical Land Cover

Legacy effects have become a prominent issue in ecosystem ecology (Christensen 1989; Richter 2007), as we are well into the Anthropocene (Crutzen 2002), an era where the landscape is increasingly influenced by man. Past land use can restructure the physical, chemical, and biological components of a system explaining variance that can last for centuries after the particular land use has ceased. Given that many of our forested wetlands in the eastern USA have been exposed to changes in onsite and

Table 3.2 Measures of site elevation, surrounding landscape relief, and surrounding landscape mean slope for the eight headwater complex study sites. The surrounding landscape was assessed in 100-m, 1-, 2-, and 5-km radius circles centered on each study site. Statistics are displayed for comparisons between headwater complex disturbance score groups

			Relief (m)				Mean slope			
D-score	Site[a]	Elevation (m)	100 m	1 km	2 km	5 km	100 m	1 km	2 km	5 km
Reference standard	83	231	25	346	481	523	14.89	29.45	32.17	24.62
	188	395	45	277	434	506	16.85	24.62	25.35	18.17
	60	310	20	210	319	492	12.23	16.12	22.10	19.26
	13	376	10	169	228	367	5.68	20.05	22.13	21.12
Stressed	140	329	13	95	144	301	6.47	11.28	11.86	14.42
	158	273	10	113	136	510	4.48	13.70	14.18	19.85
	151	213	12	75	219	470	4.99	10.27	12.48	14.87
	124	239	6	111	196	521	3.03	16.40	15.97	19.30
Test		*T*-Test	Mann–Whitney *U*	Welch	*T*-Test	Mann–Whitney *U*	*T*-Test	*T*-Test	*T*-Test	*T*-Test
Statistic		−1.441	2.4	−3.816	−3.168	7	−3.114	−3.027	−4.645	−1.831
df		6	–	3.308	6	–	6	6	6	6
p-value		0.1997	0.1465	0.0266	0.0194	0.8857	0.0207	0.0232	0.0035	0.1168

[a]Site numbers are identification numbers from the Riparia reference collection: http://wetlands.psu.edu/projects/reference_wetlands.asp

surrounding land uses since precolonial times one would expect that a headwater complex's condition could be in part a product of these legacies. For a brief history (beginning with European settlement) of forested wetlands in the USA., see Stine 2008.

A summary of historical land cover information at and around the headwater complexes used in this study can be found in Table 3.3. An investigation of atlases and historical aerial photographs exposed a stark contrast in legacies between disturbance score groups, but also a range of activities among the stressed complexes; detailed descriptions of these activities, as well as the atlases and photographs utilized for the analysis, are presented as supplementary material at http://www.riparia.psu.edu/MARbook. Atlases revealed that most of the reference standard complexes had minimal upstream activity in the late 1800s, while most of the stressed complexes had a range of upstream activities. The surveys showed that Site # 124 had a saw mill on the edge of the study site and iron ore mines on one of its first order upstream channels, while Site # 158 and Site # 151 had rural development upstream. Only Site # 140 had no upstream activities.

Later historical aerial photographs also revealed a difference between disturbance score groups. While reference standard complexes were located within forested landscapes between the late 1930s and today, stressed complexes were located in a matrix of non-forested surrounding land cover classes that varied in duration and timing. Stressed complexes were situated in landscapes that were predominantly agricultural (e.g., pasture, croplands). However, while some of these study sites and their surrounding land cover classes did not change much over this period, other complexes were more dynamic. For example, both Site # 140 and Site # 124 were situated in fairly stable landscapes, with little change to the study sites between the late 1930s and today. However, these headwater complexes were very different in terms of their land cover classes. While Site # 140 was located within a forested buffer strip during this period, Site # 124 was located on a pasture. In contrast, Site # 158 and Site # 151 have experienced a number of changes in land cover between the late 1930s and today.

Current Land Cover

While legacies might define the template on which a wetland becomes established or reestablished, the current surrounding land cover can also impact the wetland's condition and associated functions. Although in many cases wetlands are defined by jurisdictional boundaries, as mentioned earlier they are open systems in regard to their material and nutrient fluxes and act in concert with other components of the landscape at multiple scales. In other words, jurisdictional boundaries might not constitute the ecological boundaries needed to protect the condition of a wetland or its specific functions. Given that fluxes in these wetlands are highly dependent on the surrounding landscape, there is a high probability that the condition of these systems might be indirectly affected by changes to their natural disturbance regimes.

The original satellite images used to perform current land cover analyses can be found in Fig. 3.4. The percent forest cover in the 1-km radius circle surrounding the center of each headwater complex was used in the calculation of the disturbance

Table 3.3 Summary of land cover for the eight headwater complexes study sites across time. This summary was created from visual inspection of late 1800 atlas surveys (Beers 1868; Nichols 1873), aerial photographs from the late 1930s to the early 1970s (PA Geological Survey, http://www.pennpilot.psu.edu/), and orthoimages for 2005 (PA DCNR, http://www.pasda.psu.edu)

		Upstream	Onsite[b]			
D-score	Site[a]	1868–1873	1937–1942	1957–1962	1967–1972	Current
Reference standard	83	No activity	Forest	Forest	Forest	Forest Gravel road
	188	Coal and lumber sawmill Tree residences Road crossings	Forest	Forest	Forest	Forest Old canal path
	60	No activity	Forest	Forest	Forest	Forest
	13	No activity	Forest	Forest	Forest	Forest
Stressed	140	Road crossing	RFB	RFB	RFB	RFB
	158	18 residences Road Crossings	Residence	Residence	Residence Park dedicated in 1979	RFB
	151	Highly developed Residential area Road onsite	Stream channel Agricultural field	Stream channel Shrubs	Filled channel RFB	Filled channel RFB
	124	Saw mill Iron ore mines Eight residences Few road crossings	Pasture	Pasture	Pasture	Pasture

[a]Site numbers are identification numbers from the Riparia reference collection: http://wetlands.psu.edu/projects/reference_wetlands.asp
[b]This includes activity in the immediate surrounding area
RFB is used as an abbreviation for "riparian forested buffer"

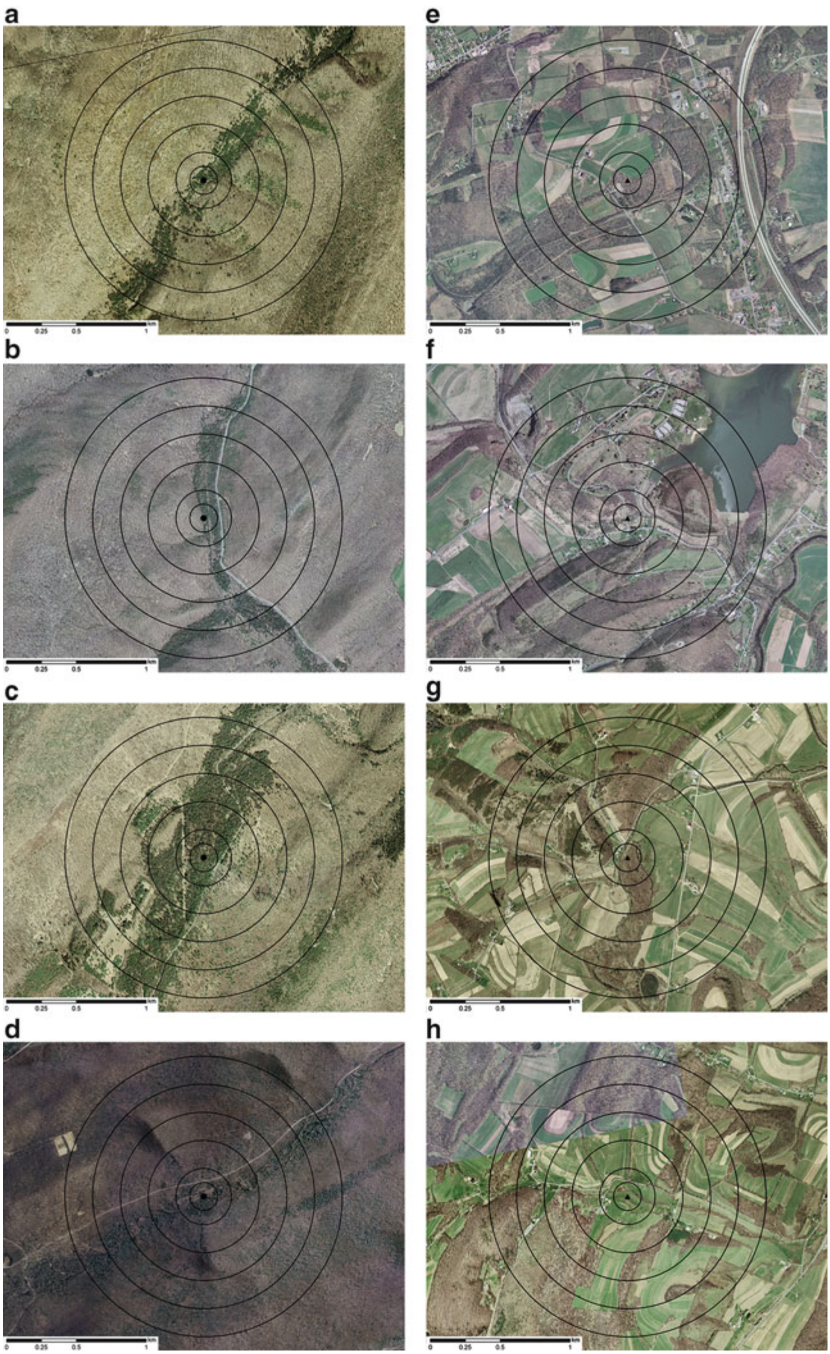

Fig. 3.4 Current land cover, as depicted in 2005 orthoimages (scale 1:10,699) obtained from the PAMAP Program (PA DCNR, http://www.pasda.psu.edu). Orthoimage include: (**a**) Site # 83, (**b**) Site # 188, (**c**) Site # 60, (**d**) Site # 13, (**e**) Site # 140, (**f**) Site # 158, (**g**) Site # 151, and (**h**) Site # 124. The areal extents used to evaluate landscape metrics are shown

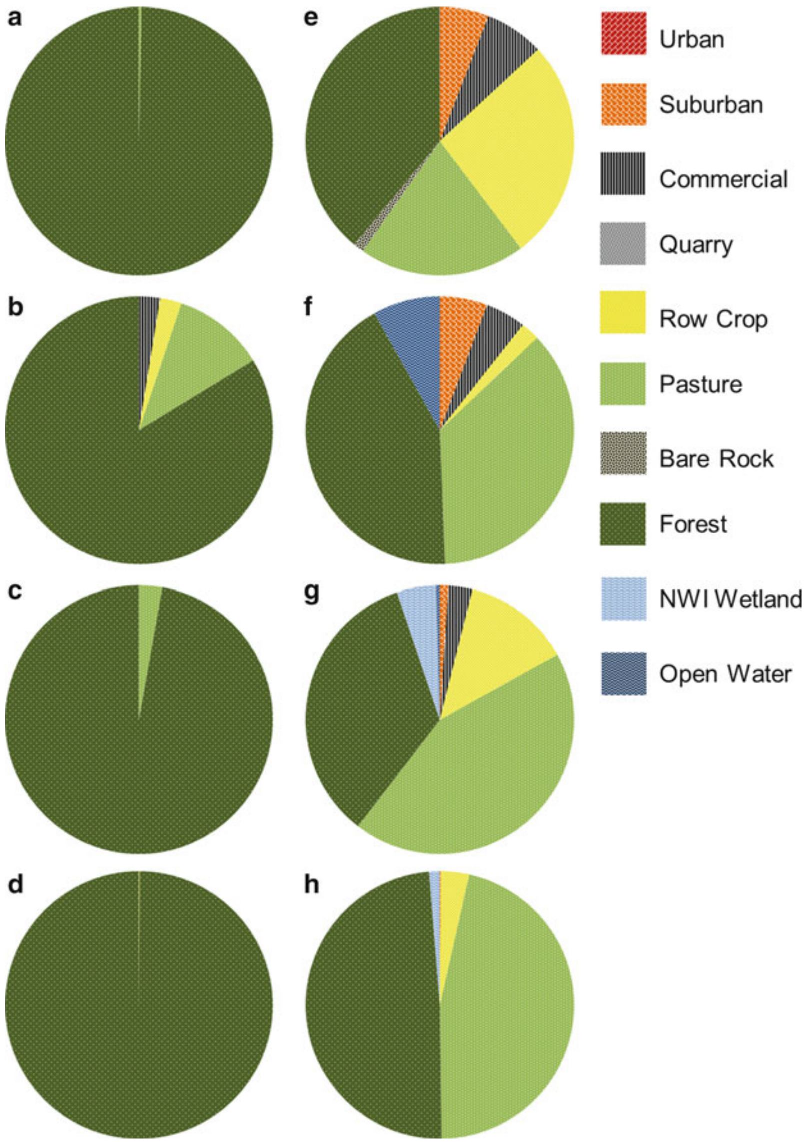

Fig. 3.5 Percent cover for land cover classes in 1-km radius circles centered on headwater complex study sites. Pie charts include: (**a**) Site # 83, (**b**) Site # 188, (**c**) Site # 60, (**d**) Site # 13, (**e**) Site # 140, (**f**) Site # 158, (**g**) Site # 151, and (**h**) Site # 124

score, which was used as a primary basis of the sampling design. Thus, as expected this parameter was significantly different between disturbance score groups. Reference standard complexes had >83% forest cover while stressed complexes were estimated to have >49% of land cover classes in pasture, row crop, suburban development, and/or commercial roadways (Fig. 3.5). Percent core forest was also significantly lower in stressed complexes (i.e., Mann–Whitney $U = 0$, p-value = 0.0286), while LDI was significantly higher in stressed complexes (i.e., T-statistic = 5.001, df = 6, p-value = 0.0024). Percent impervious surface was not significantly different between disturbance groups at this spatial extent (i.e., Mann–Whitney $U = 15$, p-value = 0.0571).

Percent forest in a 1-km radius, in combination with other 1-km radius metrics (LDI, % impervious surface, and mean patch size), have been significantly correlated with a site level benthic index of biological integrity (IBI) (Brooks et al. 2009) developed by the Maryland Biological Stream Survey (MBSS) (Boward et al. 1999). However, the 1-km radius index was only weakly correlated with site nitrate (NO_3^-) levels (Brooks et al. 2009). It was suggested that NO_3^- concentrations might only be dependent on upstream inputs, and therefore, the upstream portion of the circle might be a better indicator of NO_3^- levels. After visual inspection of the 1-km radius circles in this study, only Site # 140 showed a clear difference in upstream and downstream land cover classes. Unlike its downstream counterpart, the upstream had no suburban development or commercial roadways. It was also the only stressed complex to have a continuous forested buffer strip surrounding the upstream channel.

It is also possible that other spatial extents could be more influential in site level condition. For example, Wardrop et al. (2007b) showed that higher condition sites could occur in developed landscapes if they had an intact forested buffer immediately surrounding the site. Thus, we also looked at comparisons of land cover between disturbance groups by breaking the 1-km radius circle down into a 100-m radius circle and successive buffer strips. In the 100-m radius circles all reference standard complexes were estimated to have 100% forest cover, while stressed complexes had a gradient between 0 and 84% forest cover (Fig. 3.6). The predominant secondary land cover class at these study sites was pasture. Site # 158 also had notable levels of suburban development (8%) and commercial roadways (14%).

Using a visual inspection at this scale, classes were distributed equally across the upstream–downstream border for all headwater complexes. In addition, similar to the 1-km areal extent two-way ANOVAs revealed significant differences in percent core forest and LDI when comparing disturbance score groups across the 100-m radius circles and all buffer strips (Table 3.4). However, there was also a significant difference in impervious surface across all extents, which was absent from the 1-km areal extent. Stressed complexes had greater percent impervious surface across all extents compared to reference standard complexes.

3.2.3.2 Headwater Complex Structure

Hydrology

Hydrology is the primary driver in wetlands (Mitsch and Gosselink 2000), and as such can directly and indirectly affect ecosystem processes. Changes made to the landscapes surrounding wetlands have been associated with changes to wetlands hydrologic regimes (see Chap. 4), with concurrent acceleration of sedimentation and

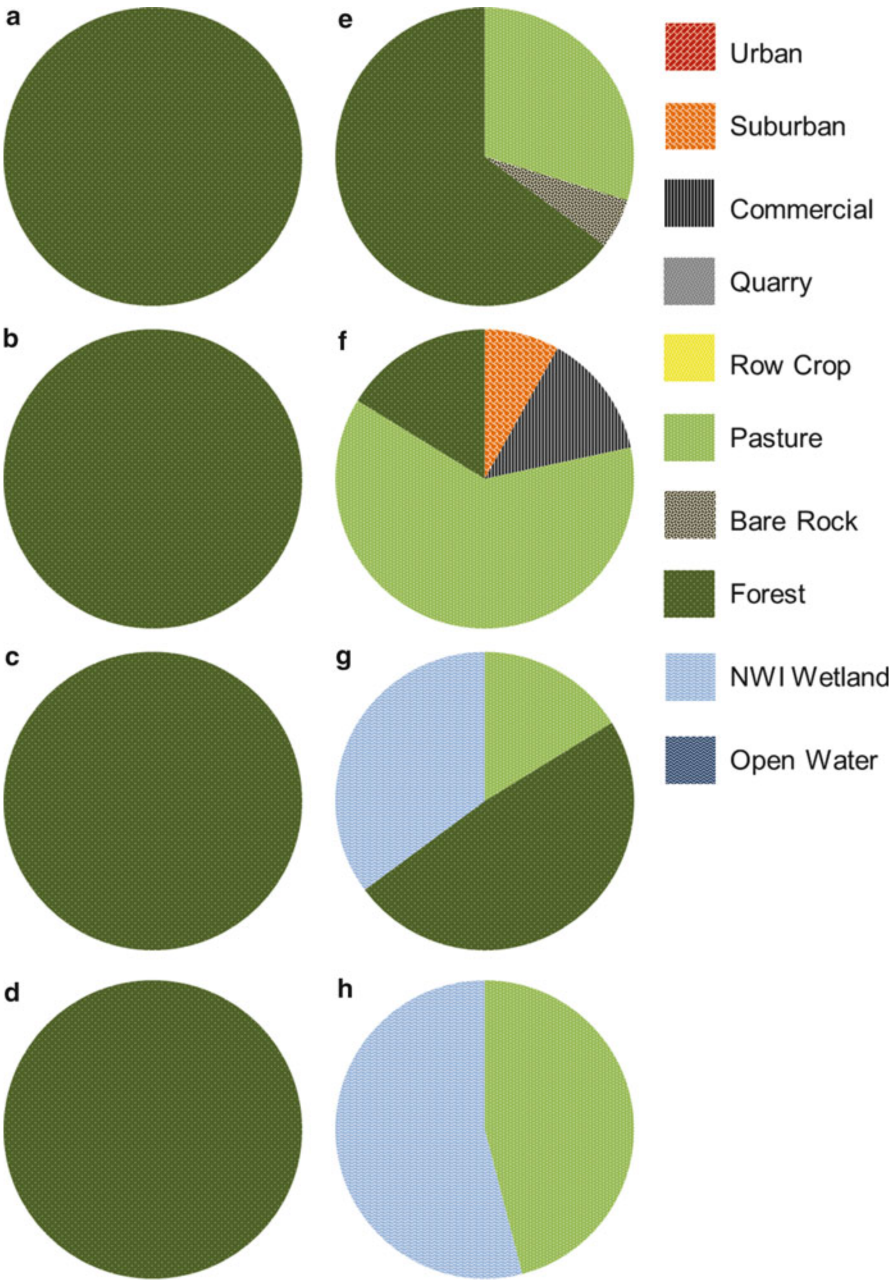

Fig. 3.6 Percent cover for land cover classes in 100-m radius circles centered on headwater complex study sites. Pie charts include: (**a**) Site # 83, (**b**) Site # 188, (**c**) Site # 60, (**d**) Site # 13, (**e**) Site # 140, (**f**) Site # 158, (**g**) Site # 151, and (**h**) Site # 124

Table 3.4 Two-way ANOVA results comparing landscape metrics between disturbance score groups and areal extents. Disturbance score groups included reference standard complexes and stressed complexes. Areal extents included 100-m radius circles centered on study sites, and buffer strips from 100 to 200 m, 200 to 400 m, 400 to 600 m, 600 to 800 m, 800 to 1,000 m

Factor	Parameter	df	*F*-statistic	*p*-value
Core forest (%)	Disturbance group	1	103.1	<0.0001
	Distance class	5	7.913	<0.0001[a]
	Interaction	5	0.657	0.6581
Land development index	Disturbance	1	142.45	<0.0001
	Distance class	5	0.3663	0.8682
	Interaction	5	0.0426	0.9989
Impervious surface (%)	Disturbance	1	9.365	0.0042
	Distance class	5	0.411	0.8377
	Interaction	5	0.886	0.5004

[a]One-way ANOVA and post hoc Tukey test did not show a significant difference among distance classes (df = 5, *F*-statistic = 2.334, *p*-value = 0.0587)

nutrient enrichment. During the past 25–50 years, elevated sediment deposition rates have been observed in forested riparian wetlands with significant anthropogenic disturbances (Johnston et al. 1984; Martin and Hartman 1986; Hupp et al. 1993; Hupp and Bazemore 1993; Kleiss 1996). Several studies have documented rates of sedimentation ranging from 0.07 to 5 cm per year in forested riparian wetlands affected by land use disturbance (Cooper et al. 1987; Hupp et al. 1993; Hupp and Bazemore 1993; Kleiss 1996). In central Pennsylvania, a study by Wardrop and Brooks (1998) showed that sediment deposition ranged from 0 to 8 cm per year across four freshwater wetland HGM classes[4] with varying levels of land use disturbance.

Werner and Zedler (2002) and Koning (2004) both reported that sedimentation is associated with an increase in soil bulk density. In a palustrine emergent marsh, Koning (2004) showed that bulk density increased by nearly threefold in field plots with 1 cm of sediment (0.33 g cm^{-1}, s.d. 0.11) compared to control plots with no sediment (0.12 g cm^{-1}, s.d. 0.02). Large-scale increases in bulk density can cause a decrease in the soil water storage capacity (Boelter 1964) and can also lead to a rise in soil temperatures. Additional work by Vargo et al. (1998) and Lockaby et al. (2005) showed that sedimentation rates as low as 0.04 cm per year and 0.20 cm per year were able to significantly decrease decomposition of emergent vegetative litter and foliar litter, respectively. Lockaby et al. (2005) also reported concurrent declines in nitrogen mineralization and microbial carbon and nitrogen biomass. Further, sediments can carry large quantities of inorganic and organic nutrients into wetlands (Johnston et al. 1984; Hupp et al. 1993). For example, Wardrop and Brooks (1998) found mean mineral and organic accretion rates of 778 g m^{-2} per year (±1417) and 550 g m^{-2} per year, respectively, in their wetland study sites. It is thought that accelerated sedimentation can overload the assimilative capacity of these wetlands (Jurik et al. 1994; Wardrop and Brooks 1998; Freeland et al. 1999).

[4]The four HGM classifications included headwater and mainstem floodplains, riparian depressions, slopes, and impoundments.

In addition, alterations to the stream channels (e.g., down cutting, stream incision) have been shown to create hydrologic disconnections between the streams and their adjacent wetlands (Bunn and Arthington 2002; Groffman et al. 2002), leading to a loss in retention functions (Noe and Hupp 2005) and overall imbalances between erosion and deposition patterns across floodplain networks (Hupp et al. 2009). Disconnected wetlands also tend to have a higher frequency of inundation or dehydration (Adamus et al. 2001; Ryan 2005). These shifts in hydrology can alter the availability of soil nutrients, carbon, and oxygen that affect the exchanges between aerobic and anaerobic processes, such as nitrification, denitrification, methane oxidation, and methanogenesis (Moore and Roulet 1993; Davidson et al. 1998; Bellisario et al. 1999).

Alterations to water level variability are also possible and have been suggested to be as important as water level in structuring wetland plant and microbial communities (Yu and Ehrenfeld 2010). Both Unger et al. (2009) and Langer and Rinklebe (2009) showed a decline in microbial biomass with stagnant inundation vs. intermittent short-term flooding. On the other hand, Fierer et al. (2003) showed that drying/rewetting cycles impacted microbial community composition in oak forest soils, but not on grassland soils. In this case, land cover was a bigger driver in microbial composition. Thus, maybe a more sensitive measure of the effects of drying and rewetting cycles on microbes is not on the microbial community composition itself, but on the microbial processes occurring within these systems. Numerous studies have shown an increase in short-term carbon and nitrogen mineralization rates with drying and rewetting cycles (Birch 1958; Sorensen 1974; Cui and Caldwell 1997), which is thought to be due to lysing of microbial cell walls or osmoregulation (Lund and Goksoyr 1980; VanVeen et al. 1985). However, on longer time scales hydrologic variability has also been shown to reduce microbial carbon mineralization (Fierer and Schimel 2002), decrease decomposition rates, and decrease overall microbial functional diversity (Schimel et al. 1999).

In the Ridge and Valley ecoregion of central Pennsylvania headwater hydrology is driven by a combination of ground water and surface water from overland (e.g., rainfall, snowmelt) and overbank flow (e.g., flooding events) (Cole et al. 1997). Overall, most of the median water level depths for the headwater complexes across dates were within the range measured earlier by Cole and Brooks (2000) across six headwater complexes of varying levels of surrounding landscape impact (−12 to −55 cm). Site # 151 was outside of this range (median = 5 cm) because of inundation for over half of the sampling period in 2006–2007, but was within the range for all other periods. Site # 13 and Site # 60 were also outside of this range for the 2010–2011 sampling period at two wells (median = 4 and −7 cm) and one well (−1 cm), respectively. However, this was consistent with work by Ryan (2005), who found median water level depths between −51 and 19 cm across 18 headwater complexes, which included those in Cole and Brooks (2000) previous study. All hydrographs followed a slight positive trajectory from August to January, which was most likely due to cessation of evapotranspiration (Cole and Brooks 2000).

Statistical comparisons were not made between disturbance score groups for hydrologic data, due to low sample size and incomplete overlap in the datasets

Table 3.5 Hydrologic parameters estimated for five of the headwater complex study sites for 6 month (August–January, between 2005 and 2008) and 12 month (2009–2010) periods

					Percent of period[b]			Percent of period where fluctuations in daily time steps were		
D-score	Site[a]	Date	Well	Days	Dry	Saturated	Inundated	10 cm	20 cm	30 cm
Reference standard	83	2005–2006	1	184	55	45	0	2.19	0.55	0.00
		2006–2007	1	184	43	57	0	2.73	0.00	0.00
		2007–2008	1	182	58	42	0	3.84	1.10	0.00
	60	2005–2006	1	184	8	92	0	0.55	0.00	0.00
		2006–2007	1	102	0	100	0	0.98	0.00	0.00
		2007–2008	1	157	6	94	0	1.86	0.62	0.62
Stressed	158	2006–2007	1	163	64	36	0	6.01	3.28	0.55
		2007–2008	1	182	76	24	0	7.69	1.65	0.55
	151	2006–2007	1	159	1	38	61	7.73	1.66	0.55
Reference standard	60	2009–2010	2	253	31	53	15	8.73	0.40	0.40
			3	253	0	53	47	1.19	0.00	0.00
			4	233	0	73	27	2.16	1.29	0.09
	13	2009–2010	1	366	0	26	74	1.37	0.27	0.00
Stressed	158	2009–2010	2	366	70	29	1	5.31	1.64	0.00
			3	366	75	25	0	5.21	1.64	1.09
			4	296	62	37	1	3.39	1.02	0.68
	151	2009–2010	2	359	32	37	31	6.56	2.95	1.64
			3	366	68	27	4	13.13	3.37	1.01
			4	333	50	4	32	11.45	1.01	3.37

[a]Site numbers are identification numbers from the Riparia reference collection: http://wetlands.psu.edu/projects/reference_wetlands.asp

[b]Hydrologic moisture regimes (i.e., inundation = >0 cm, saturation = 0 to −30 cm, dry = <-30 cm) are described by Cole et al. (2006)

collected. However, we do mention some general trends in the data. Water level data measured from the same well were fairly consistent across sampling cycles (Table 3.5). Contrarily, multiple wells at the same site, during the same sampling period, showed much higher variability. We suspect that the spatial heterogeneity between or among wells can be explained predominately by microtopographic features, discussed later. Briefly, Site # 151 and Site # 13 water level heterogeneity could be explained by microtopographic relief (i.e., LD), while Site # 60 water level heterogeneity could be explained by microtopographic roughness (i.e., LS). The only study site that was stable across both time and space was Site # 158, which contained relatively low microtopographic relief and roughness.

Taken together stressed complexes tended to be drier during sampling periods than reference standard complexes. Ryan (2005) showed similar patterns, with 80% of his reference standard complexes classified as saturated across an annual cycle, and 65% of his highly stressed complexes classified as dry or inundated across an annual cycle. The drying or wetting of wetlands in "developed" settings can occur for number of reasons. Wetter conditions can exist where areas of impervious surface cover (e.g., roads) or compaction of soil due to land management practices increase surface water runoff (Arnold and Gibbons 1996). This can also lead to overbank flooding as stream channels reach capacity at an accelerated rate. Conversely, over time high velocity stream events can incise stream channels, scouring out sediment, and drawing down the groundwater water table (Groffman et al. 2003). This process makes the riparian zone drier and as a result more similar to uplands in their soil and vegetative properties.

We found more consistent differences between disturbance score groups by evaluating measures of the intensity of water table fluctuations across daily time steps. Ehrenfeld and Schneider (1993) reported that water level ranges in white cedar swamps were much higher in suburban vs. undeveloped watersheds. Similarly, reference standard complexes tended to have lower occurrence rates of high fluctuation events, compared to stressed complexes. In other words, reference standard complexes had more stable hydrographs with fewer and less intense water table fluctuations over the course of the measurement periods (e.g., Fig. 3.7a, b) compared to stressed complexes (e.g., Fig. 3.7c, d). However, reference standard complexes still exhibited more spatial variability between well points.

Vegetation

Both variation in belowground plant species rhizospheres, via labile nutrient exudates and delivery of oxygen (Halbritter and Mogyorossy 2002), as well as the supply of nutrients through variation in tissue quality delivered during decomposition (Hooper an. Vitousek 1997; Bardgett and Shine 1999; Waldrop et al. 2000; Balser and Firestone 2005; Porazinska et al. 2003; Ushio et al. 2008) can affect wetland ecosystem processes. Land use legacies have directly altered the vegetative communities in these systems through time, and current surrounding land cover is suspected to affect vegetative communities through changes in the hydrologic regimes described above. For example, changes to the hydroperiod can affect seed dispersal and alter community composition through specific life history adaptations (Silvertown et al. 1999; Visser et al. 2000; van Eck et al. 2004; Leyer 2005; van der Hoek and Sýkora 2006). Sedimentation further affects vegetation by inhibiting germination in soil seed banks, by reducing seedling emergence (Mahaney et al. 2004), by suppressing plant productivity, and by decreasing community species richness and diversity (Jurik et al. 1994; Mahaney et al. 2004). By contrast, hydrologic and material flux alterations have been shown to increase the ability of non-native species to invade wetlands. This may be associated with opportunities for invasive species to invade by dispersal transport, cotransport of nutrients, and creation of canopy gaps (Zedler and Kercher 2004).

At the headwater complexes used in this study I' scores were significantly different between disturbance score groups, with reference standard complexes having I' scores between 42 and 53 and stressed complexes having scores between 23 and 31. This relationship was expected as Miller et al. (2006) found a strong negative linear

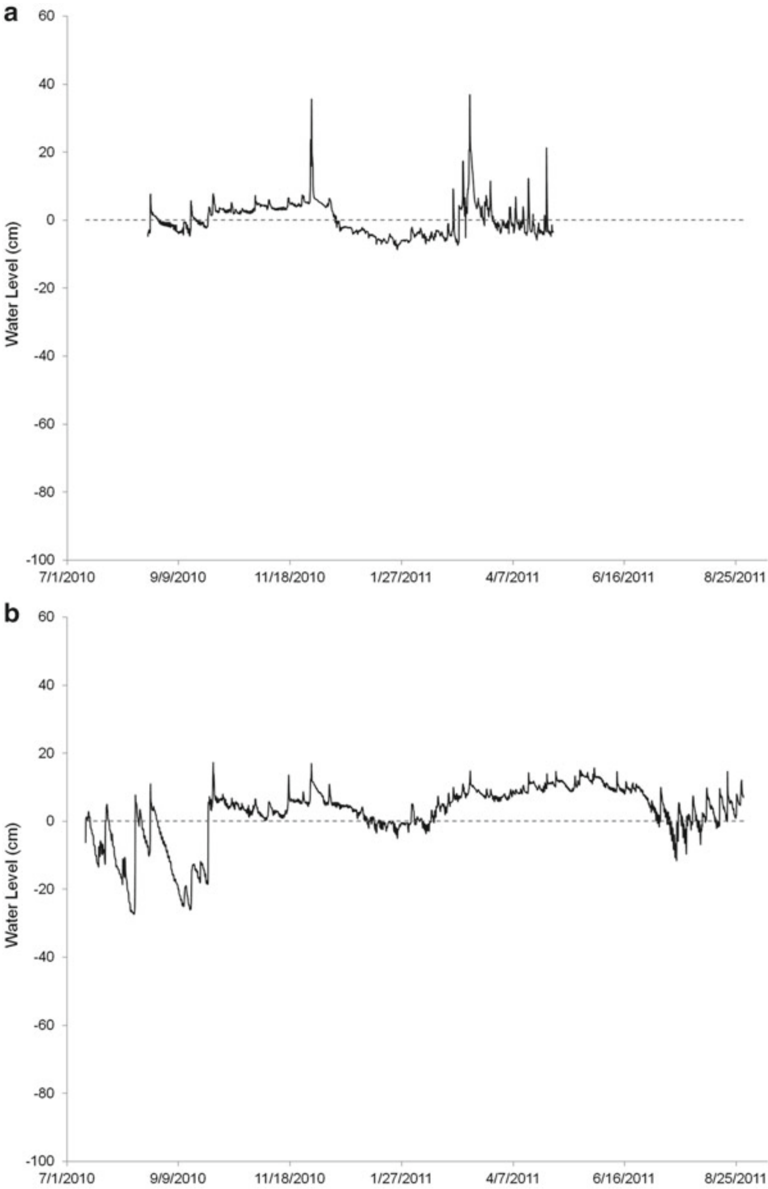

Fig. 3.7 Hydrographs for reference standard complexes (**a**) Site # 60 Well # 3 and (**b**) Site # 13 Well # 1, and for stressed complexes (**c**) Site # 158 Well # 2 and (**d**) Site # 151 Well # 3. Data from one well was selected for each site during the 2010–2011 monitoring period. *Dotted lines* indicate ground level

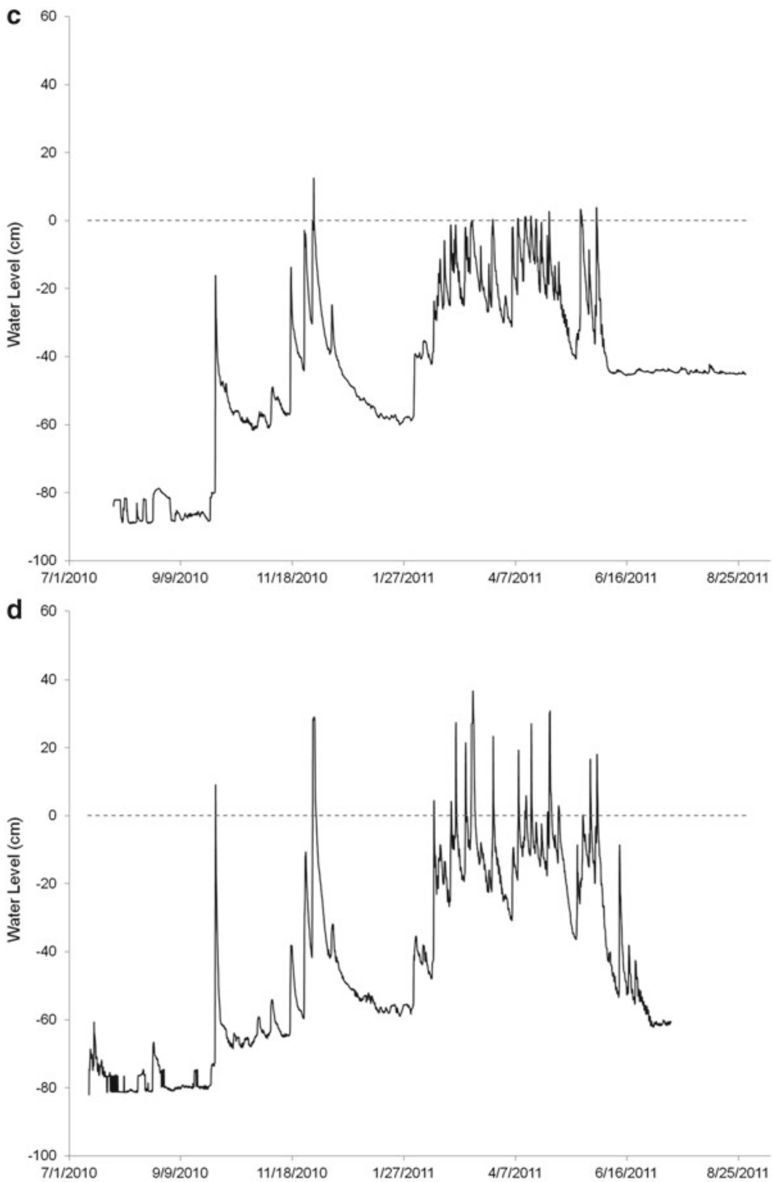

Fig. 3.7 (continued)

relationship between a site's adjusted I' score and the disturbance score. Miller et al. (2006) also found strong positive linear relationships between disturbance scores and percent invasive species cover and percent cover of *Phalaris arundinacea*. Fennessy et al. (2004) also documented higher abundances of generalist and invasive species in "disturbed" conditions. Our assessment confirmed similar differences between the

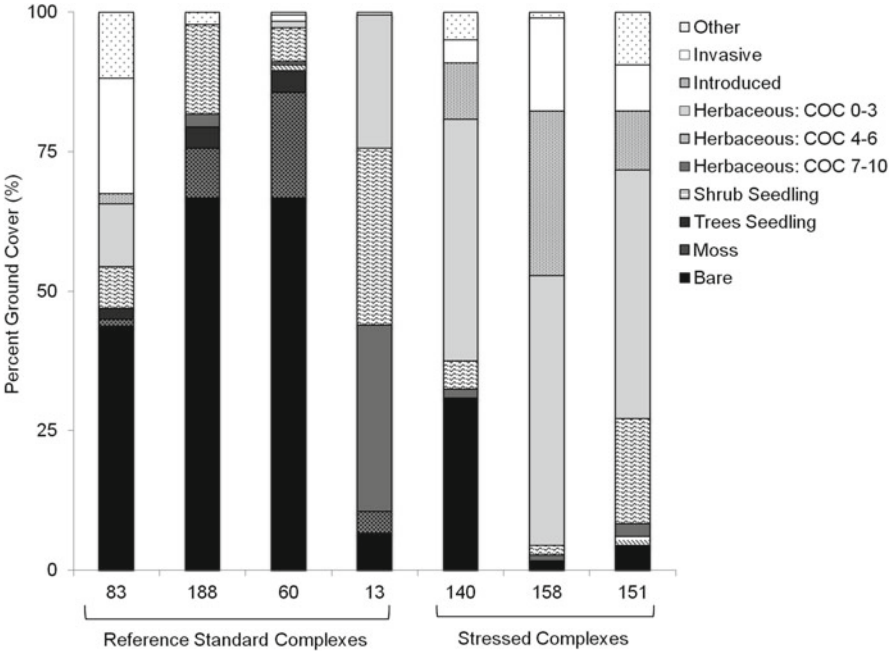

Fig. 3.8 Percent ground cover of herbaceous categories in seven of the eight headwater complex study sites. Percent cover was estimated from the sum of 1-m^2 plots. "Other" includes native herbaceous samples that could not be identified to species and therefore could not be assigned a COC value

disturbance score groups. Percent ground cover measurements varied between disturbance score groups with a dominance of non-native invasive species, native invasive species, and non-habitat specific (low COC values) ground cover groups in stressed complexes that were absent in reference standard complexes (Fig. 3.8).

There were very few species in moderate and high COC categories, except at Site # 13, where 65% of the ground cover was within these two categories. In addition, reference standard complexes, with a predominance of high perennial wetland patches, had higher percentages of bare ground (44–67%), while reference standard, riparian depression Site # 13 and stressed complexes had higher densities of herbaceous cover. Of these study sites, Site # 140 was the only headwater complex to have an appreciable amount of bare ground (31%). Site # 124, which was excluded from this portion of the analysis, also had high levels of ground cover (data not shown: average bare ground = 8%, n = 9 plots), but most ground cover was unidentifiable due to grazing. Finally, as expected only reference standard complexes, which were of older growth stages, were identified as having measurable moss (1–19%) and tree seedling (2–4%) cover.

There was significant range in tree basal area estimates across study sites. Reference standard and stressed riparian depression complexes had similar basal area estimates (i.e., Site # 13 = 9 $m^2\ ha^{-1}$, Site #124 = 8 $m^2\ ha^{-1}$). However the distribution of the trees within the sites differed; Site # 13 had trees scattered throughout the complex, while Site # 124 had a few mature trees situated within meters of the stream. On the other hand, basal area estimates for upper perennial complexes did differ across disturbance score groups. Reference standard complexes had basal areas ranging from 31 to 57 m^2 ha^{-1} and stressed complexes had basal areas ranging from 4 to 19 m^2 ha^{-1}. It should be noted that the basal area for Site # 151 (i.e., 4 m^2 ha^{-1}) was most likely underestimated. Data for this study site was taken from a previous assessment (2000), where only half the complex (i.e., the old streambed, not the original riparian zone) was assessed. This is further validated by the high percent forest cover within the 100-m radius circle areal extent.

Physiochemical Environment

Microtopography

Microtopography is an important component of wetland ecosystems, promoting vegetative structure and composition (Ettema and Wardle 2002) as well as creating both aerobic and anaerobic zones, (Reddy and Patrick 1984). The creation of small-scale gradients of soil moisture and associated oxygen concentrations is expected to lead to an accumulation of soil organic matter (SOM) in hollows where slower decomposition is expected (Bruland and Richardson 2005), with subsequent decreases in bulk density and pH levels (Ponnamperuma 1984). These gradients are also expected to enhance processes such as nitrification (e.g., on a hummock) and denitrification (e.g., in a hollow). Wolf et al. (2011) showed that microtopographic relief (i.e., LD) increased nitrification rates, while microtopographic roughness (i.e., LS) increased both nitrification and denitrification. Bruland and Richardson (2005) did not find similar differences when assessing denitrification rates. However, they suggest that sampling might have been too early (3 years since microtopographic establishment) to see differences in these rates.

This microtopography is created by a multitude of drivers, such as flooding, sedimentation, erosion, tree fall, root growth, litter fall, and animal manipulation (e.g., burrowing, tracks, excretions) (Bruland and Richardson 2005), many of which are expected to have been affected by both past and current land cover in and around study sites. However, unlike surrounding landscape relief, site level relief (i.e., LD) and roughness (i.e., LS) were not significantly different between disturbance score groups (Table 3.6). All headwater complexes had elevation ranges within 2 m, except for Site # 188, which crossed multiple active stream channels and ranged up to 2.75 m. It should be noted that both complexes with a predominance of riparian depression wetland patches had lower roughness parameter estimates compared to complexes with a predominance of upper perennial wetland patches. This is indicative of the geomorphology of riparian depressions.

Reference standard complexes with a predominance of upper perennial wetland patches tended to have higher roughness parameter estimates compared to stressed complexes with a predominance of upper perennial wetland patches; except for over-

Table 3.6 Measures of microtopographic limiting slope (LS) and limiting elevation (LD) for the eight headwater complex study sites. Statistics are displayed for comparisons between headwater complex disturbance score groups

D-score	Site[a]	Limiting slope (unitless)	Limiting elevation differences (m)
Reference standard	83	0.035	0.177
	188	0.050	0.383
	60	0.040	0.184
	13	0.018	0.347
Stressed	140	0.041	0.211
	158	0.022	0.124
	151	0.022	0.458
	124	0.016	0.434
Test		*T* test	*T* test
Statistic		0.3453	−1.2182
df		6	6
p-value		0.7417	0.2689

[a]Site numbers are identification numbers from the Riparia reference collection: http://wetlands.psu.edu/projects/reference_wetlands.asp

lap between the two most moderately stressed complexes (Site # 140 and Site # 83), which was similar to comparisons of elevation measurements. We suspect that this overlap can be explained by legacy patterns, where Site # 140 has been forested and maintained within a forested buffer since at least the late 1930s. This legacy is more similar to reference standard complexes, than other stressed complexes, which were not forested or within a forest buffers for the same length of time. Thus, we suspect that the lower roughness parameters found in Site # 158 and Site # 151, were related to both a lack of woody canopy and land management activities (i.e., lawn, fill, agricultural field), which have the potential to homogenize soils.

Temperature

It is well known that soil formation and ultimately microbial processes are influenced by climate factors, such as temperature (Jenny 1941; Brady and Weil 2002). Microbes have a range of optimal temperatures for processing (e.g., nitrification vs. ammonification, figure 7.18 in Brady and Weil (2002), mesophilic nitrifiers 25–35°C in Focht and Verstraete 1977). In addition, the rate of a particular process will also change with temperature. For example, MacDonald et al. (1995) showed variability in respiration, nitrogen mineralization, and sulfur mineralization in soil surfaces of hardwood forests for a range of temperatures. They showed an approximate doubling of these process rates with a 10°C increase in temperature. Rapid fluctuations in temperature can also affect cycling. For example, freeze-thawing periods can expedite cell lysing, alter SOM structure, and concentrations of exchangeable N-NH_4^+ and soluble phosphorus (Boone et al. 1999).

Climate data for State College, PA (40.793 °N 77.867 °W, 356.62 m above sea level) depicted a slight increase in the minimum (linear trend: $y=0.0006x-16.431$) and maximum (linear trend: $y=0.0006x-9.9957$) daily temperature readings over

Table 3.7 Annual mean daily maximum and minimum temperatures and annual precipitation and snowfall data for State College, PA at 40.793 °N 77.867 °W from 2000 through 2010. Mean values are also displayed for February 1, 1983 through December 31, 2010

	Mean daily temperature (°C)			
Year	Minimum	Maximum	Precipitation (cm)	Snow fall (cm)
2000	11.5	2.3	72.2	70.2
2001	15.8	5.2	77.2	65.9
2002	15.8	6.1	108.1	113.4
2003	14.2	4.9	139.2	205.3
2004	14.7	5.4	136.5	125.3
2005	15.3	5.5	92.1	138.0
Data collection period				
2006	16.1	6.2	94.5	22.5
2007	15.7	5.5	91.3	127.4
2008	15.1	5.1	104.5	97.7
2009	14.6	5.1	92.9	94.3
2010	15.7	5.5	87.1	73.5
Mean 1983–2010	15.0	4.5	97.5	114.7

the past decade when headwater complex sampling occurred. Precipitation data were highly variable in the first half of the past decade, with 2 years of low precipitation and 2 years of high precipitation (Table 3.7). Precipitation was consistent across the years of sample collection, within 10 cm of the average precipitation for the location. However, for all but 1 year both precipitation and snowfall were below the 117-year average. This was notably true for snowfall levels in 2006, which was the winter prior to initial soil sampling.

We expected there to be differences in microclimates between disturbance score groups. Vegetative structure can influence microclimate through processes such as direct shading, wind shielding, and evapotranspiration. Percent forest cover has been linked to both lower temperatures through buffering the effects of radiation and moderating diurnal fluctuations (e.g., McNulty et al. 2005). Given the clear alterations to the vegetative structure of stressed complexes by land use legacies and by lower percent forest cover currently surrounding these sites, temperatures were expected to be higher in these headwater complexes compared to reference standard complexes.

On average, daily temperatures ranged approximately 3.7°C across all headwater complexes. However, there were periods where daily temperature differences between the hottest and coolest complexes reached 17.5°C. There were no significant differences found between disturbance score groups for the average monthly temperatures in December (2009, data for half month), January (2010), or August (2010, data for half month) (Table 3.8). However, as expected stressed complexes had significantly higher average monthly temperatures from February (2010) to July (2010). For these months stressed complexes were estimated to be between 1.2 and 1.8°C warmer than reference standard complexes. Of the stressed complexes, Site # 140 had the lowest average monthly temperature across time. Similar to

Table 3.8 Monthly mean ground level temperature values ± standard deviation for the eight headwater complex study sites between December 2009 and August 2010. Statistics are displayed for comparisons between headwater complex disturbance score groups

D-score	Site[a]	December[b]	January	February	March	April	May	June	July	August[b]
Reference	83	0.22 ± 0.81	0.13 ± 1.32	−0.34 ± 0.26	4.80 ± 3.94	11.03 ± 5.54	13.98 ± 4.09	18.14 ± 2.78	20.59 ± 2.95	20.82 ± 2.46
standard	188	0.25 ± 0.20	−0.12 ± 0.70	−0.43 ± 0.32	3.82 ± 3.36	9.60 ± 3.20	13.15 ± 3.52	17.31 ± 2.29	19.22 ± 2.19	21.22 ± 1.89
	60	0.66 ± 0.42	0.30 ± 1.26	−0.28 ± 0.50	4.49 ± 3.72	9.92 ± 3.71	13.25 ± 3.58	17.16 ± 2.17	19.17 ± 2.31	20.94 ± 1.65
	13	1.34 ± 0.50	0.56 ± 0.85	−0.42 ± 0.43	3.47 ± 2.95	8.53 ± 3.27	12.78 ± 3.53	16.45 ± 2.32	18.12 ± 2.68	21.05 ± 2.45
Stressed	140	1.23 ± 0.29	0.97 ± 1.00	0.09 ± 0.28	4.98 ± 3.70	11.11 ± 3.23	13.54 ± 2.80	17.90 ± 1.84	20.11 ± 2.08	21.35 ± 1.40
	158	0.70 ± 0.61	0.32 ± 1.33	−0.40 ± 0.51	4.98 ± 4.52	11.53 ± 5.08	15.14 ± 3.80	18.87 ± 2.11	20.78 ± 2.62	22.11 ± 1.25
	151	−0.08 ± 0.64	−0.55 ± 1.69	−0.73 ± 0.78	5.48 ± 5.18	11.40 ± 5.19	14.47 ± 3.62	18.20 ± 1.96	20.53 ± 2.43	21.00 ± 1.92
	124	0.72 ± 0.64	0.29 ± 1.50	−0.03 ± 0.36	6.33 ± 4.38	12.39 ± 3.97	15.94 ± 3.70	20.70 ± 2.75	23.12 ± 3.10	22.39 ± 0.88
T-statistic		0.0739	0.1193	0.5241	2.959	3.151	2.614	2.301	2.207	14[c]
df		6	6	6	6	6	6	6	6	–
p-value		0.9435	0.9090	0.6190	0.0253	0.0198	0.0399	0.0610	0.0695	0.1143

[a]Site numbers are identification numbers from the Riparia reference collection: http://wetlands.psu.edu/projects/reference_wetlands.asp
[b]Summary statistics are not based on a full month of data (i.e., December = 13.25 days and August = 10.5 days)
[c]Mann–Whitney *U* test

microtopography, we expect that this too could be attributed to the relatively mature forest canopy within and buffering the site.

Soil

Soil properties are known to be tightly linked to microbial communities and their processes, as soil is both a resource for and a product of biogeochemical cycling. As mentioned in examples above, changes in both landscape and site structural components can alter soil conditions in such a way as to affect microbial processes. Past condition assessments of wetland ecosystems have revealed that several soil properties differ in a fairly predictable manner across land use legacies and/or current surrounding land use gradients. For example, reference standard wetlands tend to have higher levels of organic matter or organic carbon (Spencer et al. 1998; Freeland et al. 1999; Dinesh et al. 2004; Cohen et al. 2005; Reiss 2006; Rokosch et al. 2009; Cleveland et al. 2011), lower bulk densities (Spencer et al. 1998; Innis et al. 2000; Pennings et al. 2002; Reiss 2006), and lower soil pH levels (Reiss 2006; Cleveland et al. 2011) compared to those which are considered "impacted" or "degraded."

We found some of these same patterns when comparing our disturbance score groups. We found little annual variation (exception: Θv for variability metrics in Nov 2006, Table 3.10) in study site means and variability metrics. However, there were some significant differences between disturbance score groups for soil properties across all the dates in which they were measured. Some of the differences between disturbance groups were due to the presence of an organic horizon at three of the reference standard complexes, rather than due to differences in mineral soil horizons between disturbance score groups. For example, reference standard complexes had higher concentrations of SOM and N-NH_4^+, compared to stressed complexes (Table 3.9). However, this was only true when comparing soil surface strata, where the mineral stratum was not different between disturbance score groups. As expected, we also found relatively higher SOM levels in the mineral soils that were located in inundated portions of the headwater complexes. Many of the differences in carbon measures in this study and other studies can be attributed to past land uses; many agricultural, which are expected to have aerated and homogenized soils, accelerating decomposition rates. Further, the absence of organic horizons at stressed complexes could be due to the time requirements needed for the development of a suitable microclimate and litter inputs, and/or lasting hydrologic changes to these systems.

We also found pHw levels to be significantly higher in stressed complexes compared to reference standard complexes (Table 3.10). In this case there were no significant differences between soil strata at reference standard complexes. Higher pH levels in soil and water have also been associated with legacy effects and surrounding land uses in a number of studies (Bruland and Richardson 2005; Reiss 2006; Cohen et al. 2008; Cleveland et al. 2011). We suspect that the lower pH levels at

Table 3.9 Site means of soil properties (± standard deviation) for the eight headwater complex study sites in August 2007. Soil properties include soil organic matter (SOM), available ammonium (N-NH_4^+), and available nitrite + nitrate (N-NO_2^- + NO_3^-)

D-score and stratum	Site	SOM (%)			N-NH_4^+ (μg N · g^{-1} soil)			N-NO_2^- + NO_3^- (μg N · g^{-1} soil)		
Reference standard surface	188	14.28 ± 12.01			10.07 ± 9.76			0.00 ± 0.00		
	60	30.09 ± 14.69			20.02 ± 11.05			0.02 ± 0.10		
	13	27.71 ± 17.03			25.31 ± 22.35			0.38 ± 0.98		
Reference standard mineral	83	9.05 ± 2.65			6.46 ± 2.57			0.01 ± 0.07		
	188	8.65 ± 2.36			5.28 ± 1.45			0.00 ± 0.00		
	60	15.44 ± 6.46			9.41 ± 5.45			0.01 ± 0.09		
	13	11.62 ± 8.09			7.15 ± 3.29			0.35 ± 0.98		
Stressed mineral and surface	140	7.96 ± 2.21			6.01 ± 2.29			0.24 ± 0.40		
	158	8.20 ± 1.37			5.80 ± 2.79			0.27 ± 0.46		
	151	7.70 ± 1.00			5.92 ± 2.56			1.56 ± 1.73		
	124	12.61 ± 5.39			12.98 ± 9.13			1.84 ± 5.16		
		Mean*	SD	CoV	Mean*	SD*	CoV	Mean*	SD*	CoV
F-statistic		9.186	22.188	4.679	6.406	7.901	5.033	3.775	2.173	0.656
df		2	2	2	2	2	2	2	2	2
p-value		0.0085	0.0005	0.0451	0.0218	0.0128	0.0385	0.0701	0.1762	0.5445
Tukey test		RS RM SM	RS RM SM	RS RM SM	RS RM SM	RS RM SM	RS RM SM	RS RM SM	RS RM SM	RS RM SM

Reference standard complexes were broken into two strata, surface and mineral. The surface stratum included ground level samples, which could be organic horizon or mineral horizon samples. Site # 83 did not contain an organic horizon and therefore was not included in this stratum. The mineral stratum included mineral samples collected from either the surface or below the organic horizon. Stressed complexes did not contain an organic horizon, thus the mineral stratum was also the surface stratum. One-way ANOVA results for means, standard deviations (SD), and coefficients of variation (CoV) are displayed for comparisons between headwater complex disturbance score groups of different strata. Tukey post hoc significant differences are displayed by *underlines* (RS = reference standard surface, RM = reference standard mineral, and SM = stressed mineral). *Broken lines* indicated significant differences at an alpha <0.05.
Date were log transformed

Table 3.10 Two-way ANOVA results comparing soil properties between disturbance score groups/layers and among sampling dates. Soil properties include pHw (in water), volumetric water content (Θ*v*), and gravimetric water content (Θ*g*)

Factor		Parameter	df	*F*-statistic	*p*-value	Tukey honestly significant differences
pHw	Mean	Disturbance group	2	45.85	<0.0001	RS RM SM
		Month	3	0.168	0.9171	
		Interaction	6	0.035	0.9998	
	SD	Disturbance group	2	1.217	0.3096	
		Month	3	0.060	0.9806	
		Interaction	6	0.013	1.0000	
	CoV	Disturbance group	2	4.912	0.0138	RS RM SM
		Month	3	0.035	0.9910	
		Interaction	6	0.022	1.0000	
Θv	SD	Disturbance group	1	40.77	<0.0001	R S
		Month	3	3.592	0.0283	Nov06 Aug06 May07 Aug07
		Interaction	3	2.370	0.0957	
	CoV	Disturbance group	1	31.64	<0.0001	R S[a]
		Month	3	10.74	0.0001	
		Interaction	3	3.440	0.0327	
Θ*g*	SD	Disturbance group	2	5.380	0.0117	RS RM SM
		Month	2	1.956	0.1634	
		Interaction	4	0.492	0.7415	
	CoV	Disturbance group	1	9.916	0.0007	RS RM SM
		Month	3	1.379	0.271	
		Interaction	3	1.167	0.3501	

Disturbance score groups included reference standard complexes and stressed complexes. Reference standard complexes were broken into two layers, surface and mineral. The surface layer included ground level samples, which could be organic horizon or mineral horizon samples. Site # 83 did not contain an organic horizon and, therefore, was not included in this layer. The mineral layer included mineral samples collected from either the surface or below the organic horizon. Stressed complexes did not contain an organic horizon, thus the mineral layer was also the surface layer. Θ*v* measurements were only collected on the surface. Sampling dates included August 2006 (only pHw and Θ*v*), November 2006, May 2007, and August 2007. Tukey post hoc significant differences are displayed by *underlines* (RS = reference standard surface, RM = reference standard mineral, and SM = stressed mineral, or August 06 = Aug06, November 2006 = Nov06, May 2007 = May07, and August 2007 = Aug07). *Broken lines* indicated significant differences at an alpha < 0.05

[a]Due to the significant interaction term *t*-tests were conducted for comparisons of disturbance score groups for each month sampled. All months showed significant difference between disturbance score groups, except for May 2006

reference standard complexes are in part due to the high concentrations of SOM in the surface layer. SOM tends to acidify soils by forming soluble complexes with base cations, which are then leached (Brady and Weil 2002). In addition, because SOM itself contains a number of acid functional groups, it provides a source of H^+ ions (Brady and Weil 2002). Further, the mineral soils below this organic horizon are still within the rooting zone, where base cations might also be translocated into plant biomass (e.g., Yamashita et al. 2008). This is further confirmed by lower concentrations of exchangeable cations (i.e., Ca and Mg) found in both soil strata of reference standard complex soils (Moon 2012).

Although there were no statistically significant differences among disturbance score groups for N-NO_2^- + NO_3^- mean concentrations, with one exception (i.e., Site # 13) N-NO_2^- + NO_3^- concentrations tended to be higher in stressed complexes. Cleveland et al. (2011) found similar trends across the disturbance gradient for relative N-NO_2^- + NO_3^- availability from free resin bags left at sites for 28 days during their 2008 mid-summer sampling period. We suspect that nitrification rates are higher in stressed complexes given their warmer drier surface soil conditions. Nitrifying bacterial growth is also inhibited in acidic soils where nitrification rates are drastically reduced by pH levels at and below 6.0 (e.g.,Wild et al.1971). Cleveland et al. (2011) results support this relationship, showing that of the parameters they measured pH was the best predictor of N-NO_2^- + NO_3^- availability at the sample level. Finally, similar to Hurd and Raynal (2004) the high concentrations of N-NO_2^- + NO_3^- found in a few patches of Site # 13 could be attributed to the Speckled Alder (*Alnus incana*), an actinorhizal N^2-fixing shrub found at this site.

Although there have been a number of studies looking at the relationships between anthropogenic disturbance and mean values of soil properties, fewer studies have been conducted to evaluate heterogeneity in soil properties in wetlands across disturbance gradients. Results from this case study reveal clear differences in measures of site variability for SOM, N-NH_4^+, and soil moisture measurements between disturbance score groups. For these soil properties reference standard complexes had higher levels of variation compared to stressed complexes. Ranges for SOM at reference standard complexes were comparable to ranges found in similar spatial studies (Gallardo 2003; Bruland and Richardson 2005; Cohen et al. 2008). However, with the exception of Site # 124, SOM ranges were relatively lower in stressed complexes compared to wetlands surrounded by anthropogenic activities or restored wetlands in other studies (Bruland and Richardson 2005; Cohen et al. 2008). In addition, all study sites had notably lower ranges of N-NH_4^+ compared to those found by Gallardo (2003) in a floodplain forest, but were comparable to those found in natural and created freshwater wetlands of Virginia (Moser et al. 2009) and a seasonal freshwater wetland in West Virginia (Dick and Gilliam 2007).

3.3 Concluding Remarks

In order to provide a summary snapshot view of the differences found in headwater complexes with different disturbance scores, we put each of the disturbance score groups in terms of a wetland conceptual model presented by Mitsch and Gosselink (2000) (Fig. 3.9). This model focuses on three major wetland components: the existences of, and interactions among: (1) hydrology, (2) biota, and (3) the physiochemical environment. The hydrologic regime, with concurrent movement of sediment and

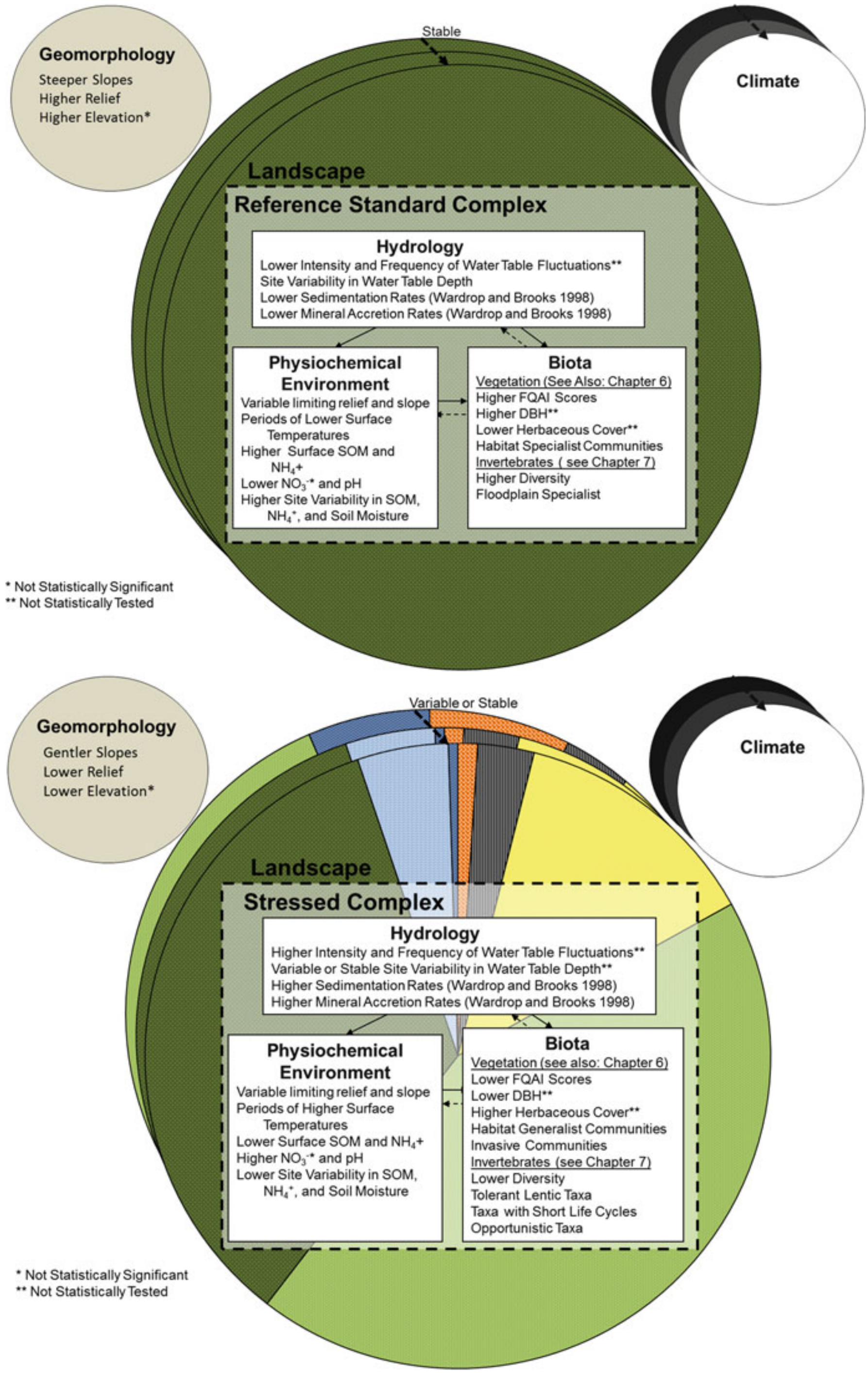

Fig. 3.9 Conceptual models of landscape scale and site scale condition at (**a**) reference standard and (**b**) stressed headwater complex study sites

chemicals, is thought of as the primary modifier of the physiochemical environment and the biotic composition, with subsequent feedbacks. Geomorphology and climate are traditionally layered onto this model and define the potential for a wetland to exist in a given location, with wetlands most likely in low gradient topographies of cool or wet climates (Mitsch and Gosselink 2000). Given a wetland's energetic openness, we add to this model the spatial and temporal dimensions of the landscapes to which our wetlands belonged.

Our comparison of disturbance score groups revealed clear relative differences in many of the components of this model. Stressed complexes existed in broader valleys with gentler slopes than our reference standard complexes. They existed in landscapes which varied in both historical land use and frequency of land use change (through late 1800s). Greater than 50% and >16% of current land cover were of non-forested, non-wetland, or non-water land cover classes in the 1-km and 100-m areal extents, respectively, while reference standard complexes had <16 and 0% at the same extents. On average, stressed complexes had higher LDIs, higher percentages of impervious surface, and lower percentages of core forest across all spatial scales compared to their reference standard counterparts. At the site scale, stressed complexes had more frequent and intense fluctuations in their hydrographs and higher herbaceous cover with a predominance of invasive and habitat generalist species. Ground level temperatures readings were slightly elevated for spring and summer months at stressed complexes. Finally, soils at these sites had lower levels of SOM and NH_4^+, higher levels of NO_3^- and pHw (for all dates measured), and lower site level variability for SOM, NH_4^+, and soil moisture.

One future avenue of management-driven inquiry for structural characteristic datasets such as those described above, are for use in process-based models. Direct measurements of ecosystem processes is always desirable, but is often infeasible because of resource constraints. Process-based models, such as PnET-N-DNDC or Wetland DNDC, provide a method for using baseline structural datasets to predict ecosystem processes (Li et al. 2000; Zhang et al. 2002). These specific models are broken down into ecological drivers (i.e., climate, soil properties, vegetation, anthropogenic activities, and hydrology) that act on the soil environment (e.g., temperature, moisture, pH, substrates, etc.), which in turn control processes (e.g., nitrification, denitrification, decomposition, etc.) (Li et al. 2000). Although these models can have limitations (Lamers et al. 2007), they provide a way to begin to understand the collective effects of wetland components on ecosystem processes mediated by microbial communities, as well as to understand which components are most influential.

A second avenue of inquiry for management application is the evaluation of the relevant spatial and temporal scales of stressors or multiple-stress functions. As shown here, having a narrow but stable buffer strip around Site # 140, made it more similar in many structural condition measures to reference standard complexes, than to other complexes within its stressed category. While stressors emanating from a 100-m wide buffer surrounding the site might be able to be addressed via BMPs, the entirety of stressors associated with the 1-km radius area surrounding the site clearly exceeds the authority and capability of most management programs. Identifying the relevant spatial and temporal scales required to protect or restore an ecosystem's processes or other condition measures allows us to set reasonable expectations for policy and management programs.

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Chapter 4
Hydrology of Mid-Atlantic Freshwater Wetlands

Kristen C. Hychka, Robert P. Brooks, and C. Andrew Cole

Abstract Hydrology is a key variable in the structure and function of a wetland; it is a primary determinant of wetland type, and it drives many of the functions a wetland performs and in turn the services it provides. However, wetland hydrology has been understudied. Efforts by scientists from Riparia, a wetland and aquatic systems research center at Penn State University, have advanced the understanding of wetland hydrology in the Mid-Atlantic Region over the past two decades primarily through a series of studies at a set of long-term monitoring sites. This work contributed to four primary issues in wetland hydrology: validation of regional hydrogeomorphic classification schemes, establishment of reference criteria for monitoring and assessment, identification of targets for restoration or mitigation, and evaluation of the hydrologic behavior of created vs. non-created wetlands. This chapter (1) summarizes some of the key findings of hydrologic studies of wetlands from the published and non-published research of wetland scientists associated with Riparia and secondarily, (2) describes general, seasonal, and inter-annual hydrologic patterns of the water level data that has been collected at some of the long-term monitoring sites or "reference sites."

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4.1 Introduction

Hydrology is a key variable in the structure and function of a wetland; it is a primary determinant of wetland type, and it drives many of the functions a wetland performs and in turn the services it provides. However, wetland hydrology has been understudied. Though there has been important work on understanding complete water budgets at a single wetland, there are fewer studies that look across a number of sites encompassing a range of wetland types under a range of human disturbance settings. And, there remains a major technical challenge "to determine an average or characteristic hydroperiod for sites on which there is no hydrologic data, or for which hydrologic data cover only a short period of time" (p. 91) (National Research Council 1995). This characteristic hydroperiod, or hydrologic regime, of a wetland can be characterized by the magnitude, frequency, duration, timing, and rate of change of hydrologic events such as inundation or soil saturation (Poff et al. 1997). Modifications to any aspect of this regime can have cascading effects on aquatic ecosystems (Karr 1991; Karr and Chu 1999). The timing and duration of inundation and saturation can influence recruitment from seedbanks (Seabloom et al. 1998) and survival of herbaceous and woody plant species (Harris and Marshall 1963; Mountford and Chapman 1993; Poiani and Johnson 1993; Miller and Zedler 2003; Magee and Kentula 2005). Reduction in the magnitude and dynamics of flooding can result in a reduction in the biophysical complexity of a wetland ecosystem (Richter et al. 2003), which in turn can shift the invertebrate communities both on the surface (Richards and Host 1994; Lammert and Allan 1999) and in hyporheic zones (Poole et al. 2006). Further, the timing, duration, and dynamics of the hydrologic regime influence the biogeochemical environment of wetland soils (Richardson and Vepraskas 2001).

Riparia, formerly the Cooperative Wetlands Center, is a wetland and aquatic systems research center at Penn State University. Efforts by Riparia scientists have advanced the understanding of wetland hydrology over the past two decades. The objectives of this chapter are (1) to summarize some of the key findings of hydrologic studies of wetlands in the Mid-Atlantic Region (MAR) from the published and non-published research of wetland scientists associated with Riparia and secondarily, (2) to describe general, seasonal, and inter-annual hydrologic patterns of the water level data that has been collected at some of the long-term monitoring sites or "reference sites." The discussion and analysis in this chapter are focused on freshwater wetlands, which has been the primary focus of research within the MAR by researchers in Riparia.

4.2 Riparia's Hydrologic Studies

Several long-term, regional, multi-wetland hydrologic studies began in the 1990s in the Pacific Northwest (Shaffer et al. 1999), North Carolina (Rheinhardt et al. 1999), Ohio (Fennessy et al. 2004), and in Delaware and Maryland (Weller et al. 2007;

Whigham et al. 2007) in order to better understand wetland hydrology across wetland types and disturbance conditions. In 1993 researchers from Riparia began to establish a network of monitoring sites in wetlands across types and across a range of land use settings to help to fill the knowledge gap about wetland hydrology in the MAR, particularly in the Appalachian Plateau and the Ridge and Valley Physiographic Provinces. The sites were classified following the hydrogeomorphic (HGM) approach, in which two of the key characteristics of the classification scheme are water source and hydrodynamics (Brinson 1993). All of the work at the reference sites contributed to four primary issues in wetland hydrology: validation of regional HGM classification schemes, establishment of reference criteria for monitoring and assessment, identification of targets for restoration or mitigation, and evaluation of the hydrologic behavior of created vs. non-created wetlands (Cole and Brooks 2000a). This section chronologically presents the goals and key findings of the hydrologic wetland studies that have been performed in Riparia and have contributed to the understanding of wetland hydrology in the four key areas mentioned above.

First, a note on wetland classification used in this chapter. During the period 1993 until the present, the regional HGM classification system has been modified as more was learned about these ecosystems. Although this book generally follows the most recent terminology (Brooks et al. 2011), in this chapter we refer to the original terms to facilitate continuity with most of the published papers on wetland hydrology. When first mentioned, both terms are provided. Readers can refer to Chap. 2 of this book for more details about changes in wetland classification.

Initial efforts confirmed the a priori HGM classification by finding differences between the classes particularly in terms of median duration of saturation in the root zone (upper 30 cm). Cole et al. (1997) instrumented 24 wetlands across four HGM subclasses: slope, depression (riparian depression as per Cole et al. 1997), riverine lower perennial (mainstem floodplain as per Cole et al. 1997), and riverine upper perennial (headwater floodplain as per Cole et al. 1997) with shallow water level monitoring wells and piezometers where monthly measurements were taken during the growing season (Cole et al. 1997). They found riparian depressions and slopes had more groundwater contribution than the floodplain classes and duration of saturation in the growing zone ranged across classes with the most saturation in riparian depressions, the least in the floodplain systems, and slopes in between. They also concluded that headwater floodplains were fed primarily through overland flow while mainstem floodplains were driven by overbank flooding. The next efforts continued to validate the regional HGM classification scheme, and built upon earlier findings by collecting data at more reference wetlands ($n = 30$) in the same HGM subclasses as the previous study, collecting water level data outside the growing season, and looking at differences in disturbance across sites (Cole and Brooks 2000a). Automatic water-level recorders took measurements every 3–6 h throughout the year. There were similar differences between subclasses in terms of percent time of water in the root zone for the whole year and not just the growing season. Disturbance was also a key factor in wetland hydrology, and may override HGM subclass characteristics. For example, median water levels for moderately disturbed depressions were standing water (8 cm) while low disturbance depressions were saturated (−7 cm). Using data from this study and a similar study of hydrology in Oregon wetlands, it was also determined that low frequency measurements (7 days or less) could accurately predict annual and monthly water level statistics (Shaffer et al. 2000).

Studies also investigated the differences in hydrology between natural and created systems in Pennsylvania (Cole and Brooks 2000b) and New York (Cole et al. 2006). In the Pennsylvania work, created mainstem floodplain systems had much higher median water depths, and most notably much more standing water throughout the year, than did natural systems. Similarly, in the New York study, created sites ($n = 5$) were generally wetter than comparable natural sites ($n = 3$), and, despite designing for saturated soils, many had open water habitats. These studies provide a particularly important contribution to the understanding of created wetlands, as some of the data were collected over a 10-year period. Most hydrologic monitoring of constructed wetlands does not extend beyond 1 year (Zedler 2000), while vegetation and hydric soil conditions may take several years to decades to establish and hydrologic data from any single year may not represent mean hydrologic behaviors because of interannual variation in weather conditions.

Comparisons of wetland hydrology data from Pennsylvania and Oregon were used to evaluate the transferability of HGM functional models across regions (Cole et al. 2002). Three years of hydrology data from wetlands ($n=18$ in Pennsylvania and $n=15$ in Oregon) in three HGM subclasses (slope, headwater floodplain, and mainstem floodplain) were compared across a range of disturbance levels. The hydrology was similar for the slope wetlands in terms of monthly median and interquartile range of depth of water, percent of time in the root zone, and the percent of time soils were saturated or inundated across regions, but not for the two riverine subclasses. Variation between years was relatively small, but did affect the percent of time the median water levels were in the root zone for the headwater subclasses. The wettest periods were in the spring (March–April) while the driest periods were in the late fall to early winter (November–December). Additionally, less standing water in Pennsylvania wetlands may be due to differences in soils, particularly higher percentages of fine sediments at the Oregon sites. Based on these data, HGM functional models were found to be robust across regions for the slope subclass but not for the riverine subclasses.

The transferability of HGM functional models was also evaluated though a study that took the HGM classification north and south along the Appalachians into the Catskills and Adirondacks of New York and northwestern Virginia (Cole et al. 2008; Peterson-Smith et al. 2009). Water level data were collected at 6-h increments for up to 3 years at 53 minimally disturbed wetlands in three HGM subclasses (headwater floodplain, slope, and riparian depression). They found that headwater floodplains and slope wetlands were hydrologically similar by subclass between Pennsylvania and Virginia, but different in New York, while riparian depressions were similar throughout. Hydrologic differences were attributed to high beaver activity and snow cover in the New York sites. Again, even within the MAR some of the HGM functional models are robust and others are not based on HGM subclass.

Hydrology has played a key role in a number of other studies performed by Riparia researchers, even if it was not the primary focus of study. One such study identified plant species that are indicators of groundwater contribution to a wetland (Goslee et al. 1997). Another study, done in collaboration with the USGS, used hydrology to help explain amphibian breeding in small, isolated wetlands in the Delaware Water Gap (Julian 2009). Julian assessed both the connectivity of wetlands to surface water bodies (strictly isolated, seasonally isolated, or permanently connected) and hydrologic stability (the proportion of wetland area inundated in June relative to the high water period (mid-April)) based on visual surveys of 125 wetlands. One wetland biogeochemical study used hydrology to characterize wetlands in looking at decomposition in 12 headwater complex wetlands across a range of disturbance (Ryan 2005). The hydrologic regime was characterized as moderately inundated, saturated, moderately saturated, or dry based on a clustering of temperature and hydrologic metrics generated from 1 year of water level data (percent time inundated or saturated and the number of flooding events in three duration categories: less than or equal to 1 day, 1–2, 2–7 days, and greater than 1 week). Continuing work is also contributing to the understanding of wetlands as part of the larger riverine landscape, which is an important interdisciplinary problem in achieving integrated water resources management (Ward 1989; Thorp et al. 2006). Researchers in Riparia have contributed to the integration of wetland and riverine studies in some of the work outlined above, but also in other efforts with aquatic invertebrates (Laubscher et al. 2004) and with the development of integrated rapid assessment techniques (Brooks et al. 2009). Finally, there is on-going work quantifying the ecosystem services provided by wetlands including provision of habitat, flood storage, and nitrogen attenuation and how these services relate to hydrology in the face of land use and climate change (Shortle et al. 2009; Hychka 2010; Yetter et al. 2011; Hychka et al. in prep).

4.3 Water Level Patterns by Type and Disturbance

In this section we will present some of the hydrologic data collected through Riparia and associated researchers in order to build upon the analyses in the literature presented above. Specifically, we will discuss some of the hydrologic behaviors of the wetlands across wetland types and disturbance in relation to seasonal and interannual response to climatic drivers. The relationship between wetland hydrology and climatic drivers has not been studied extensively by Riparia researchers and is important particularly in the face of forecasted climatic changes.

4.3.1 Sites

Data are presented for a subset of wetlands that have been monitored by Riparia as part of a reference collection (www.riparia.psu.edu). The subset was selected to cover a range of wetland types and disturbance levels. Sites were excluded if the records spanned less than 40% of the study period, if site-level disturbances were so unique that the water level information might not be generalizable, or if they were constructed wetlands. The coarse condition classes used (reference standard, least disturbed, and disturbed) are based on landscape and site-level characteristics of the wetlands, where reference standard sites were chosen to represent high ecological integrity. The sites include depressions (reference standard $n=4$, disturbed $n=2$), riverine upper perennial or headwater (reference standard $n=3$, disturbed $n=3$), riverine lower perennial or mainstem (least disturbed $n=1$, disturbed $n=2$), and slope (reference standard $n=2$, disturbed $n=4$) (Table 4.1). Water levels were recorded using automatic recording devices installed in shallow, slotted, PVC well casings (Cole and Brooks 2000a). Water levels were recorded every 4 h over a 13-year period (1996–2009) with individual well records covering 43–89% of that time. The gaps in the data are not random and are more likely to occur during winter months and extreme events where the water levels may have gone above or below the measurement interval for the instrument.

This dataset is unique and highly valuable in the region for providing information about wetland hydrology over multiple years and across wetland types and disturbance levels. The discontinuous nature of the data does make it difficult to perform certain statistical analysis; even means may not be representative and many time series analyses require complete or nearly complete series. However, there is a lot to be learned from exploratory statistics and visualizations, which is the approach that will be taken in this chapter.

4.3.2 Climate

Seasonal: Temperature and evapotranspiration (ET) patterns of the MAR of the USA have a strong seasonal signal with warm summers with high ET and cool or cold winters with very low ET. Precipitation, on the other hand, does not show a strong annual cycle (Najjar 1999). More specifically, the Köppen climate regions in the Mid-Atlantic are all Mid-Latitude Humid with the north and the spine of the Appalachians in Severe Mid-Latitude Humid continental with a severe winter, no dry season, and either a hot (Dfa) or warm (Dfb) summer (Godfrey 1999) (Fig. 4.1). Most of the non-Appalachian portions of Virginia, southern Maryland, and the Delmarva Peninsula are Mild Mid-Latitude Humid subtropical with a mild winter, no dry season, and either a hot (Cfa) or warm summer (Cfb). The result of the seasonal patterns of temperature, evaporation, and precipitation on the region's hydrology is that typically spring has higher levels of soil moisture and stream flows, while the summers have lower soil moisture and mean stream flows (Fig. 4.2) (Pennsylvania NRCS 2000). And, though the region has a humid climate, the late fall and early spring are the only periods of recharge in the hydrologic cycle when the soils are not frozen and there is low ET demand from plants (Swistock 2007).

Inter-annual: There is also a wide range in the inter-annual climatic conditions in the region. Precipitation and temperature vary on a roughly decadal cycle, which may be driven by the North Atlantic Decadal Oscillation (NAO) with winter

Table 4.1 Sites with hydrologic data used for exploring freshwater wetland hydrology in this chapter

Class	Disturbance	Site name	Site #	Start	End	Days with records	*n*	% Total period with data
Depression	Ref std	Clarks Trail	13	10/22/96	03/09/08	2840.50	11,362	0.61
Depression	Ref std	McCall Dam	5	12/07/96	05/15/08	2941.75	11,767	0.63
Depression	Ref std	Sand Spring	6	10/22/96	07/20/07	3454.25	13,817	0.74
Depression	Ref std	Whipple Dam	10	09/05/97	01/09/08	3024.50	12,098	0.65
Depression	Disturbed	Canoe Creek	7	10/23/96	06/11/09	3910.00	15,640	0.84
Depression	Disturbed	Tadpole	52	06/17/97	06/27/08	3238.75	12,955	0.70
Riverine Upper Perennial	Ref std	Laurel Run	60	10/09/97	10/31/07	3079.50	12,318	0.66
Riverine Upper Perennial	Ref std	Tuscarora	87	06/28/00	12/20/08	2956.25	11,825	0.64
Riverine Upper Perennial	Disturbed	Buffalo Run	18	06/11/97	07/10/09	3949.00	15,796	0.85
Riverine Upper Perennial	Disturbed	Nittany B&B	53	02/20/98	06/27/08	3205.00	12,820	0.69
Riverine Upper Perennial	Disturbed	Thompson Run	57	07/22/97	04/15/08	3313.00	13,252	0.71
Riverine Upper Perennial	Disturbed	Water Authority	26	12/08/96	03/12/04	2296.75	9,187	0.49
Riverine Lower Perennial	Least dist	Bald Eagle Creek	3	10/19/96	05/15/08	3901.50	15,606	0.84
Riverine Lower Perennial	Disturbed	Fravel	32	11/06/96	07/02/09	4125.25	16,501	0.89
Riverine Lower Perennial	Disturbed	Lock Haven	58	07/31/97	07/02/09	3655.75	14,623	0.79
Slope	Ref std	McGuire Rd	24	09/18/97	06/10/08	2975.25	11,910	0.64
Slope	Ref std	Swamp White Oak	55	07/11/97	03/16/08	3146.00	12,584	0.68
Slope	Disturbed	BESP PEM	2	12/08/96	01/06/03	2006.50	8,026	0.43
Slope	Disturbed	Cumberland Valley	67	06/17/98	05/13/05	2279.75	9,119	0.49
Slope	Disturbed	Shavers Creek	23	11/02/96	06/10/08	3428.50	13,714	0.74
Slope	Disturbed	Windy Hill Farm	23	11/02/96	04/21/05	2923.75	11,695	0.63

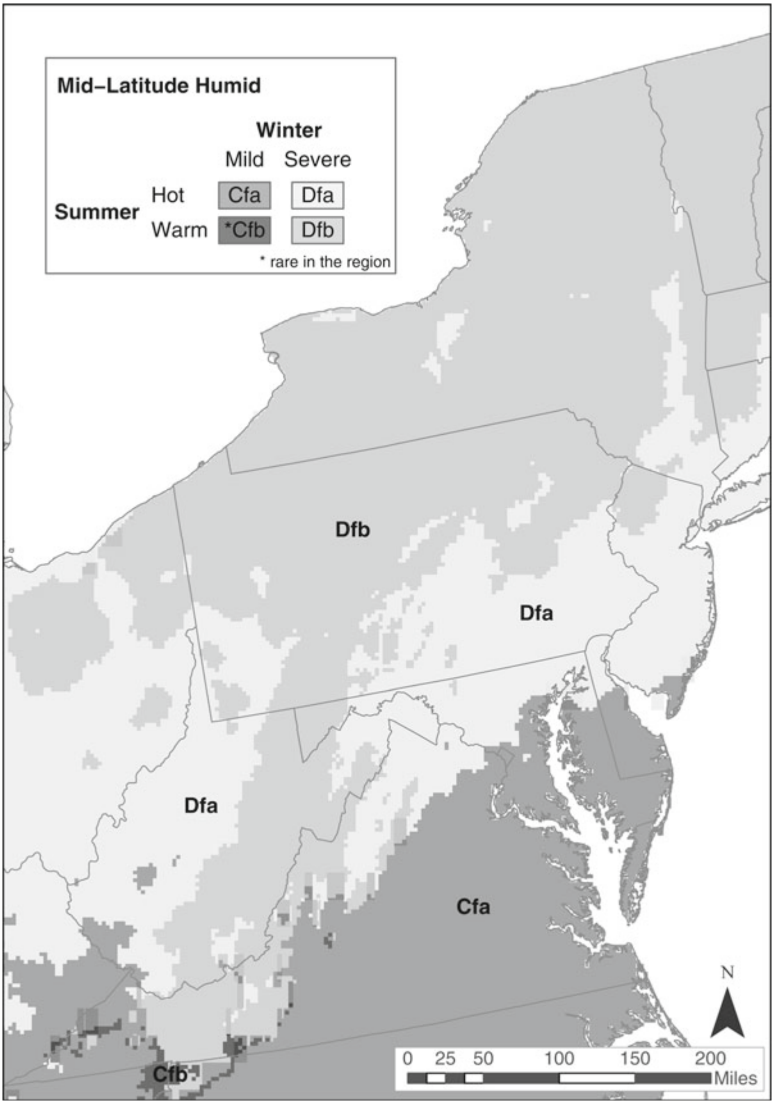

Fig. 4.1 Köppen climatic regions for the Mid-Atlantic Region

precipitation at Pennsylvania study climate stations showing an inverse relationship with the NAO (Willard and Cronin 2007; Ning et al. 2012). During the period of study there were periods of both extremely moist conditions (Palmer Drought Hydrologic Index (PHDI) +4.0 and above) and extreme drought (PDHI −4.0 and below) (Palmer 1968) (Fig. 4.3).

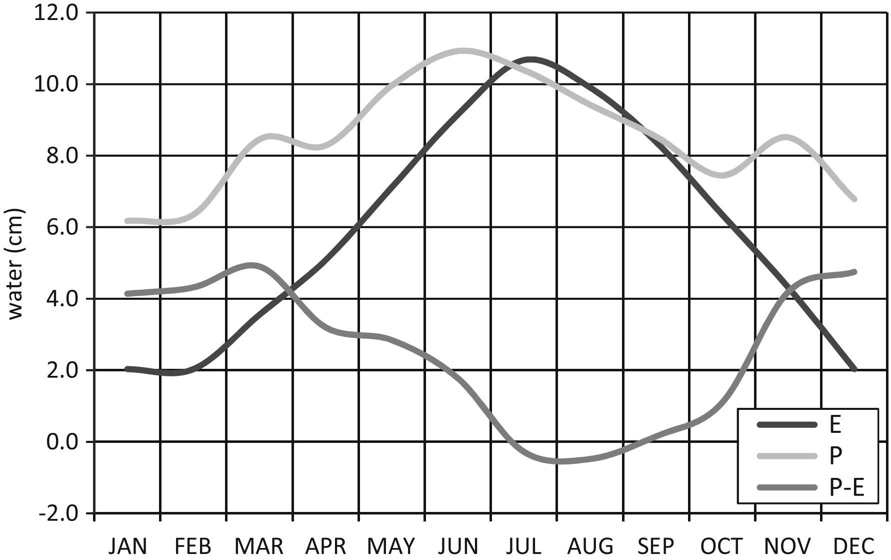

Fig. 4.2 Water balance for Centre County, Pennsylvania showing monthly evaporation (E), precipitation (P), and net water balance (P−E) all in cm. Yearly rainfall totals 101.2 cm and evaporation is 70.6 cm: http://www.pa.nrcs.usda.gov/technical/Engineering/PaRainEvapRunoff.pdf

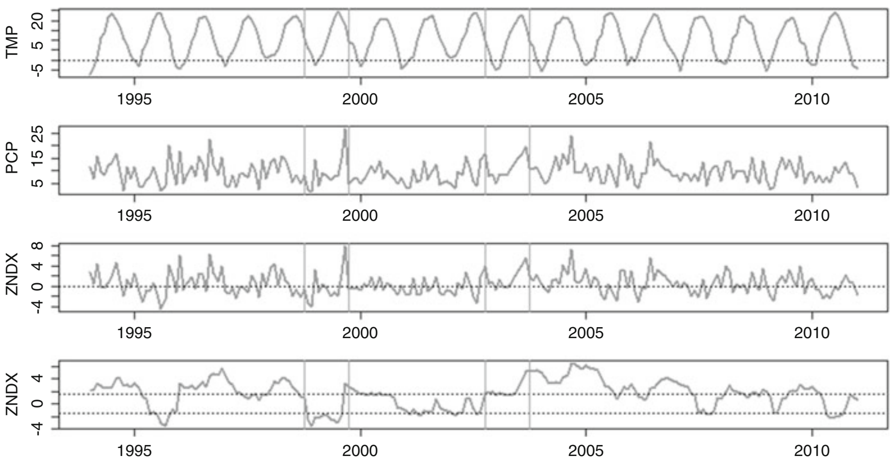

Fig. 4.3 Climatic drivers and drought indices for the Middle Susquehanna climate division. From top to bottom: mean monthly temperature (°C) (TMP), precipitation (cm) (PCP), Palmer Hydrologic Drought Index (PHDI), and the Z-index or deviation from normal precipitation (ZNDX). *Horizontal dotted lines* indicate: 0 °C on the temperature plot, the mean in the ZNDX plot, and the normal wetness range (−1.5 to 1.5) in the PHDI plot. A dry (1999) and a wet (2003) water year are demarcated with pairs of *vertical lines*

4.3.3 Hydrologic Patterns

General: Some of the general differences between wetland types shown in previous work can be seen in the simple time series plots of the individual well water levels (Fig. 4.4). There is the most variability in water levels at individual sites for the two riverine classes, a moderate level in slopes, and the least in the depressional wetlands. However, some of the hydrologic dynamics across sites are harder to determine from looking at the simple time series. Empirical cumulative distribution functions (ECDFs) are another way to visualize an entire dataset that allows for pattern detection without losing some of the details required when imposing metrics. The curves presented are for all observations (black) and for individual wells for reference (grey) and disturbed (white) sites across the four wetland types (Fig. 4.5) and represent the percent of time the water level was at or below a certain height. For example, the observations of all the depressional wetland wells are at or below approximately 5 cm half of the time and below −10 cm less than 10% of the time. ECDFs are also an effective way to convey the variability in hydrographs within a wetland type.

In the depressional wetlands, water levels are either standing water or in the root zone nearly all of the time, with more time in standing water and less time in the root zone in some of the disturbed sites. Reference riverine upper perennial sites also have water levels nearly always in the upper 30 cm with little standing water. This is in marked contrast to the disturbed sites which have much less time of saturation in the root zone and more standing water. Lower perennial riverine systems have water levels below the root zone about 40% of the time with standing water roughly 10% of the time. Lower perennial riverine wetlands showed less variability across disturbance, with the exception of the left most curve representing a much drier site than the others. Slope wetlands show little time in standing water and are saturated most of the time. All of the reference standard slope wetlands are saturated 75% of the time or more, while some of the disturbed sites follow similar curves to the reference sites, others are much drier as seen in the two left most curves.

Seasonal: Previous work on seasonality of wetland hydrology with the Riparia reference wetlands looked at slope, riverine upper perennial, and riverine lower perennial wetlands across a range of disturbance levels and showed that the wettest times of the year were in the spring (March–April) while the driest were in the late fall to early winter (November–December) (Cole et al. 2002).

Seasonality of wetland hydrology is seen in a dot plot of the median water level for meteorological spring (March, April, and May) vs. summer (June, July, and August) for the study wells by type and disturbance level (Fig. 4.6). If the median for the two seasons was the same, the point would fall on or close to the diagonal, points above the diagonal the median water level is higher in spring, and for points below the diagonal water levels are higher in summer. Similar to previous findings, the diagram shows that nearly all of the wells have higher water levels in the spring, with only three of the depressional wetlands slightly wetter in the summer. Depressional wetlands' median conditions are mostly slightly wetter in the spring,

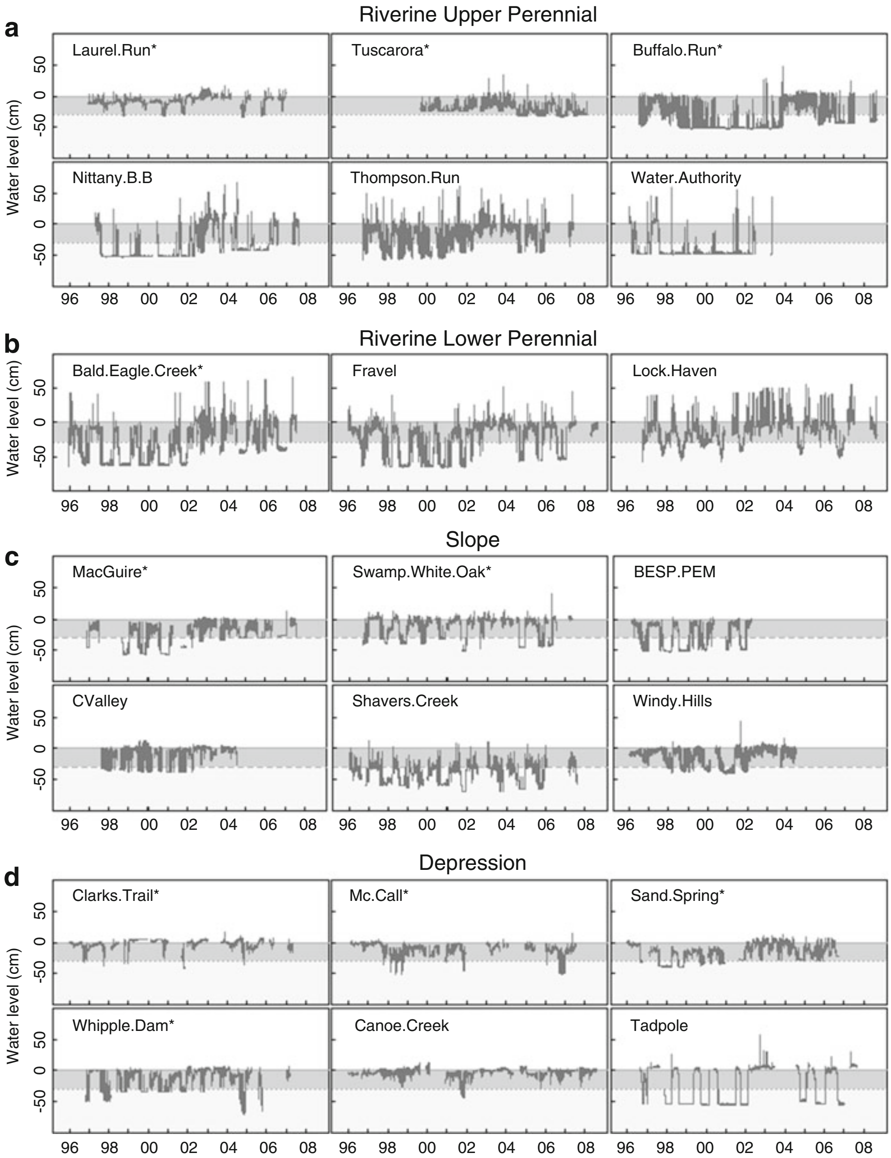

Fig. 4.4 Water levels in selected (cm) (**a**) upper perennial riverine, (**b**) lower perennial riverine, (**c**) slope, and (**d**) depressional wetlands. Ground is 0 (*solid line*), positive values are above the surface, and negative are below the surface. Growing zone is the upper 30 cm (*dashed line*)

but saturated in both spring and summer, with the notable exception of two disturbed sites where median values are in standing water in both spring and summer. Riverine upper perennial systems show a number of differences across disturbance in terms of seasonality. The reference wetlands are the only systems that have satu-

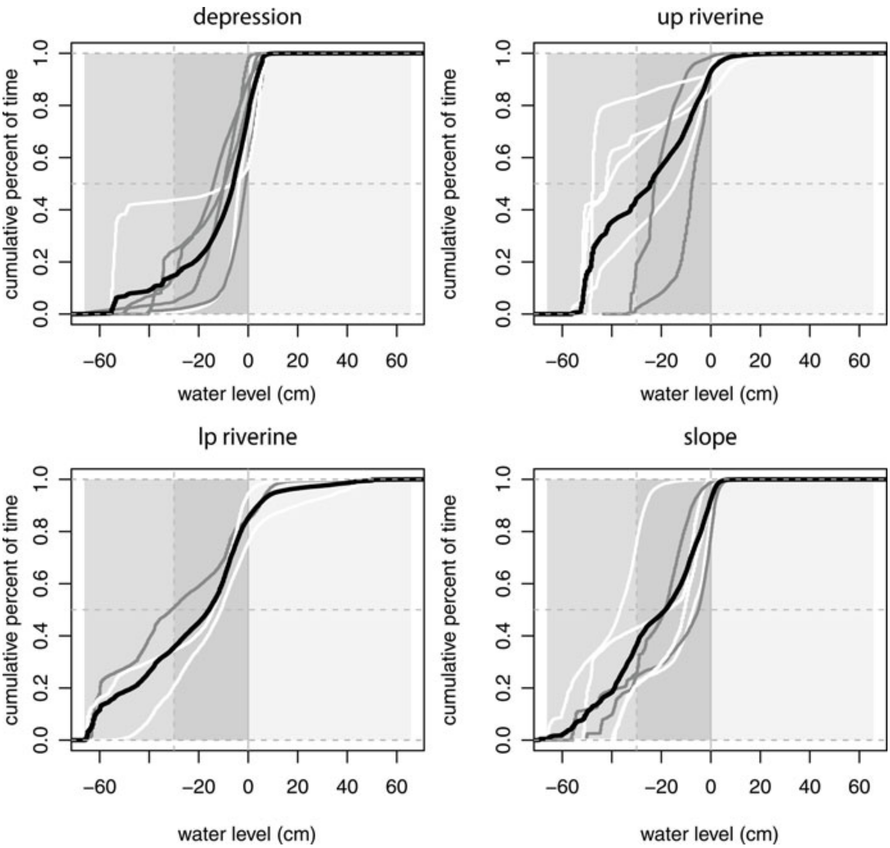

Fig. 4.5 Empirical distribution functions of water levels for individual reference condition (*grey*) and disturbed (*white*) wetlands and for all sites combined (*black*) in four classes of wetlands. The *dark shaded area* indicates the root zone (surface (0 cm) to −30 cm). Where the distribution curve crosses the *horizontal dotted line* is the depth that the water level is at or below 50% of the time

rated median conditions in both spring and summer. None of the disturbed sites had saturated median conditions in the summer and all were either saturated or not saturated in the spring, with the exception of one site that had standing water in the spring and saturated conditions in the summer. All three of the reference systems showed very little difference between spring and summer medians, while there was a much bigger range in the disturbed sites with three of the sites showing greater than 20 cm difference in median conditions between spring and summer. Riverine mainstem systems, as a whole, showed a high degree of variability between spring and summer median conditions. The degree of seasonal variation in wetland hydrology varies by HGM type and disturbance level.

Inter-annual: The period of record for these wells has spanned a range of very wet to very dry conditions (Fig. 4.3), including water year 1999 (10/1/1998–9/30/1999) which had mild to moderate drought conditions with a mean PDHI of −1.7 and

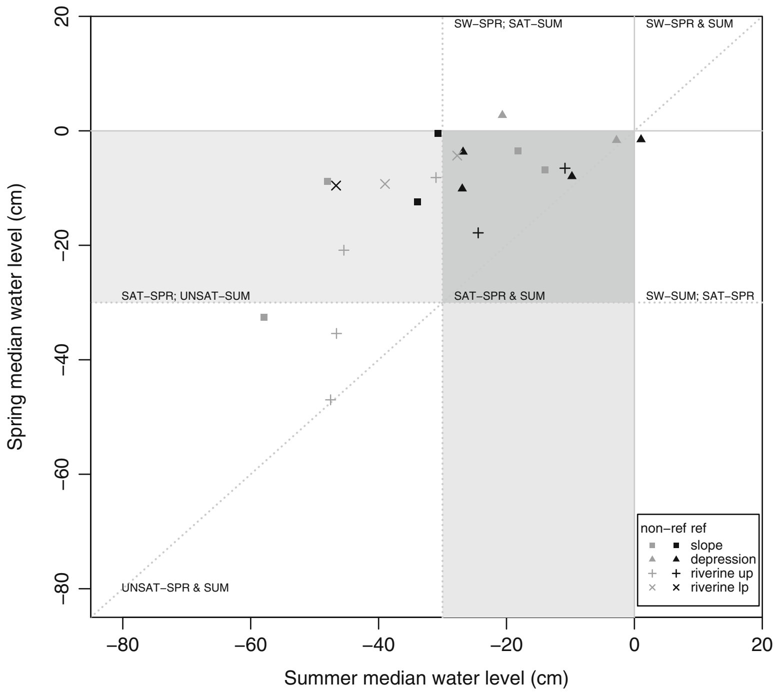

Fig. 4.6 Shows median water level (cm) in summer vs. spring for depression, upper perennial riverine, lower perennial riverine, and slope wetlands in reference standard or least disturbed (*black*) and disturbed (*grey*) condition. Wetlands with median water levels in the growing zone (upper 30 cm) for at least one season include those saturated for both spring and summer (*dark grey*, quadrant) and those saturated in spring but not summer (*light grey*, quadrant)

water year 2003 (10/1/2002–9/30/2003) which had mild to moderate wetness with a mean PDHI of 2.2. The wetlands displayed very different hydrologic behaviors during these wet and dry years. For example, depressional wetlands showed a distinct pattern of saturation in the reference wetland and inundation for the disturbed site during the spring and drawdown out of the root zone in the fall of the dry year, whereas in the wet year the reference wetland stayed saturated throughout the year with periods of inundation and the disturbed site was inundated throughout the year (Fig. 4.7a). One of the wettest riverine upper perennial wetland sites in the dry year showed a small, but distinct summer drawndown in water level, whereas in a wet year the same system showed no summer drawdown and a relatively long period of standing water in the summer and early fall (Fig. 4.7b). In a much drier, disturbed site there was little time that the root zone was saturated; however, in an extremely wet year the site was saturated most of the year and similarly had standing water for extended periods. In two lower perennial riverine sites (Fig. 4.7c), both reference and disturbed sites showed distinct seasonal patterns with a period of saturation and inundation in the spring and a drawdown out of the root zone in the summer.

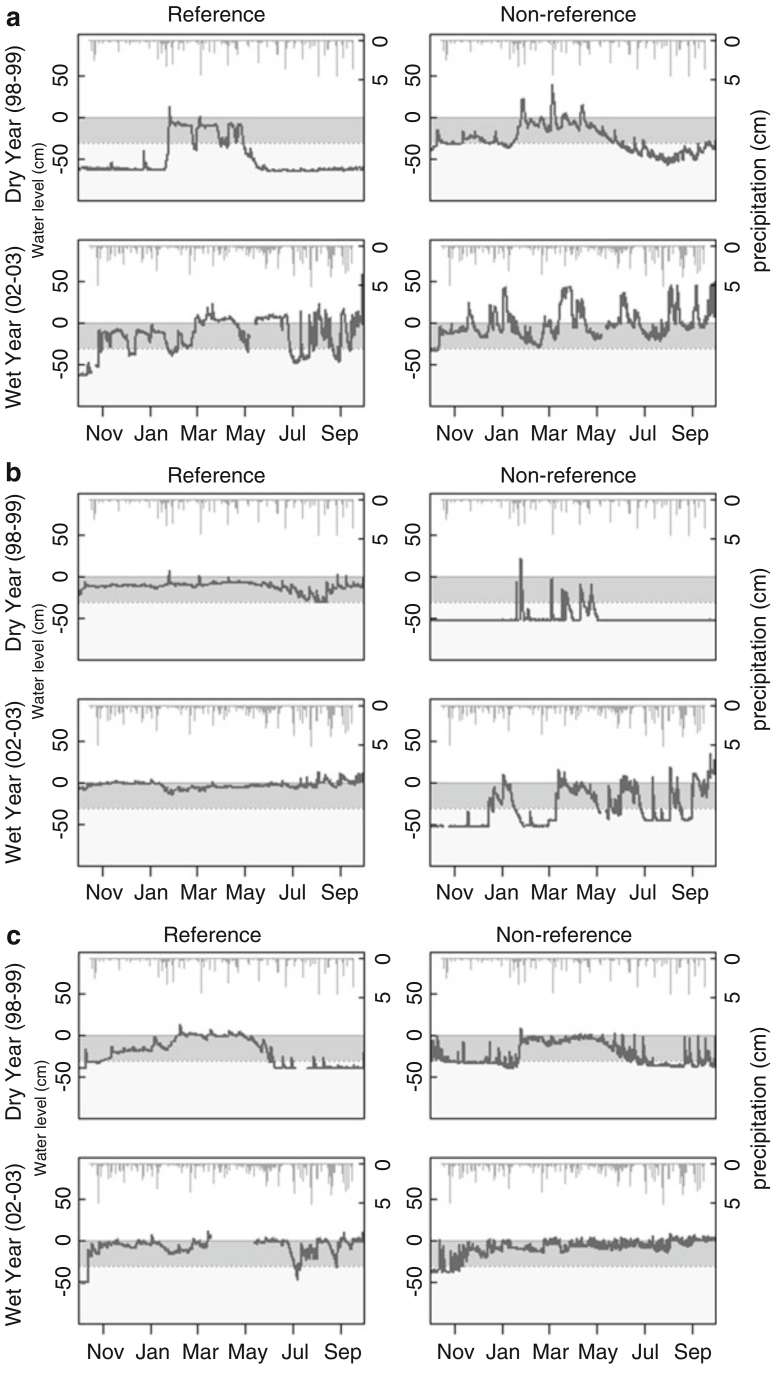

Fig. 4.7 Water level (in cm from ground level) for reference and non-reference wetlands in a dry (*top*, 1998–1999) vs. wet water year (*bottom*, 2002–2003) across four classes ((**a**) depression, (**b**) upper perennial riverine, (**c**) lower perennial riverine, and (**d**) slope). The ground level is 0 (*solid grey line*) and growing zone is the upper 30 cm (*grey dashed line*). Secondary *y*-axis shows the daily precipitation (cm)

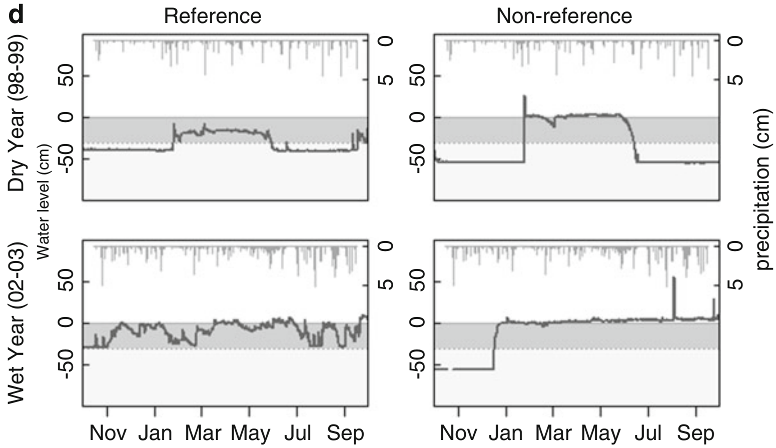

Fig. 4.7 (continued)

However, in the wet year the reference wetland was primarily inundated or saturated with a few periods where the water level fell below the root zone, and the non-reference wetland was either saturated or inundated. Slope wetlands similarly showed differences between wet and dry years, as both showed generally saturated conditions in spring with a drawdown out of the root zone in summer during dry years, and in wet years both wetlands were nearly consistently saturated with some periods of standing water (Fig. 4.7d).

In summary, the presentation of the data in this section demonstrates differences in how wetlands respond to climatic drivers such as drought and seasonal variation and has implications for the understanding of the potential vulnerability of wetlands to a changing climate.

4.4 Conclusions and Future Directions

Research by scientists at Riparia has helped to fill a critical gap in understanding about freshwater wetland hydrologic behavior. The reference wetland collection has helped to validate regional HGM classification and understand differences in wetland hydrologic behaviors across a disturbance gradient. Work comparing wetland hydrology across regions and in extending the models north and south within the Appalachians has further helped to validate the models and understand the geographic extent to which HGM classifications and models can be useful. Characterizing hydrologic characteristics of wetlands in the reference collection by HGM type and across a disturbance gradient, such as percent time the water is in the growing zone or in standing water, has established reference criteria in the MAR for monitoring and assessment. These same hydrologic characterizations can be used to set regionally appropriate targets for restoration or mitigation sites based on the disturbance in the site's setting. Extensive work with mitigated wetlands demonstrated that created wetlands do not mimic the hydrology and functioning of analogous natural wetlands, which has played an important role in efforts to ensure that wetland mitigation accounts for losses in function and not just area (Moreno-Mateos et al. 2012, see Chap. 10 of this book). Ongoing work continues to address the multiscale and multidisciplinary problems of coupling wetland hydrology and functioning with hydrologic, land use, and climate models. Though this area of research presents some "wicked problems" (Freeman 2000), it is also situated at a critically important nexus in water resources management. Additionally, the many studies that have linked hydrology to other wetland characteristics and processes are critically important in understanding what drives wetland functioning particularly across a human disturbance gradient.

Some of the patterns in wetland hydrologic behavior presented in this chapter indicate the importance of accounting for inter-annual climatic conditions, particularly drought status, when performing wetland studies and assessments. The results during extremely wet or dry years may be quite different from the predominant site conditions. Many wetland studies, however, do not extend beyond one or two field seasons due to the nature of academic studies and funding for short-term projects. The observed inter-annual variability at the Riparia reference wetlands emphasizes the importance of maintaining long-term monitoring sites to give hydrologic context to shorter wetland studies. Climate change scenarios differ in projected changes in the timing and magnitude of precipitation in the MAR, though most forecast increases in the magnitude and duration of summer time deficits (Cowell and Urban 2010). Observing past behaviors of wetlands across a disturbance gradient to past climatic extremes can give insight into possible trajectories of change in wetlands in the face of future changes in climatic drivers.

The hydrologic data collected through efforts of Riparia's scientists are unique in the region and are critically important to wetland managers, practitioners, and researchers in performing wetland assessments, designing and evaluating created wetlands, and understanding the hydrologic role of wetlands in a watershed context. Particularly in the face of changing climate and land use, it is essential to maintain and expand upon the current network of wetland hydrology monitoring. Historic wetland hydrology records are relatively short, so it is also important to maintain the collection of hydrologic covariates that have longer records, such as temperature, precipitation, and stream flow. Finally, water resources decision-making is not solely based on scientific information, but also on a blend of economic, cultural, and political factors. As the timing and availability of water in the MAR are projected to shift under climate change, it is important to use science to inform decisions-making, but also to expand multidisciplinary research that incorporates the socio-economic context in which water resource decisions are made.

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Chapter 5
Hydric Soils Across Pennsylvania Reference, Disturbed, and Mitigated Wetlands

Patrick Drohan and Robert P. Brooks

Abstract Soils were compared among natural HGM wetland types, contrasting reference standard wetlands to disturbed wetlands of that type. These latter sites were disturbed to some extent, from stressors occurring on site, within the buffer, and/or from the surrounding landscape. As expected the soils affected by glaciation in the northeastern and northwestern corners of Pennsylvania were wetter than soils from other ecoregions, and the number and density of wetlands was higher. Soil texture was coarser where hydraulic energies were greater, such as in riverine wetlands. Disturbed wetlands tended to have finer textured soils composed of more silt and clay, suggesting that they receive inputs of eroded sediments from the surrounding landscapes. Organic matter was higher in some HGM types, such as fringing and riparian depressions where soils are more saturated or inundated. Soils data from Riparia's set of reference wetlands is available through a searchable web interface at their website. A review of the literature comparing reference wetlands to mitigation projects continues to indicate the latter are not reaching the functional performance of natural wetlands.

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5.1 What Are Hydric Soils?

Hydric soils, formed when exposed to sufficient water levels, naturally occur in wetlands. In the United States, the presence of hydric soils (Fig. 5.1) is one of the three requirements for defining and delineating a jurisdictional wetland (Environmental Laboratory 1987). In this chapter, we compared the hydrologic, physical, and chemical characteristics of soils occurring in reference and disturbed wetlands. First, we review the literature and past studies conducted by Riparia at Penn State regarding the formation and attributes of hydric soils, including the poor performance of mitigation projects with regard to replicating the characteristics of soils typical of natural wetlands. Then, we use these available data to compare among the various types of wetlands occurring in the physiographic regions or ecoregions across Pennsylvania. Riparia's soils data was collected from sites distinguished with the hydrogeomorphic (HGM) classification system described by Cole et al. (1997) for Pennsylvania, so we use the original HGM subclass terms in this chapter. Where initially mentioned, we indicate the new, preferred terminology as defined by Brooks et al. (see Chap. 2) for the Mid-Atlantic Region (MAR).

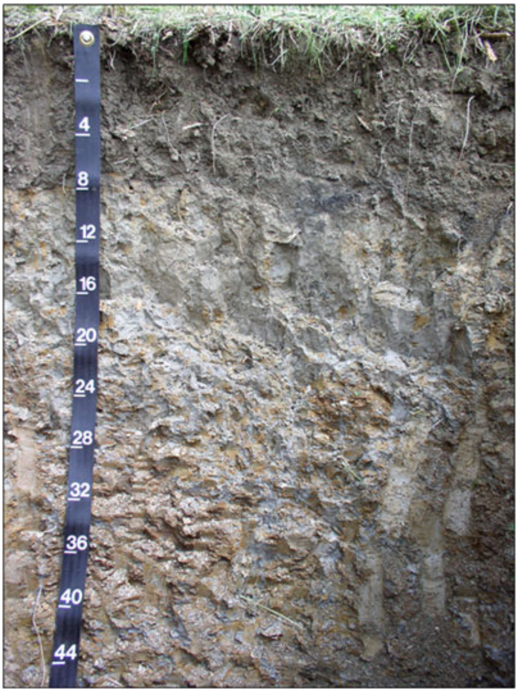

Fig. 5.1 Canadice soil profile (courtesy of Alex Dado, USDA-NRCS, Pennsylvania). Canadice (Fine, illitic, mesic Typic Endoaqualfs) is a hydric soil in Pennsylvania typically meeting two USDA-NRCS NASIS criteria: water table at less than or equal to 1.0 ft from the surface during the growing season if permeability is less than 6.0 in./h in any layer within 20 in.; or soils that are frequently ponded for long duration or very long duration during the growing season. Scale on tape in inches

5.1.1 What Distinguishes a Hydric Soil?

The USDA-NRCS defines a hydric soil as "a soil that formed under conditions of saturation, flooding or ponding long enough during the growing season to develop anaerobic conditions in the upper part" (USDA-NRCS 2010; Federal Register 1994). As with all soils, the genesis of hydric soils is dependent on the soil forming factors (Jenny 1941). The soil forming factors (together in various combinations and degree of influence) produce additions, losses, transformations, and translocations of organic and inorganic components, which then yield the soil (Simonson 1959). However, the morphology of a hydric soil is distinctly dependent on the presence of water for some length of time sufficient to induce anaerobic conditions that result in chemical reduction and thus mineralogical transformation (Vepraskas and Sprecher 1997). The electron transfer responsible for reduction is dependent on microorganisms, which indirectly provide the electron for reduction via their utilization of organic matter during bacterial respiration (Gobat et al. 2004). Once oxygen is driven from the system via saturation, microorganisms will then utilize organic matter via respiration, and thus, reduce O_2, NO_3^-, Fe and Mn oxides, sulfate, and carbon dioxide (Vepraskas and Faulkner 2001). If the soil remains saturated for a long enough period of time, reduction will result in the formation of redoximorphic features (Vepraskas and Sprecher 1997). Until 2003, it was generally thought that unless the soil temperature was above "biological 0," or 5°C (Vepraskas and Sprecher 1997), reduction was insignificant due to a lack of biological activity. However, in 2003, the US National Technical Committee on Hydric Soils voted (National Technical Committee for Hydric Soils 2003) to change the definition of hydric soils to "the soil temperature, at a depth of 50 cm (19.7″), below which the growth and function of locally adapted plants are negligible." For an excellent review on the topic of "Biological Zero" in the context of this recent change see Rabenhorst (2005).

It is important to note that soils may meet the criteria for a hydric soil, but not currently be in a landscape that could result in the genesis of such a soil (e.g., an aquic condition in Soil Taxonomy). In such cases, the soil would still meet the requirements for a hydric soil. In these cases it is also unlikely that a wetland would be present because of the potential absence of the two other jurisdictional requirements: hydrological conditions supporting a wetland and appropriate wetland flora.

5.1.2 Reduction–Oxidation Processes

Documentation of reduction in a soil can be determined via the measure of a soil's redox potential (a measure of electron availability that is expressed in millivolts) (DeLaune and Reddy 2005), the use of Indicator of Reduction in Soils (IRIS) tubes (Castenson and Rabenhorst 2006; Jenkinson and Franzmeier 2005), or via the use of dyes (Childs 1981) such as alpha, alpha-dipyridyl (USDA-NRCS 2010). A measure of redox potential is dependent upon determining the voltage that exists between a platinum wire in the presence of a soil solution and a reference electrode also in contact with the soil solution. The voltage measured indicates the chemical species that is present (Ponnamperuma 1972). As the redox potential decreases (becomes more negative) the terminal electron acceptor changes in the sequence of nitrate (NO_3^-), manganic manganese (Mn^{4+}), ferric iron (Fe^{3+}), sulfate (SO_4^{2}), and CO_2 (Mitsch and Gosselink 2000). For further information on the theory behind soil redox potential see Vepraskas and Faulkner (2001) and on making field redox potential measurements see Vasilas and Vasilas (2004). IRIS tubes are polyvinyl-chloride (PVC) tubes used to assess soil reduction (Jenkinson and Franzmeier 2005), which are coated with mostly ferrihydrite paint (Rabenhorst and Burch 2006; Rabenhorst et al. 2008). When installed in a soil that has some degree of saturation (part of the soil or all), the ferrihydrite paint will be depleted via reduction if oxygen is depleted, and the percent of paint removal can be equated to the soil profile being in a reduced condition or not (Castenson and Rabenhorst 2006). While the use of IRIS tubes is still new, the technique shows great promise. Alpha, alpha-dipyridyl dye will confirm the presence of ferrous iron (thus an indicator of reducing conditions) and possibly aquic conditions via development of pink color upon reaction with the soil (National Technical Committee for Hydric Soils NTCHS 2010).

5.1.3 Field Identification of a Hydric Soil

When determining the presence of a hydric soil it is essential to examine the site itself and how the landscape interacts with the soil in question. One should contrast conditions in dry and wet areas to be sure that one understands how water moves across the site. Ultimately, the identification of a hydric soil is dependent upon observable field indicators that occur in soil horizons. Hydric soil indicators are derived in large from "the accumulation or loss of iron, manganese, sulfur, or carbon compounds in a saturated and anaerobic environment (USDA-NRCS 2010)." It is recommended (USDA-NRCS 2010) that a soil be excavated to 50 cm for adequate examination of potential hydric soil indicators. Once excavated, the field personnel will attempt to identify an indicator for one of three potential groups (all soils; sandy soils; and loamy and clayey soils) and within their respective major land resource area (MLRA). The identification of a hydric soil indicator requires field experience and ideally collaboration with an experienced soil scientist. For the MAR, the Mid-Atlantic Hydric Soils Committee has produced an outstanding guide to assist with the identification of hydric soils in the region through their production of a region-specific, hydric soil guide (Mid-Atlantic Hydric Soils Committee 2004). Personnel working in wetland science in the MAR are strongly encouraged to study this guide closely before beginning field investigations.

5.2 Creating Hydric Soils During Wetland Mitigation

The construction of a mitigated wetland is usually meant to offset a loss of some existing wetland area. However, duplicating the function of a natural wetland is dependent on reproducing a complex array of processes, which in the case of a hydric soil, is dependent largely upon the presence of water (Davis 1995). In order to compare the success of a mitigation wetland project, the mitigated wetland will often be compared to a natural wetland in an ecosystem deemed to be similar in HGM form, function, and the ecosystem services the wetland should provide (D'Avanzo 1989; Davis 1995; ITRC 2005). Ultimately, how the mitigated wetland is created is dependent on location (providing the right hydrology Cole and Brooks 2000a, b) and physical makeup (Bishel-Machung et al. 1996, Stolt et al. 2000; Daniels and Whittecar 2004; Campbell et al. 2002).

Choosing an appropriate site for a wetland mitigation project involves first assessing what soils in the state, where the project is to be created, and meet the USDA-NRCS hydric soils criteria (Daniels and Whittecar 2004). Such soils will likely already occur in a landscape position where adequate water is already present to produce the anaerobic conditions necessary for reduction leading to the genesis of a hydric soil. Finding such soils can be aided via the use of the Web Soil Survey. Once a candidate soil and associated site is found, a site-specific soil survey is created to map soils in the area that meet the hydric soil criteria vs. those that do not. Site-specific soil surveys to determine hydric soil status will typically begin in the wettest areas, where hydric soils are known to occur, and then work outwards towards the areas of the landscape where hydric soils are less likely to occur (USDA-NRCS 2010). Use of the USDA-NRCS field indicators to properly identify the hydric soil is essential in this step.

5.2.1 Choosing the Proper Site for a Mitigated Wetland

Since 1993, Riparia has assessed wetlands throughout Pennsylvania (Brooks et al. 1996; Brooks 2004; Peterson-Smith et al. 2009), in order to understand landscape relationships between reference wetlands, mitigated (constructed) wetlands, and their HGM origins. At each of the reference wetlands, a suite of parameters has been monitored (Brooks 2004; Brooks et al. 2006) that has been found to be more or less strongly related to the quantity of water, the wetland type (Babb et al. 1997), and/or disturbance (Brooks et al. 1995) (http://www.riparia.psu.edu). This research has resulted in new techniques for rapid assessment of wetlands (Brooks et al. 2004), new insights into the use of indicators (Goslee et al. 1997; Niemi et al. 2004; Wardrop et al. 2005; Brooks et al. 2005, 2006; Miller et al. 2006, Miller and Wardrop 2006; Brooks et al. 2007; Hychka et al. 2007; Wardrop et al. 2007a, b, c; Rheinhardt et al. 2007; Brooks et al. 2009), a greater understanding of how mitigated wetlands can be more effectively created to act like natural wetlands (Brooks et al. 2005, 2006; Miller et al. 2006; Miller and Wardrop 2006; Brooks et al. 2007; Hychka et al. 2007;

Wardrop et al. 2007a, b, c; Rheinhardt et al. 2007; Brooks et al. 2009), and why they may not function properly. For example, early research by Babb et al. (1997) found little relationship with water quality and HGM class, and concluded that local lithologies were likely a more significant driver of water quality. In other cases, external factors, such as precipitation, can overwhelm a mitigated wetland as was seen with an acidic mine drainage treatment wetland experiencing heavy precipitation (Stark et al. 1994).

For effective mitigation that results in hydric soils, reference wetlands across a range of HGM classes must be examined in order to understand how soil and hydrology coevolve (Campbell et al. 2002) and how function varies by HGM class (whether natural or mitigated) (Cole and Brooks 2000a, b).

Because water is the dominant component defining a wetland from other ecosystems, the characterization of a wetland's hydrology is essential for determining the wetland's function (Cole et al. 1997; Stolt et al. 2000). Given a wetland's HGM class is tied to the landscape, certain wetland types are more dependent on local hydrological response, and do so in a predictable and consistent way. Such is the case in central Pennsylvania with slope and riparian wetland areas (Cole et al. 1997). This is an important realization given that a dataset of reference wetlands with predictable and consistent hydrological response can naturally lend itself to the development of a matrix of performance criteria (Brooks et al. 1996). Such a dataset can be referred to in mitigation situations, or for the identification of watersheds adversely affected by human disturbance (Myers et al. 2006). It is also important to recognize that such a dataset provides invaluable information on local parent materials that result in difficult to detect hydric soils (USDA-NRCS 2010).

Cole and Brooks (2000a, b) concluded that while created wetlands can meet jurisdictional requirements, their hydrologic behavior is not the same as that which is defined by an HGM subclass; mitigation plans must take this into account for full wetland functionality. Supporting this statement were results from Cole et al. (2002), who concluded that for specific HGM subclasses and settings in central Pennsylvania, wetlands classified similarly and, which depend on surface water additions, are more likely to have different wetland functions than wetlands that are hydrologically supported by regional water tables. Such differences can lead to differences in the water depth and/or duration of soil saturation, and thus change a wetland's hydrologic dependence from that of one dominated by anaerobic soil conditions to one reflective of aerobic conditions. This would certainly have an effect on the formation of hydric soil indicators. Note that in the case of a mitigated wetland, soils in some cases may not be tied to local parent materials due to typical prescriptions used in mitigation efforts; trucked in raw sands, silts, or compost may be used instead for mitigation. Lastly, like Reinhardt et al. (1997), Cole et al. (2002) concludes that HGM models are best used where developed due to regional variability in many processes; however at what scale such a recommendation applies to is unknown; we suspect that the scale is soil dependent. This is supported by results from Cole et al. (2002) who found that similar landforms between the United States and Europe, and within the United States (Cole et al. 2008; Peterson-Smith et al. 2009), did not result in similar conclusions about wetland function and process.

5.2.2 *Differences in Soils Between Natural and Mitigated Wetlands*

Following mitigation, wetlands will respond at different rates when developing the characteristics associated with hydric soil morphology: organic matter accumulation, reduction, and redoximorphic feature materialization. How fast this occurs varies, but in the case of redoximorphic features, has been found to be as quick as 5 years (Vepraskas et al. 1999), where saturation occurs longer, organic matter decomposition slows (Day and Megonigal 1993), and redoximorphic feature generation may be slower. Regardless, a common result of a mitigated wetland is a difference in soil organic matter from that of a natural wetland. Given the importance of soil organic matter in developing redoximorphic features, it, along with hydrology, would seem to serve as an important bottleneck in hydric soil indicator development. Bishel-Machung et al. (1996) and Campbell et al. (2002), working in central Pennsylvania, found that soil organic matter was lower in mitigated than natural wetlands; similar results were found by Cummings (1999) in Virginia. Stauffer and Brooks (1997) showed that by adding organic matter to created wetland soils, vegetation development in the wetland was assisted. A similar conclusion was reached by Sutton-Grier et al. (2009) in North Carolina and by Bailey et al. (2007) and Bruland et al. (2009) in Virginia (note that in Virginia, soil organic matter thresholds are required to be met for mitigated wetlands [Daniels et al. 2005]). When additions of soil organic matter were not added, levels in created wetlands were less than in reference wetlands (Stauffer and Brooks 1997).

Cole et al. (2001) examined relationships between above and belowground biomass on wetlands monitored earlier by Bishel-Machung et al. (1996) and Campbell (1996). Cole et al. (2001) concluded that natural vs. mitigated wetlands had differing soil organic matter contents (2–6% in created wetlands vs. 12–21% in reference wetlands) potentially resulting from export due to flooding, culvert transport, or soil property dynamics and variations in litter quality. We speculate that differences may also be due to interspecific competition between wetland plant species (Mahaney et al. 2004a), or anthropogenic stressors on wetland species (Mahaney et al. 2004b; Walls et al. 2005; Miller et al. 2006, Miller and Wardrop 2006), which select for species better adapted to stress. Such competition or stress-dependent selection may result in species that produce more or less lignin, and which are more or less resistant to degradation.

Soil organic matter content is just one parameter that has been found to differ between natural and mitigated wetlands. Campbell et al. (2002) found that mitigated wetlands had greater bulk densities, a higher soil chroma, and more rock fragments than natural wetlands. Soils in natural wetlands had more silts while mitigated wetlands had more sand. When wetland age was examined, significant differences in soils and vegetation characteristics were found between younger and older created wetlands; however, it could not be concluded older mitigation sites were becoming more similar to natural wetlands. These trends are similar to those recently reported by Gebo (2009) and Gebo and Brooks (2012). They compared

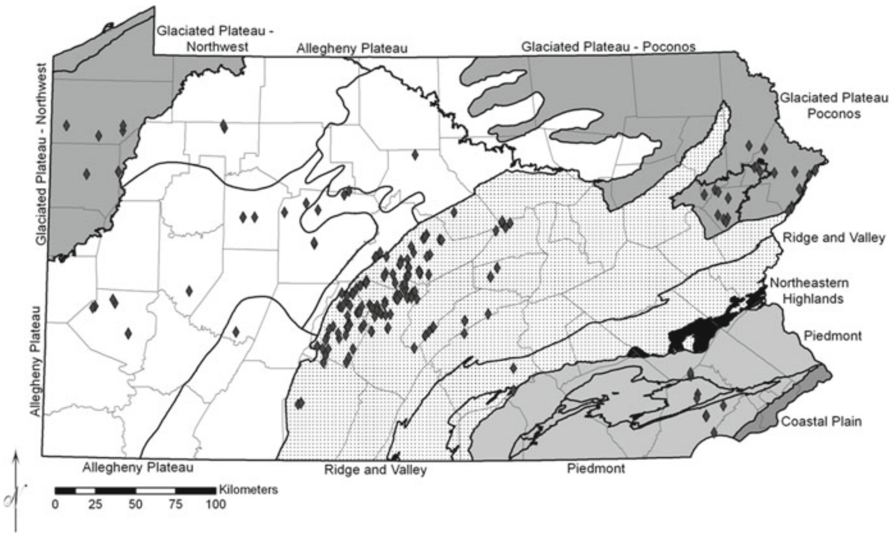

Fig. 5.2 Distribution of Riparia's Reference wetlands across Pennsylvania, USA

data from the same set of reference wetlands to a large sample of 72 mitigation sites from across Pennsylvania, and found similar trends indicating that mitigation sites were not performing as well as natural reference wetlands of the same HGM type. Their analyses included soils data, as well as variables related to various ecological functions (see Chap. 12 for more details).

5.3 How Do Reference Standard and Disturbed Pennsylvania Wetlands Differ?

In order to better understand how disturbed wetlands differ from natural wetlands across Pennsylvania, we examined undisturbed "Reference Standard (Ref. Std.)" wetlands vs. others that had varying degrees of disturbance ("Disturbed"). Field and laboratory data collected from 220 study sites (Fig. 5.2) sampled once during the period 1993–2003 were used in this analysis. Study sites across Pennsylvania occur in the following physiographic provinces: Ridge and Valley Province (143 sites), Piedmont (9), Allegheny Plateau (38 sites), Glaciated Poconos (24 sites), and Glaciated Plateau (8 sites). Study sites represent seven HGM types (see key in Chap. 2): lacustrine (fringing) (17 sites), riverine upper perennial (headwater floodplain) (67 sites), riverine impoundment beaver (11 sites), depression, seasonal/temporary (isolated) (17 sites), and riverine lower perennial (mainstem floodplain) (42 sites) (Table 5.1). Data on soil physical properties includes particle size determination via the hydrometer method (Gee and Bauder 1986) and field estimates of texture both at a 20 cm depth. For sites where laboratory particle size data were missing, but field

Table 5.1 Mean soil physical differences for each physiographic region (ecoregion) by disturbance status

Physiographic province	Sand	Silt (%)	Clay	*n*	Sand combo	Silt combo (%)	Clay combo	*n*	OM (5 cm)	*n* (%)	OM (20 cm)	*N*
Ridge and Valley												
Disturbed	25 (13)	48 (11)	26 (9)	17	38 (20)	29 (14)	32 (15)	113	10.9 (6.9)	19	6.5 (2.6)	18
Ref. Std.	47 (15)	31 (10)	20 (8)	21	42 (17)	28 (10)	27 (15)	26	14.6 (11.3)	21	6.4 (4.1)	18
p-value	0.001	0.001	0.103		0.023	0.471	0.040		0.752		0.505	
Piedmont												
Disturbed	32 (16)	46 (6)	20 (9)	2	53 (22)	25 (16)	17 (9)	7	7.5 (2.4)	2	6.1 (2.9)	2
Glaciated Poconos												
Disturbed	57 (15)	40 (25)	14 (12)	7	41 (17)	27 (13)	28 (16)	9	16.1 (18.2)	7	9.4 (12.3)	7
Ref. Std.	77 (21)	16 (18)	6 (2)	2	62 (*)	27 (*)	10 (*)	1	3.0 (2.2)	2	2.7 (2.3)	2
p-value	0.858	0.151	0.858		0.115	*	*		0.151		0.858	
Glaciated Plateau												
Disturbed	*	*	*	0	11 (4)	43 (16)	48 (11)	5	*	0	*	0
Ref. Std.	*	*	*	0	20 (*)	20 (*)	60 (*)	1	*	0	*	0
p-value	*	*	*		*	*	*		*		*	
Allegheny Plateau												
Disturbed	26 (13)	38 (8)	32 (13)	4	29 (16)	34 (11)	35 (16)	29	8.6 (2)	4	6.2 (1.3)	4
Ref. Std.	*	*	*	0	35 (3)	35 (3)	28 (5)	2	*	0	*	0
p-value	*	*	*		0.131	0.735	0.278		*		*	

p-value indicates significant differences between Disturbed and Ref. Std. wetlands ($\alpha = 0.05$, mood median test). Standard deviation in parenthesis; n = sample size

texture data available, the mid-point of the field texture's percent sand, silt, and clay was used to derive the sand, silt, and clay combination value. Chemical data include the percent organic matter at 5 and 20 cm (Schulte 1995), acidity (Eckert and Thomas Sims 1995), CEC (Ross 1995), exchangeable Ca, Mg, and K (Wolf and Beegle 1995), soil pH (Eckert and Thomas Sims 1995) and P (at 5–20 cm) (Wolf and Beegle 1995), Kjeldahl N at 5 and 20 cm (Isaac and Johnson 1976), nitrate N at 5 and 20 cm (Griffin 1995), and Total N at 5 and 20 cm (Bremner 1996). Soil morphologic data include the soil matrix Munsell value and chroma at a depth of 5 and 20 cm, the presence (1 = absent, 2 = present) of redoximorphic features at 5 and 20 cm, and the degree of saturation (dry = 1, moist = 2, saturated = 3, inundated = 4) at 5 and 20 cm. Within any one site, the median value from the study transect was determined, and used as the representative site value. Across all wetlands in the state, within physiographic regions, and HGM classes, data were analyzed by disturbance status (*Ref. Std.* or *Disturbed*) using a nonparametric Mood Median test or a Tukey Multiple Comparisons Test (Minitab, Inc. 2000; Orlich 2000). An $\alpha = 0.05$ was used to test for significant differences.

5.3.1 Soil Hydrologic Characteristics of Ref. Std. Versus Disturbed Pennsylvania Wetlands

Soil physical properties significantly differed in Ref. Std. vs. Disturbed wetlands across Pennsylvania; however, no differences in the indicators of a wetland's hydrology were found. Disturbed wetlands across the state were found to have significantly lower sand and higher silt and clay (Fig. 5.3a, b). No difference was seen with the percent organic matter (Fig. 5.3c). Regardless of disturbance, redoximorphic features tend to be absent (Fig. 5.4a), but soils tend to be moist to saturated (Fig. 5.4a) with horizons that exhibit high soil color values and low chromas (Fig. 5.4b) indicative of a hydric soil with a gleyed condition (Bg horizonation).

Across physiographic provinces of the state (Table 5.1) the Glaciated Plateau and Glaciated Poconos tended to have the wettest soils with moist to saturated conditions and the lowest chroma, but the Ridge and Valley and Allegheny Plateau had wetland soils with the highest soil color value. Higher soil color values in the Ridge and Valley and Allegheny Plateau may be due to the predominance of sandstone, and thus, quartz, in soils where these wetlands are found. High quartz could impart a higher value color in a reduced state due to the loss of iron during reduction.

Across physiographic regions of the state (Table 5.1), only the Ridge and Valley Province, Glaciated Poconos, and Allegheny Plateau could be examined for differences between Disturbed and Ref. Std. wetlands. Within the Ridge and Valley, Ref. Std. wetlands have significantly higher sand and lower silt and clay, but no difference in organic matter (Table 5.1). As with the statewide data, no differences in hydrologic indicators are seen, and soils tended to exhibit high value and low chroma colors indicative of hydric soils; redoximorphic features tended to be absent, suggesting horizons exhibit gley conditions. Soils tended to be moist to saturated (Table 5.2).

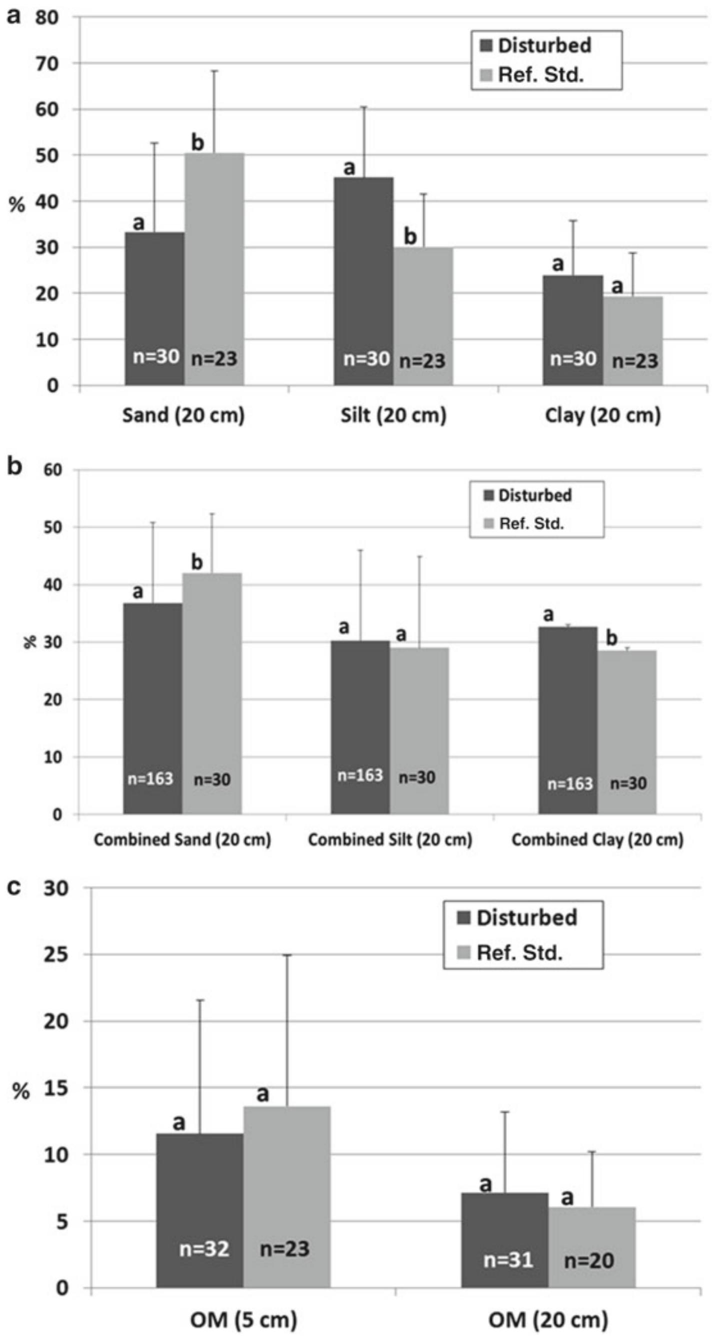

Fig. 5.3 Mean physical property differences across all Pennsylvania wetlands and hydrogeomorphic (HGM) classes, by disturbance status. Different letters above bars of a single property indicate significant differences ($\alpha=0.05$, mood median test). Sample size in bars

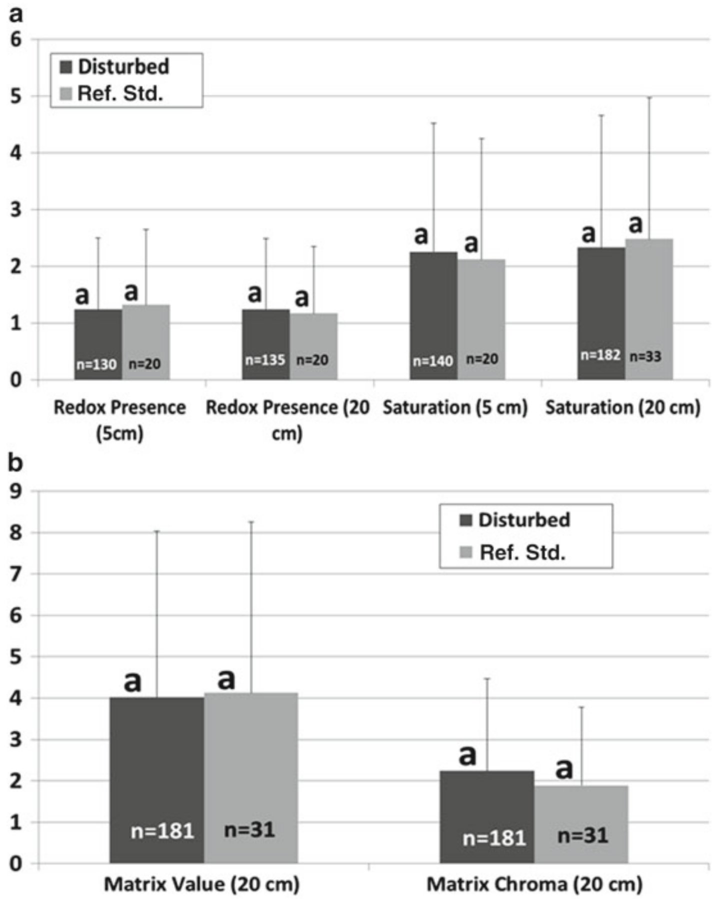

Fig. 5.4 Mean soil morphology differences across all Pennsylvania wetlands and hydrogeomorphic classes, by disturbance status. Different letters above bars of a single property indicate significant differences ($\alpha = 0.05$, mood median test). Sample size in bars

When HGM types are examined across the state, a trend was seen in Ref. Std. wetlands with higher energy environments (mainstem floodplains, headwater floodplain, and slope) having higher sand contents than lower energy environments (all other wetlands) (Table 5.3). The percent silt and organic matter was frequently found to be higher in the lower energy environments (fringing and isolated depression for example). No strong trends were suggestive of differences in hydrology and resulting effects on soil morphology across the state's HGM types. For example, redoximorphic features tended to be few and the matrix value is often indicative of a high value hydric soil condition, while wetter environments (fringing and depressions types) tended to have the highest soil color values and saturation condition (wet to inundated) (Table 5.4).

Table 5.2 Mean soil morphological differences for each physiographic region by disturbance status

Physiographic province	Matrix val (5 cm)	*n*	Matrix val (20 cm)	*n*	Matrix chr (5 cm)	*n*	Matrix chro (20 cm)	*n*	Redox (5 cm)	*n*	Redox (20 cm)	*n*	Satuatio*n* (5 cm)	*n*	Saturation (20 cm)	*n*
Ridge and Valley																
Disturbed	3.6 (0.6)	94	4.1 (0.9)	116	1.9 (0.8)	94	2.3 (1.2)	116	1.1 (0.3)	93	1.2 (0.3)	95	2.2 (0.6)	96	2.3 (0.7)	116
Ref. Std.	3.6 (1.1)	14	4.3 (1.1)	24	1.9 (1.2)	14	1.8 (1.1)	24	1.2 (0.4)	14	1.1 (0.2)	14	1.1 (0.8)	14	23 (0.9)	25
p-value	0.671		0.356		0.839		0.054		0.309		0.503		0.599		0.386	
Piedmont																
Disturbed	3.8 (1.1)	6	4.2 (0.7)	8	2.5 (1.0)	6	1.8 (0.8)	8	1.7 (0.4)	5	1.3 (0.5)	6	1.6 (0.5)	6	2.0 (0.5)	8
Glaciated Poconos																
Disturbed	2.5 (0.6)	10	3.2 (1)	17	1.7 (0.6)	10	1.3 (0.4)	17	1 (0)	8	1 (0)	10	2.9 (0.7)	10	2.7 (0.8)	18
Ref. Std.	3.0 (0)	2	3 (0)	2	2.5 (0.7)	2	2.2 (0.3)	2	1 (0)	2	1 (0)	2	2.2 (0.3)	2	3.0 (1.0)	3
p-value	0.121		0.253		0.19		0.115		*		*		0.121		0.476	
Glaciated Plateau																
Disturbed	3.0 (0.6)	6	2.8 (0.7)	6	1.5 (0.5)	6	1.1 (0.4)	5	1.8 (0.4)	6	1.8 (0.4)	6	2.8 (0.4)	6	2.8 (0.4)	6
Ref. Std.	3.0 (1.4)	2	3.0 (1.4)	2	1 (0)	2	1 (0)	2	2.0 (0)	2	2.0 (0)	2	3.5 (0.7)	2	3.5 (0.7)	2
p-value	0.346		0.346		0.206		0.206		*		*		*		*	
Allegheny Plateau																
Disturbed	3.6 (0.8)	23	4.2 (0.8)	34	2.2 (1.1)	23	2.5 (1.4)	34	1.5 (0.4)	18	1.4 (0.4)	18	1.9 (0.9)	22	2.0 (0.8)	34
Ref. Std.	4.0 (0)	2	3.6 (0.5)	3	3.0 (0)	2	2.5 (0.5)	3	1.5 (0.7)	2	1.0 (0)	2	1.5 (0.7)	2	2.3 (1.5)	3
p-value	0.157		0.136		0.049		0.393		1		0.224		0.803		0.887	

p-value indicates significant differences between Disturbed and Ref. Std. wetlands ($\alpha = 0.05$, mood median test). Standard deviation in parenthesis; $n =$ sample size

Table 5.3 Mean soil physical differences for each hydrogeomorphic class by disturbance status

Hydrogeomorphic class	Sand	Silt	Clay	*n*	Sand combo	Silt combo	Clay combo	*n*	OM (5 cm)	*n*	OM (20 cm)	*n*
	%				%				%			
Fringing												
Disturbed	20 (5)	19 (11)	32 (8)	3	31 (20)	35 (17)	35 (15)	11	11 (2)		7 (1)	3
Ref. Std.	15 (8)	39 (4)	16 (7)	4	41 (15)	35 (10)	25 (24)	4	17 (15)		10 (7)	4
p-value	0.047	0.270	0.008		0.095	0.876	0.310		0.659		0.414	
Headwater floodplain												
Disturbed	30 (20)	49 (16)	22 (7)	5	38 (19)	30 (14)	32 (13)	11	8.8 (2.9)		6.4 (2.8)	6
Ref. Std.	47 (19)	28 (12)	25 (8)	4	56 (22)	25 (16)	21 (10)	4	7.7 (6.0)		3.6 (2.8)	3
p-value	0.099	0.016	0.764		0.152	0.317	0.049		0.197		0.635	
Headwater impoundment beaver												
Disturbed	37 (6)	39 (2)	21 (2)	2	32 (17)	30 (11)	38 (11)	12	37.8 (17.0)	3	25.1 (15.1)	2
Ref. Std.	*	*	*	*	*	*	*	*	*	*	*	*
p-value	*	*	*		*	*	*		*		*	
Isolated depression												
Disturbed	16 (5)	47 (4)	37 (9)	2	33 (17)	31 (16)	35 (10)	8	7.9 (1.0)	2	5.6 (0.3)	2
Ref. Std.	64	22	11	1	36 (21)	27 (8)	34 (18)	6	39.8 (*)	1	5.6 (*)	1
p-value	*	*	*		1	0.04	0.198		*		*	

Mainstem floodplain												
Disturbed	33 (14)	45 (7)	20 (9)	5	46 (21)	30 (13)	24 (13)	30	6.6 (1.5)	5	4.9 (1.4)	5
Ref. Std.	76 (15)	16 (13)	8 (4)	3	45 (22)	27 (9)	28 (20)	5	3.4 (1.7)	3	3.0 (1.7)	3
p-value	0.028	0.028	0.028		0.679	1	0.679		0.028	0	0.465	
Riparian depression												
Disturbed	57 (22)	34 (17)	8 (5)	5	33 (23)	26 (15)	40 (18)	18	10.4 (5.5)	5	3.8 (1.9)	5
Ref. Std.	39 (8)	39 (6)	23 (6)	4	39 (8)	39 (6)	23 (6)	4	21.9 (10.6)	4	7.8 (3.2)	4
p-value	0.016	0.099	0.003		0.269	0.027	0.027		0.099		0.099	
Slope												
Disturbed	29 (18)	50 (21)	32 (10)	8	32 (22)	33 (14)	36 (18)	27	8.8 (1.7)	8	7.0 (3.1)	8
Ref. Std.	49 (19)	28 (11)	22 (11)	7	39 (13)	25 (9)	34 (14)	7	12.6 (10.7)	7	5.5 (2.9)	6
p-value	0.072	0.019	0.189		0.034	0.671	0.803		0.782		0.28	

p-value indicates significant differences between Disturbed and Ref. Std. wetlands ($\alpha = 0.05$, mood median test). Standard deviation in parenthesis; n = sample size

Table 5.4 Mean soil physical differences for each hydrogeomorphic class by disturbance status

Hydrogeomorphic class	Matrix val (5 cm)	*n*	Matrix val (20 cm)	*n*	Matrix chr (5 cm)	*n*	Matrix chro (20 cm)	*n*	Redox (5 cm)	*n*	Redox (20 cm)	*n*	Saturation (5 cm)	*n*	Saturation (20 cm)	*n*
Fringing																
Disturbed	4.1 (1)	8	4.2 (0.7)	11	2.1 (0.8)	8	23 (1.4)	11	1.5 (0.5)	8	1.3 (0.4)	8	2.8 (0.7)	8	2.6 (0.7)	11
Ref. Std.	5.0 (*)	1	4.3 (1.3)	4	1.0 (*)	1	1.8 (0.5)	4	1.0 (*)	1	1.0 (*)	1	4.0 (*)	1	3.5 (0.8)	6
p-value	*		0.770		*		0.930		*		*		*		0.013	
Headwater floodplain																
Disturbed	3.7 (0.7)	54	4.1 (0.8)	54	2.1 (0.7)	62	2.5 (1.2)	62	1.2 (0.4)	51	1.2 (0.4)	51	2.1 (0.6)	54	2.1 (0.6)	62
Ref. Std.	4.0 (1.0)	3	4.3 (0.6)	3	2 (0)	3	2.0 (1.0)	3	1.7 (0.6)	3	1.0 (0)	3	1.7 (1.2)	3	1.3 (0.6)	3
p-value	0.847		0.922		0.288		0.727		0.04		0.436		0.46		0.423	
Headwater impoundment beaver																
Disturbed	3.1 (0.7)	8	3.9 (1.6)	11	2.3 (1.4)	8	1.9 (1.1)	11	1.1 (0.4)	7	1 (*)	7	2.6 (0.9)	7	3.2 (0.8)	12
Ref. Std.	*		*		*		*		*		*		*		*	
p-value	*		*		*		*		*		*		*		*	
Isolated depression																
Disturbed	3.1 (0.4)	7	4.1 (1.2)	11	1.3 (0.5)	7	1.59091	11	1.6 (0.5)	7	1.4 (0.5)	7	2.8 (07)	7	2.7 (1.1)	11
Ref. Std.	2.7 (1.3)	9	3.8 (1.4)	6	2.2 (2.2)	5	2.33333	6	1.0 (*)	5	1.0 (*)	5	2.2 (0.4)	5	3.0 (0.9)	6
p-value	0.004		0.627		0.679		0.858		0.038		0.091		0.079		0.901	

Mainstem floodplain																
Disturbed	3.4 (0.7)	23	3.6 (0.7)	34	2.3 (1.2)	23	2.5 (1.4)	34	1.2 (0.4)	23	1.3 (0.4)	23	2.0 (0.8)	23	2.1 (0.7)	34
Ref. Std.	3.3 (0.8)	6	3.3 (0.8)	7	2.2 (1.0)	6	2.0 (0.8)	7	1.5 (0.5)	6	1.3 (0.5)	6	2.4 (1.0)	6	2.5 (1.0)	7
p-value	0.775		0.839		0.494		0.819		0.096		0.842		0.132		0.112	
Riparian depression																
Disturbed	3.4 (1.1)	17	4.0 (1.0)	21	1.9 (0.7)	17	1.8 (0.8)	21	1.2 (0.4)	14	1.2 (0.4)	15	2.7 (0.7)	17	2.6 (0.8)	21
Ref. Std.	*	0	5.3 (1.0)	4	*	0	1.5 (0.6)	4	*	0	*	0	*	0	2	4
p-value	*		0.119		*		0.93		*		*		*		0.075	
Slope																
Disturbed	3.2 (0.8)	22	4.2 (1.1)	31	1.8 (0.8)	22	2.0 (1.4)	31	1.4 (0.5)	20	1.5 (0.5)	21	2.3 (07)	22	2.3 (0.7)	31
Ref. Std.	4.0 (0.7)	5	4.5 (1.2)	7	1.8 (0.4)	5	1.6 (0.7)	7	1.3 (0.4)	5	1.3 (0.4)	5	1.6 (0.5)	5	1.9 (0.7)	7
p-value	0.076		0.791		0.296		0.566		0.84		0.759		0.143		0.85	

p-value indicates significant differences between Disturbed and Ref. Std. wetlands ($\alpha = 0.05$, mood median test). Standard deviation in parenthesis; n = sample size

Several significant differences were seen within HGM classes between Disturbed and Ref. Std. wetlands. For example, Ref. Std. wetlands were frequently characterized as having significantly higher sand than reference wetlands (Table 5.3), especially in higher energy environments (floodplain and slope classes). Lower sand content in Disturbed wetlands was likely due to erosional inputs of silt and clay (higher in Disturbed wetlands) from the surrounding landscape. Interestingly, no significant difference in the percent organic matter is found between Disturbed and Ref. Std. wetlands in any HGM class. In some cases, Ref. Std. environments had higher organic matter (fringing, riparian depression), while in others it is lower (mainstem floodplain and slope).

5.3.2 Soil Chemical Characteristics of Ref. Std. Versus Disturbed Pennsylvania Wetlands

Soil chemical characteristics in wetlands can be an important indicator of differences due to local lithologies or land use influences. Across Pennsylvania, a combination of both factors is seen to affect wetland chemistry.

First, across all Pennsylvania wetlands, several significant differences in soil chemistry were seen between Ref. Std. and Disturbed wetlands. Disturbed wetlands had significantly lower acidity, higher pH, Ca, and Mg saturation (Fig. 5.5), but no difference in nitrogen or phosphorus (Fig. 5.6). This is perhaps suggestive of fresh inputs of minimally weathered products into these systems, which can provide ample bases and buffer acidification.

Closer examination of soil chemical differences across physiographic regions (Table 5.5) suggests a trend with the Ridge and Valley Province having the highest soil pH, Ca, and Mg saturation (most likely due to the regional limestone influence in valleys). No trend in CEC across regions was noted. Across all physiographic regions no significant differences or trends were seen in nitrogen or phosphorus (Table 5.6).

Similar to statewide results, within physiographic regions, Ref. Std. wetlands always had a lower pH, higher Acidity, and lower Ca, K, or Mg saturation, although differences were not always significant; this is most likely due to the small sample sizes with some regions. This result also parallels all HGM classes (Table 5.7). Within physiographic regions (Table 5.6), trends were complicated most likely by land use differences between regions. For example, in the Ridge and Valley, Disturbed wetlands had significantly higher Kjedhal N with trends suggestive of higher Nitrate N, but not Total nitrogen or phosphorus (Table 5.8). Agricultural runoff is a potential source of nitrogen in these wetland systems given the predominance of agricultural land use in the Ridge and Valley province. A similar trend is seen across the Allegheny Plateau where an agricultural land use is still common, but different from that of the Ridge and Valley (pasture vs. row crop agriculture).

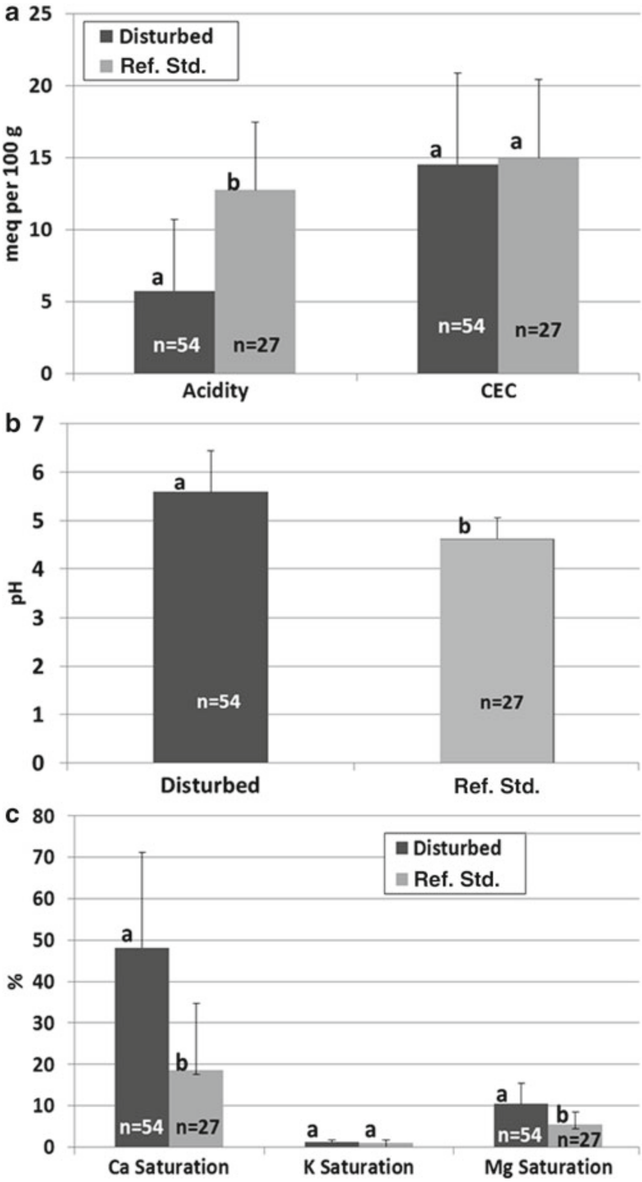

Fig. 5.5 Mean soil chemical differences across all Pennsylvania wetlands and hydrogeomorphic classes, by disturbance status. Different letters above bars of a single property indicate significant differences ($\alpha = 0.05$, mood median test). Sample size in bars

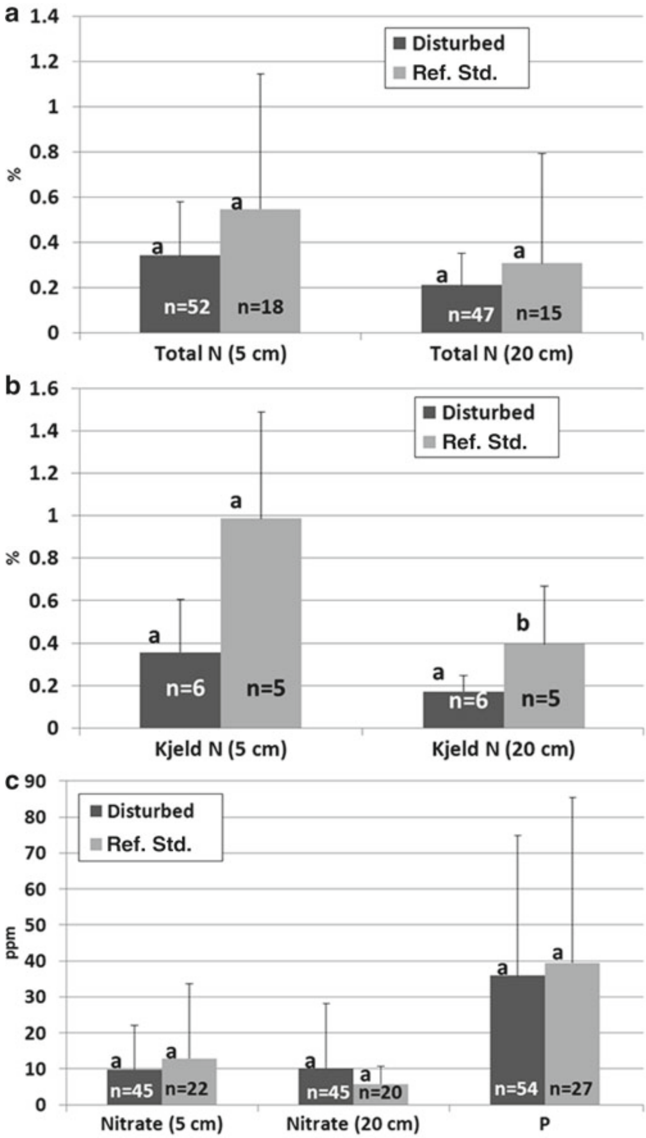

Fig. 5.6 Mean soil nitrogen and phosphorus differences across all Pennsylvania wetlands and hydrogeomorphic (HGM) classes, by disturbance status. Different letters above bars of a single property indicate significant differences ($\alpha = 0.05$, mood median test). Sample size in bars

5.4 Conclusions

Soils were compared among natural HGM wetland types, contrasting Ref. Std. wetlands (assumed to be representative of a specific HGM type and the least disturbed) to disturbed wetlands of that type. These were assumed to be disturbed to some

Table 5.5 Mean soil chemical differences for each physiographic region by disturbance status

Physiographic province	n	pH	Acidity	CEC	Ca saturation	K saturation	Mg saturation
			——meg 100 g^{-1}——		——————%——————		
Ridge and Valley							
Disturbed	21	5.8 (1.1)	5.54 (6.15)	15.08 (4.76)	52.21 (26.62)	1.32 (0.53)	10.62 (5.49)
Ref. Std.	22	4.6 (0.4)	13.47 (6.69)	14.11 (3.71)	15.61 (3.43)	1.08 (0.67)	5.14 (2.73)
p-value		0.001	0.001	0.876	0.001	0.094	0.001
Piedmont							
Disturbed	6	5.2 (0.5)	8.12 (5.24)	15.1 (3.26)	36.01 (13.83)	1.26 (0.56)	11.92 (4.68)
Glaciated Poconos							
Disturbed	1	5.5 (*)	2 (*)	5.2 (*)	52.8 (*)	0.9 (*)	8.0 (*)
Ref. Std.	1	4.1 (*)	13.9 (*)	33.2 (*)	45.1 (*)	0.4 (*)	12.6 (*)
Glaciated Plateau							
Disturbed	6	5.4 (0.5)	5.66 (2.11)	14.66 (2.39)	51.65 (11.8)	1.06 (0.3)	10.7 (33.1)
Ref. Std.	2	5.0 (0.1)	7.75 (1.76)	19.25 (7.0)	45.72 (2.91)	0.70 (0.56)	7.40 (0.56)
p-value		0.102	0.673	1.000	1.000	0.673	0.102
Allegheny Plateau							
Disturbed	20	5.5 (0.8)	5.40 (4.18)	14.06 (5.36)	46.24 (23.7)	1.06 (0.38)	9.89 (5.03)
Ref. Std.	2	4.6 (0.3)	9.25 (3.88)	10.95 (5.02)	10.62 (3.14)	0.85 (0.07)	2.95 (0.35)
p-value		0.138	0.138	1.000	0.138	0.138	0.138

p-value indicates significant differences between Disturbed and Ref. Std. wetlands ($\alpha = 0.05$, mood median test). Standard deviation in parenthesis; n = sample size

extent, from stressors occurring on site, within the buffer, and/or from the surrounding landscape. In addition, we reviewed the literature regarding how the characteristics of natural wetlands from the reference set are not being replicated in mitigation projects.

As expected the soils affected by glaciation in the northeastern and northwestern corners of Pennsylvania were wetter than soils from other ecoregions. The number and density of wetlands is much higher in these glaciated ecoregions. Soil texture was coarser where hydraulic energies were greater, such as in riverine wetlands. Disturbed wetlands tended to have finer textured soils composed of more silt and clay, which may indicate that they receive inputs of eroded sediments from the surrounding landscapes. Organic matter was not appreciably different between Ref. Std. and Disturbed sites, but was higher in some HGM types, such as fringing and riparian depressions where soils are more saturated or inundated.

Our overall findings indicate that soil characteristics in Ref. Std., disturbed, and mitigated sites are highly variable. Although there is considerable overlap in the ranges for many soil-related variables across HGM types, there are some significant differences and trends that can help guide practitioners towards improved designs for mitigation projects, resulting in improvements in their functional performance.

Table 5.6 Mean soil nitrogen and phosphorus differences for each physiographic region by disturbance status

Physiographic province	Kjeld N (5 cm)	Kjeld N (20 cm)	*n*	Nitrate N (5 cm)	*n*	Nitrate N (20 cm)	*n*	Total N (5 cm)	*n*	Total N (20 cm)	*n*	P (20 cm)	*n*
	——% ——			——ppm——				——%——				——kg——	
Ridge and Valley													
Disturbed	0.52 (0.26)	0.22 (0.07)	3	15.52 (17.53)	17	9.58 (9.58)	17	0.34 (0.26)	23	0.19 (0.08)	22	38.71 (58.99)	21
Ref. Std.	0.98 (0.50)	0.39 (0.27)	5	14.96 (23.33)	17	5.85 (4.69)	16	0.38 (0.28)	13	0.19 (0.15)	11	42.50 (50.48)	22
p-value	0.465	0.028		0.086		0.169		0.729		0.325		0.876	
Piedmont													
Disturbed	*	*	0	3.80 (2.97)	6	3.89 (2.84)	5	0.41 (0.25)	6	0.24 (0.12)	5	49.46 (16.09)	6
Glaciated Poconos													
Disturbed	*	*	0	*	0	*	0	0.02 (*)	1	*	0	13.44 (*)	1
Ref. Std.	*	*	0	4.2 (*)	1	*	0	2.29 (*)	1	*	0	40.32 (*)	1
Glaciated Plateau													
Disturbed	*	*	0	8.73 (3.38)	6	22.2 (37.4)	6	0.47 (0.34)	6	0.32 (0.3)	6	32.85 (24.19)	6
Ref. Std.	*	*	0	8.60 (0.56)	2	7.85 (9.54)	2	1.13 (1.23)	2	1.07 (1.3)	2	20.16 (6.33)	2
p-value	*	*		0.206		1.000		1.000		1.000		1.000	
Allegheny Plateau													
Disturbed	0.19 (0.03)	0.12 (0.02)	3	6.31 (6.86)	16	8.29 (17.19)	16	0.27 (0.08)	16	0.16 (0.10)	14	30.8 (15.23)	20
Ref. Std.	*	*	0	3.40 (2.68)	2	2.47 (1.09)	2	0.25 (0.09)	2	0.14 (0.03)	2	25.2 (0.78)	2
p-value	*	*		1.000		1.000		1.000		1.000		1.000	

p-value indicates significant differences between Disturbed and Ref. Std. wetlands ($\alpha = 0.05$, mood median test). Standard deviation in parenthesis; n = sample size

Table 5.7 Mean soil chemical differences for each hydrogeomorphic class by disturbance status

Hydrogeomorphic class	*n*	pH	Acidity	CEC	Ca saturation	K saturation	Mg saturation
			——meq 100 g^{-1}——		——————%——————		
Fringing							
Disturbed	5	6.1 (0.9)	2.34 (2.01)	12.49 (4.92)	60.93 (20.00)	1.13 (0.54)	12.25 (3.83)
Ref. Std.	4	4.8 (0.9)	11.78 (6.72)	20.73 (8.62)	37.73 (25.53)	0.55 (0.34)	6.85 (4.10)
p-value		0.294	0.099	0.343	0.764	0.343	0.294
Headwater floodplain							
Disturbed	15	5.4 (0.8)	7.26 (6.64)	13.93 (3.31)	42.86 (25.17)	1.25 (0.44)	8.86 (4.51)
Ref. Std.	3	4.5 (0.2)	11.45 (3.13)	10.85 (2.64)	11.12 (2.21)	0.82 (0.38)	4.58 (0.89)
p-value		0.058	0.058	0.527	0.058	0.671	0.058
Headwater impoundment beaver							
Disturbed	2	5.2 (0.2)	8.60 (5.06)	19.0 (15.0)	39.2 (16.2)	1.2 (1.0)	8.2 (2.9)
Ref. Std.	*	*	*	*	*	*	*
p-value		*	*	*	*	*	*
Isolated depression							
Disturbed	3	4.8 (0.3)	11.42 (2.47)	15.17 (0.35)	20.30 (11.01)	1.22 (0.64)	5.12 (4.18)
Ref. Std.	6	4.3 (0.3)	19.23 (6.43)	15.59 (3.29)	8.08 (3.16)	1.28 (0.41)	2.68 (0.52)
p-value		0.343	0.058	0.134	0.343	0.635	0.343
Mainstem floodplain							
Disturbed	11	6.2 (1)	3.07 (3.24)	15.94 (6.28)	61.85 (23.29)	1.10 (0.45)	11.57 (4.30)
Ref. Std.	5	4.8 (0.3)	8.10 (2.43)	13.56 (6.90)	25.53 (21.81)	0.94 (0.47)	5.36 (2.29)
p-value		0.007	0.007	0.59	0.106	0.838	0.007
Riparian depression							
Disturbed	0	*	*	*	*	*	*
Ref. Std.	3	4.9 (0.2)	13.42 (9.00)	13.77 (4.15)	15.27 (7.34)	1.60 (1.56)	7.70 (2.36)
p-value		*	*	*	*	*	*
Slope							
Disturbed	18	5.5 (0.6)	5.74 (3.72)	14.12 (3.63)	46.29 (18.24)	1.19 (0.46)	11.77 (5.43)
Ref. Std.	6	4.6 (0.3)	11.12 (4.77)	14.33 (3.05)	15.79 (3.02)	0.92 (0.43)	6.57 (3.64)
p-value		0.009	0.005	0.237	0.005	0.059	0.059

p-value indicates significant differences between Disturbed and Ref. Std. wetlands ($\alpha = 0.05$, mood median test). Standard deviation in parentheses; n = sample size

Table 5.8 Mean soil nitrogen and phosphorus differences for each hydrogeomorphic class by disturbance status

Hydrogeomorphic class	Kjeld N (5 cm)	*n*	Kjeld N (20 cm)	*n*	Nitrate N (5 cm)	*n*	Nitrate N (20 cm)	*n*	Total N (5 cm)	*n*	Total N (20 cm)	*n*	P (20 cm) kg ha^{-1}	*n*
	—— % ——				—— ppm ——				%		%			
Fringing														
Disturbed	0.17		0.12	1	3.76 (1.91)	4	3.10 (0.87)	4	0.42 (0.17)	3	0.17 (0.12)	3	31.92 (22.56)	5
Ref. Std.	0.38		0.33	1	4.69 (1.10)	4	10.76 (10.32)	2	1.14 (0.88)	3	0.15 (*)	1	30.45 (12.48)	4
p-value	*	*		0.157		1.000		0.414		*		0.764		
Headwater floodplain														
Disturbed	*	0	0.26	1	11.5 (21)	12	4.8 (3.2)	12	0.27 (0.14)	17	0.19 (0.14)	16	29.12 (10.75)	15
Ref. Std.	*	0	*	0	2.8 (0.4)	3	2.3 (1.1)	2	0.20 (0.10)	3	0.17 (0.10)	3	173 (1.27)	3
p-value	*		*		0.070		0.127		0.531		0.596		0.090	
Headwater impoundment beaver														
Disturbed	*		*	*	5.1	1	10.4	1	0.69 (0.58)	3	0.21 (0.07)	2	11.53 (9.07)	2
Ref. Std.	*		*	*	*	*	*	*	*	*	*	*	*	*
p-value	*		*		*		*		*		*		*	
Isolated depression														
Disturbed	0.52 (0.27)	3	0.21 (0.09)	2	14.77 (7.38)	3	6.78 (2.35)	3	*	0	*	0	148.06 (115.14)	3
Ref. Std.	1.15 (0.52)	3	0.45 (0.37)	3	25.59 (38.15)	5	5.59 (3.19)	5	0.86	1	0.18	1	98.90 (72.13)	6
p-value	0.414		0.136		0.136		0.465		*		*		0.343	

Mainstem floodplain														
Disturbed	0.201 (0.04)	2	0.12 (0.04)	2	13.01 (5.6)	10	22.20 (28.72)	10	0.25 (0.08)	10	0.18 (0.08)	9	31.91 (19.69)	11
Ref. Std.	*	0	*	0	5 (3.7)	5	4.37 (5.78)	5	0.58 (0.80)	5	0.51 (0.83)	5	21.73 (4.59)	5
p-value	*		*		0.010		0.143		0.464		0.577		0.330	
Riparian depression														
Disturbed	*	0	*	0	*	0	*	0	0.48 (*)	1	0.15 (*)	1	*	0
Ref. Std.	1.09(*)	1	0.29 (*)	1	*	0	7.0 (7.4)	2	*	0	0.59 (*)	1	21.47 (10.65)	3
p-value	*		*		*		*		*		*		*	
Slope														
Disturbed	*	0	*	0	7.18 (8.24)	15	9.05 (18.03)	15	0.38 (0.26)	18	0.26 (0.21)	16	28.65 (19.6)	18
Ref. Std.	*	0	*	0	20.53 (17.49)	5	6.12 (3.04)	4	0.34 (0.22)	6	0.16 (0.10)	4	21.06 (10.64)	6
p-value	*		*		0.121		0.213		1.000		0.264		1.000	

p-value indicates significant differences between Disturbed and Ref. Std. wetlands ($\alpha = 0.05$, mood median test). Standard deviation in parenthesis; n = sample size

Data from Riparia's reference set of wetlands, including soils data, is now available through a searchable web interface (see http://www.riparia.psu.edu/MARbook for links to this database).

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Chapter 6
Hydrophytes in the Mid-Atlantic Region: Ecology, Communities, Assessment, and Diversity

Sarah J. Chamberlain, Denice Heller Wardrop, M. Siobhan Fennessy, and Doug DeBerry

Abstract Hydrophytes, or wetland plants, are the most conspicuous and perhaps most colorful element of wetland systems. In the mid-Atlantic region, hydrophytes have been the focus of many studies, resulting in a wealth of information on wetland classification, vegetation stressors, and plant-based assessment tools. For example, exploration of the relationship between hydrophytes and the physical aspects of wetlands has led to a new hydrogeomorphic classification of headwater systems that combines three previously distinct classes. Studies of stressors have shown that plants respond differentially to human-mediated disturbances in the surrounding landscape. Reed canary grass (*Phalaris arundinacea*), a native but highly invasive wetland grass in regional wetlands, exhibits increased establishment, growth, and biomass in response to nutrient additions, and surprisingly, in some instances, to increased sedimentation, while blue vervain (*Verbena hastata*), a denizen of freshwater wetland habitats, is intolerant of increased sediment loading. Hydrophtyes have also served as the foundation for some of the most powerful wetland assessment tools in the region. Floristic Quality Assessment (FQA) and biotic indices have been developed by a number of states within the region and in the case of FQA, for the region as a whole. This chapter examines the role of hydrophytes in these studies, as well as spotlights invasive and special status wetland species found in wetland habitats in the region.

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6.1 What Is a Hydrophyte?

Wetland plants or *hydrophytes* are the most conspicuous and perhaps most colorful element of wetland systems. Although the term has been in use since the late 1800s, our modern concept of a hydrophyte comes from Tiner (1988, 1999) who defines hydrophytes as plants that tolerate varying degrees of soil wetness and that typically grow in wetlands or in the shallow water of lakes, rivers, or streams. By this broad definition, hydrophytes encompass not only true wetland taxa, but also wetland cohorts of mostly upland (dry) species that periodically occur in wetlands. Cattail (*Typha* spp.) a plant usually found growing in standing water is a hydrophyte as is tulip poplar (*Liriodendron tulipifera*), a tree species that occurs mostly in mesic or floodplain forests. Plants like red maple (*Acer rubrum*), which is highly adaptable to many environmental conditions, are also hydrophytes, even though they could easily occur in either wetlands or drier uplands. Tiner (1999) provides an excellent and exhaustive review on the concept of a hydrophyte as well as the idea of wetland plant ecotypes.

In this chapter, we take a regional approach to hydrophytes, highlighting topics central to the mid-Atlantic region (MAR) including the relationship between plants and regional wetland classification, studies of stressors common to wetland plants, existing plant-based statewide and regional bioassessment tools, and invasive and special status hydrophytes common to the MAR.

6.2 Mid-Atlantic Floristic Setting

The MAR lies within the Appalachian and Atlantic Coastal Plain floristic provinces of North America. As its name implies, the region lies mid-way along the latitudinal gradient of the North American continent thus, in addition to temperate species, it shares many floristic elements with boreal habitats to the north as well as more southerly subtropical communities. To the east, bays and inlets along the Atlantic coast support maritime plant communities while near the western boundary, montane communities gradually give way to the rolling prairies of the Midwest (Braun 1950). Each of these broad geographic areas contributes elements to the regional vegetation resulting in a rich and varied floristic landscape.

Approximately 65–70% of the MAR is broad-leaved deciduous forest (Jones et al. 1997; Greller 2000), the majority of which constitutes secondary growth, the result of large-scale deforestation during the colonial period (McKenney-Easterling et al. 2000). Today, forested areas are concentrated primarily in the western portion of the MAR in West Virginia and parts of Pennsylvania, and in portions of the Coastal Plain, while agriculture and urban lands dominate the central and eastern sections (Polsky et al. 2000). Jones et al. (1997) estimated that greater than 70% of the Delmarva Peninsula has been deforested to make way for both agriculture and urban development. Even in areas of the MAR where forest is the dominant land cover, fragmentation has resulted in the loss of plant taxa, as well as facilitated the introduction of undesirable non-native and weedy plants (Noss and Csuti 1997).

Once dominated by chestnut (*Castanea americana*) and oak (*Quercus prinus*, *Q. alba*, and *Q. rubra*), MAR forests now support a mix of hardwood and evergreen trees that can be further organized into associations based on the dominant taxa. The southern hardwood association is dominated by oak (*Quercus*) and hickory (*Carya*) and comprises 46% of all forested areas in the MAR, while the northern hardwood association (37%) is a mixture of maple (*Acer*), beech (*Fagus grandifolia*), birch (*Betula*), and eastern hemlock (*Tsuga canadensis*) (Greller 2000; McKenney-Easterling et al. 2000). To the north, the shorter growing season and lower temperatures favor the growth of evergreen trees. In some areas, spruce (*Picea*)-fir (*Abies*) forests are locally abundant, representing more northerly species at the southern end of their natural range. Near the coast, pine forests (*Pinus palustris*, *P. taeda*, *P. glabra*, *P. virginiana*) become increasingly abundant, predominating in poor, sandy soils. This area also contains taxa more closely aligned with the southeastern Coastal Plain that reach the northern limit of their range here, aided by the more moderate maritime climate. These include warm temperate and subtropical broad-leaved evergreen species such as magnolia (*Magnolia grandiflora*) and holly (*Ilex opaca*).

The understory layer is generally well developed. Understory trees and shrubs include dogwood (*Cornus florida*), serviceberry (*Amelanchier arborea*), iron wood (*Carpinus caroliniana*), witch hazel (*Hamamelis virginiana*), hop hornbeam (*Ostrya virginiana*), and viburnums (*Viburnum*). Evergreen shrubs including rhododendron (*Rhododendron*) and laurel (*Kalmia*) are locally abundant, particularly where hemlock comprises part of the canopy (Braun 1950). A variety of perennial and annual forbs, ferns, and grasses form a generally well-developed herbaceous layer. Although native species still dominate the regional flora (67%) (Chamberlain and Ingram in press), non-native taxa are becoming increasingly prevalent as urban and agricultural development advances into natural areas.

The Great Lakes (in particular, Lake Erie) influence temperatures in the northwestern portion of the MAR. Consequently, this area supports a number of taxa that are disjunct populations of species typically found in coastal areas (Reznicek 1994). Disjunct populations of taxa typically associated with prairies also occur in shale barrens and limestone glades and on granite outcrops throughout the region (Greller 2000).

Wetland plant communities range from freshwater tidal marshes near the coast to high elevation bogs (Table 6.1). Red maple swamps are prevalent throughout the region (Rogers and McCarty 2000), as are herbaceous marshes dominated by cattail (*Typha*) and various graminoid taxa (*Juncus*, *Carex*, *Scirpus*, *Schoenoplectus*, and grasses). Seasonally flooded riparian plant communities persist along streams and rivers. Dominant species include hydrophytic trees and shrubs, as well as more mesic woody species adapted to a seasonal flooding regime including maple, birch, ash (*Fraxinus*), elm (*Ulmus*), sycamore (*Platanus occidentalis*), tulip poplar, ironwood, and willow (*Salix*). The headwaters of these systems support plants typical of floodplain, slope, and depressional wetlands.

Several unique wetland plant communities occur within the region. Atlantic white cedar (*Chamaecyparis thyoides*) swamps, once prevalent in coastal backwaters,

Table 6.1 Regional HGM subclasses with examples of wetlands and their associated plant communities within the mid-Atlantic region

HGM subclass	Example wetland types in the MAR	Distribution	Dominant plant communities
Palustrine			
Flats	Wet flatwood forests[a]	Coastal areas of Delaware, Maryland, and Virginia	Loblolly pine (*Pinus taeda*), red maple (*Acer rubrum*), and sweetgum (*Liquidambar styraciflua*), frequently with scattered pond pine (*Pinus serotina*); small trees and shrubs include sweetbay (*Magnolia virginiana*), blackgum (*Nyssa sylvatica*), red bay (*Persea palustris*), and coastal dog-hobble (*Leucothoe axillaris*)
Slopes	Calcareous fens[a]	Mountains of Pennsylvania, Maryland, and Virginia	Mosaic of shrubs and herbaceous openings; shrubs include willows (*Salix* spp.), smooth alder (*Alnus serrulata*), swamp rose (*Rosa palustris*), alder buckthorn (*Rhamnus alnifolia*), and chokeberries (*Aronia arbutifolia* and *A. prunifolia*); herbaceous layer consists of many sedges (*Carex* spp.), orchids (*Cypripedium reginae* and *Spiranthes lucida*), and various ferns
Depressions			
Temporary	Vernal pools/ponds[b]	Throughout MAR	Herbaceous; vegetation highly variable seasonally, annually and among sites; typically support a dense graminoid component of three-way sedge (*Dulichium arundinaceum*), mannagrass (*Glyceria acutiflora*), rice cut-grass (*Leersia oryzoides*), bulrush (*Scirpus* spp.), sedges (*Carex* spp.) and soft rush (*Juncus effusus*), as well as ferns (Virginia chain fern [*Woodwardia virginica*], royal fern [*Osmunda regalis*], and cinnamon fern [*Osmunda cinnamomea*]) and other forbs
Seasonal	Delmarva bays[c]	Coastal areas of Delaware, Maryland, and Virginia	Forest, shrub, or herbaceous communities; vegetation highly variable seasonally, annually and among sites; forested sites composed of red maple (*Acer rubrum*), sweet gum (*Liquidambar styraciflua*), black gum (*Nyssa sylvatica*) and loblolly pine (*Pinus taeda*); shrub communities are exclusively buttonbush (*Cephalanthus occidentalis*); herbaceous sites dominated by sedges (*Carex* spp., and *Rhynchospora* spp.) and other graminoids (*Juncus* spp., *Eleocharis* spp., *Dichanthelium* spp., *Panicum* spp., and *Glyceria* spp.)
Perennial	Shenandoah Valley sinkhole ponds[a]	Mountains of Virginia	Herbaceous communities dominated by sedges including toothed flatsedge (*Cyperus dentatus*), Barratt's sedge (*Carex barrattii*), slender sedge (*Carex lasiocarpa* var. *americana*), Torrey's bulrush (*Schoenoplectus torreyi*), black-fruited spikerush (*Eleocharis melanocarpa*), and maidencane (*Panicum hemitomon*); forbs include northern St. John's wort (*Hypericum boreale*) and dwarf burhead (*Echinodorus tenellus*) and the endemics Virginia sneezeweed (*Helenium virginicum*) and Virginia quillwort (*Isoetes virginica*)

Riverine			
Riparian forests	Atlantic white cedar swamps[d]	Coastal areas of Delaware, Maryland, and Virginia	Dominated by Atlantic white cedar (*Chamaecyparis thyoides*); common woody associates include cranberry (*Vaccinium macrocarpon*) and inkberry (*Ilex glabra*); herbaceous associates include a variety of graminoids (twig rush [*Cladium mariscoides*], white beaksedge [*Rhynchospora alba*], giant reed [*Phragmites australis*], *Juncus abortivus*, *Eleocharis olivacea*, and *Panicum ensifolium*, as well as royal fern [*Osmunda regalis*], and cinnamon fern [*Osmunda cinnamomea*])
Estuarine			
Tidal marshes	Freshwater[a]	Coastal areas of Delaware, Maryland, and Virginia	Dominated by arrow arum (*Peltandra virginica),* dotted smartweed (*Polygonum punctatum*), wild rice (*Zizania aquatica*), pickerelweed (*Pontederia cordata*), tearthumb (*Polygonum arifolium* and *P. sagittatum*) and beggar's ticks (*Bidens laevis* and *B. coronata*)
	Saltwater[a]	Coastal areas of Delaware, Maryland, and Virginia	Low marsh dominated by smooth cordgrass (*Spartina alterniflora*); associates include giant cordgrass (*Spartina cynosuroides*) and saltmarsh bulrush (*Schoenoplectus robustus*); high marsh dominated by saltmeadow cordgrass (*Spartina patens*) and saltgrass (*Distichlis spicata*)
	Brackish[e]	Coastal areas of Delaware, Maryland, and Virginia	Dominated by black needle rush (*Juncus roemarianus*) with associated species (saltgrass [*Distichlis spicata*] saltmeadow cordgrass [*Spartina patens*], smooth cordgrass [*Spartina alterniflora*] and giant cordgrass [*Spartina cynosuroides*])

[a]Virginia Department of Conservation and Recreation: Natural Heritage Program (http://www.dcr.virginia.gov/natural_heritage/) (2012)
[b]Fike (1999)
[c]McAvoy and Bowman (2002)
[d]Sipple and Klockner (1984)
[e]Harrison (2001)

have been reduced to remnant stands (Rogers and McCarty 2000), although this species is commonly planted in forested wetland restoration projects in the Coastal Plain. Bald cypress (*Taxodium distichum*) swamps reach the northern limit of their natural range in southern Delaware (Stalter 1981). Appalachian bogs, a wetland type which encompasses six community associations ranging from vulnerable to critically imperiled (NatureServe 2010), characteristically support a mosaic of tree or shrub patches interspersed with herbaceous openings. Many species of special concern occur in these wetlands including orchids and carnivorous taxa. Unique wetland plant communities are also found near the coast in Delmarva Bays and sea-level fens, both rare wetland types that face an unknown future under a changing climate.

Aside from habitat loss, one of the biggest threats to MAR wetlands is colonization by invasive plant species. Of all invasive wetland plants of freshwater wetlands in the region, perhaps the best known is purple loosestrife (*Lythrum salicaria*), a native of Europe and Asia that was introduced for ornamental and medicinal purposes. Native invasives such as reed canary grass (*Phalaris arundinacea*) also impact wetlands in the MAR. This species is an increasing problem in emergent wetlands in the region as it readily forms dense monoculture stands outcompeting other native vegetation and decreasing the value of wetlands to wildlife and humans.

6.3 Plant Communities and HGM Classification

Plants are a fundamental component of most wetland classification systems. Perhaps the most widely recognized of these is the Cowardin system, developed in the late 1970s for the National Wetlands Inventory (Cowardin et al. 1979). More recently, resource managers have favored the hydrogeomorphic (HGM) classification method (Brinson 1993), which emphasizes hydrologic site characteristics over biological attributes. This emphasis, while fundamental to the approach, may result in wetlands that are physically, but not floristically distinct.

Studies examining the relationship between plant communities and HGM classification have produced mixed results. Working in forested wetlands in New Jersey, Ehrenfeld (2005) reported no relationship between plant species composition and HGM subclass. Peterson-Smith et al. (2008) found that plant community composition corresponded to underlying hydrogeomorphology on a subregional scale, but this relationship weakened when examined at the larger, regional scale (in this case, the region studied encompassed high elevation wetlands from northern New York State to Virginia). Peterson-Smith et al. (2008) suggested that differences in substrates, types of stressors, and the influence of surrounding wetlands may account for the lack of a strong correlation.

As in Peterson-Smith's study, exploratory tools including ordination, clustering and CART (Classification and Regression Tree) analysis can aid in clarifying the link between floristic composition and HGM subclass. In Pennsylvania, the relationship between plant communities and HGM subclass was examined using

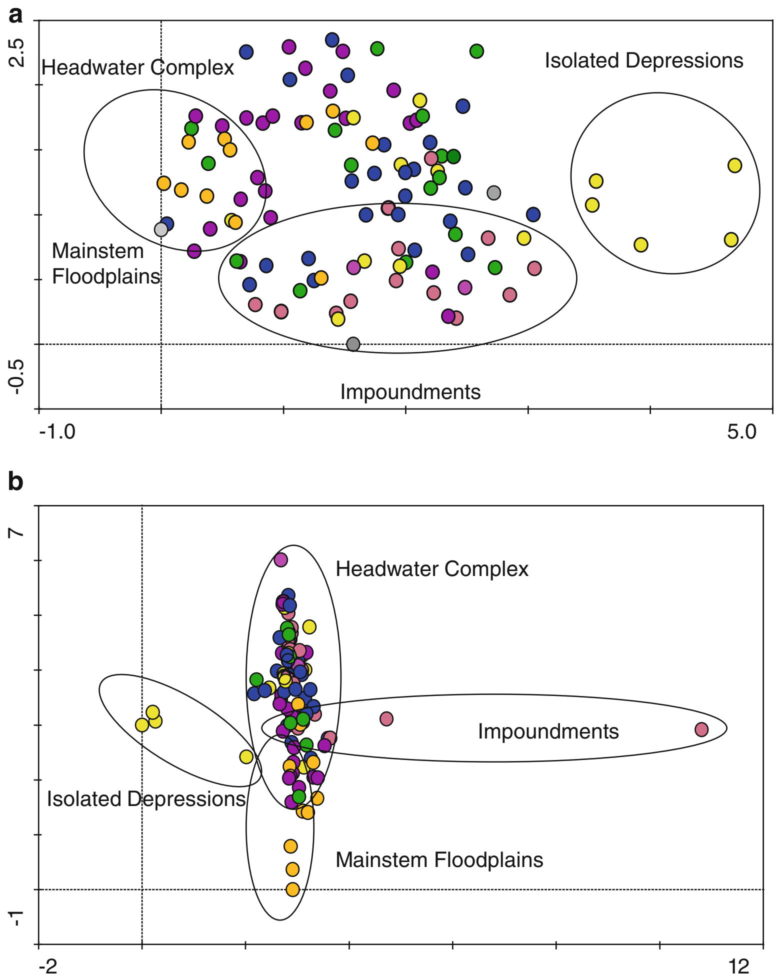

Fig. 6.1 (**a**) Detrended correspondence analysis (DCA) of HGM wetland subclasses based on plant species occurrence. Subclasses appear to be distributed along axis 1 by a gradient in hydrology which accounts for 38% of the variability. Wetland sites range from drier mainstem floodplain wetlands to wetter depressional wetlands. Axis 2 which explains 24% of variability may be related to wetland condition with impoundments clustering at the lower end and headwater complex wetlands at the higher end of the condition scale. (**b**) DCA of HGM wetland subclasses based on plant species abundance. Axis 1 explains 89% of the variability in the data and axis 2 65%. Abundance data results in a more distinct cluster for riverine headwater complex wetlands

detrended correspondence analysis (DCA) (ter Braak and Smilauer 1997). DCA identified three distinct HGM subclasses: mainstem floodplains, isolated depressions, and impoundments corresponding to gradients in hydrology and potentially condition (Fig. 6.1a, b). The remaining sites: headwater floodplains, riparian depressions, and slopes were floristically distinct from other HGM subclasses, but not substantially different from one another. Both plant species presence–absence and abundance data produced similar clusters, but abundance data produced more defined clusters. These data suggest this metric, which is a measure not just of where plants occur but where they thrive, may be the better indicator of the two.

The lack of floristic distinction among headwater wetlands has led to the formation of a new subclass referred to as riverine headwater complex (Brooks et al. 2009). Its genesis is illustrative of the issues that arise with classification systems based solely on physical characteristics. Though differences in hydrology that distinguish HGM subclasses (geomorphic setting, water source, and hydrodynamics) may be valid in a physical context, they appear to be less important in a biological context. In other words, plant species are influenced more by the presence, frequency, and duration of water at a particular site than where it came from and how it got there. Floristic data, therefore, can provide a more discerning filter with which to refine these classes on a regional level.

6.4 Factors Influencing Plant Communities

The ability of plants to establish, grow, and flourish at a particular site is influenced by a combination of abiotic and biotic factors working at different spatial and temporal scales. Typically, climatic factors influence vegetative patterns over broad geographic areas (Braun 1950), while regional or local environmental factors (parent geologic material, topography, slope, and shading) are more important at the site level (Billings 1952). In wetlands, important factors influencing plant community composition are water depth, hydroperiod, leaf litter, soil texture, and various water chemistry parameters such as pH and conductivity (Spence 1982; Grace and Wetzel 1982; Ewel 1984; Pip 1984; Rey Benayas et al. 1990; Brinson 1993; Rey Benayas and Scheiner 1993; Weiher and Keddy 1995). For example, in the glaciated regions of northeastern Pennsylvania, oligotrophic bogs support plant species that thrive on acidic, low nutrient soils. Vernal ponds form where local precipitation or snow melt fills depressional areas early in the growing season that then dry out during the hot summer months. These wetlands support taxa adapted to the cycle of inundation followed by desiccation. Once plants become established at a site, competition is the primary factor controlling composition (Connell 1983; Schoener 1983; Fowler 1986; Weiher and Keddy 1995).

Humans also have had an influence on plant community composition. Since early civilization, humans have directly altered the floristic landscape by clearing lands for agriculture and habitation, selectively cultivating plants for food and fiber, and either deliberately or accidentally introducing or eliminating species. Humans also indirectly influence plant community structure through the introduction of stressors that affect resource levels or alter site conditions (Hobbs and Huenneke 1992). Our present day landscapes are, thus, a reflection of man's influence in addition to the natural processes that govern the development, structure, and distribution of plant communities (*synecology*), and the interaction between individual species and their surrounding environment (*autecology*).

In heavily developed landscapes, anthropogenic disturbances can exceed natural disturbance levels in intensity, frequency, and duration (Taft et al. 1997) and, therefore, exert a primary influence on plant community composition (Hodgson 1986). Magee and Kentula (2005) reported a shift from native to invasive and non-native taxa following even slight changes in hydrology that mimicked patterns common in urban environments. Habitat fragmentation also creates conditions favorable to the establishment of disturbance tolerant taxa (Lopez and Fennessy 2002), particularly non-native species (Pyle 1995).

In the MAR, urbanization and agriculture are the primary types of human-mediated disturbances affecting regional plant communities. These activities produce both excess sediments and nutrients, which, in turn, influence plant community structure. Because plants exhibit different tolerances to these stressors, some taxa are able to cope and even thrive, while others are selectively filtered out of the community (Dittmar and Neely 1999).

Sedimentation is a chronic disturbance that stresses plants and/or the seed bank by altering light conditions and temperature, introducing sediment-borne pollutants, or changing the depth, permeability, and other features (O_2, moisture) of the substrate (Wardrop 1997). Sedimentation has been reported to inhibit germination and reduce biomass in some species (Mahaney et al. 2004a), and to decrease both species richness (Jurik et al. 1994; Dittmar and Neely 1999; Mahaney et al. 2004b) and diversity (Dittmar and Neely 1999). Nutrient enrichment impacts wetland plants by altering nutrient cycles and shifting competitive interactions among species. High nutrient levels generally favor plants that are able to opportunistically consume excess resources and rapidly increase biomass (Wetzel and van der Valk 1998; Galatowitsch et al. 1999). These species are often non-native or aggressive native species (cattail [*Typha*], reed canary grass [*Phalaris arundinacea*], common reed [*Phragmites australis*], purple loosestrife [*Lythrum salicaria*], and duckweed [*Lemna*]) that quickly form monotypic stands in nutrient-enriched systems (Hobbs and Huenneke 1992; Galatowitsch et al. 1999). Grasses are also favored over forbs in enriched habitats (Hobbs and Huenneke 1992).

We can examine how plants respond to stressors using controlled greenhouse experiments, which mimic ambient conditions. Using germination trials, Wardrop and Brooks (1998) were able to classify wetland plants into tolerance categories based on their response to varying magnitudes of sedimentation (Table 6.2). They reported that plants categorized as moderately tolerant to very tolerant actually increased in cover with increasing sedimentation loads.

In a follow-up study, Mahaney et al. (2004a) simulated three HGM subclasses (depression, slope, and upper perennial (headwater) floodplain) in a greenhouse setting and examined the emergence and growth of 14 wetland taxa subjected to sediment or nutrient stress. Although both stressors influenced species, they found responses varied by HGM subclass. Sedimentation, for example, had the greatest impact on taxa in simulated riparian depressions, but virtually no effect on plants in headwater floodplain microcosms. Differences in physical parameters among HGM subclasses (i.e., soil moisture, organic matter) may account for this varied response and Mahaney et al. (2004a) suggest that additional experiments are necessary to tease out these potentially confounding factors.

Table 6.2 Some common wetland plants of the mid-Atlantic region as indicators of increased sedimentation or water source

	Sediment tolerance[a]				Indicator[b] of	
	Very	Moderately	Slightly	Intolerant	Surface water	Groundwater
Asclepias syriaca				X	X	
Aster vimineus				X		
Brachyelytrum erectum		X				X
Carex emoryi		X				X
Carex folliculata		X				X
Carex intumescens			X			X
Carex lurida	X					
Carex prasina		X				
Carex retroflexa		X				
Carex stricta		X				
Carex vulpinoidea		X				
Cirsium arvense				X		
Dipsacus sylvestris	X					
Dulichium arundinaceum	X					
Equisetum arvense			X			
Euthamia graminifolia			X			
Impatiens capensis	X					
Juncus canadensis			X			
Leersia oryzoides	X					
Lysimachia nummularia					X	
Mentha arvensis					X	
Phalaris arundinacea		X				
Poa pratensis				X	X	
Polygonum saggitatum	X					
Sagittaria latifolia			X			
Solidago canadense			X			
Solidago patula	X					X
Solidago uliginosa	X					
Symphyotrichum novae-angliae	X					
Symplocarpus foetidus		X			X	
Thelypteris noveboracensis		X				X
Triadenum virginicum		X				
Urtica dioica				X	X	
Verbena hastata				X		

[a]Wardrop and Brooks (1998)
[b]Goslee et al. (1997)

Nitrogen enrichment influenced growth in only about one-half of the taxa examined and was least effective in riparian depression wetlands. In some taxa, there was an observed increase in belowground biomass and this partitioning of resources to belowground structures may account for the apparent lack of response in aboveground growth.

No clear relationship was found when stressor sensitivity was examined based on occurrence in either impacted or least disturbed wetlands (Mahaney et al. 2004a). Plants typically found in impacted wetlands were no less sensitive than those from the least disturbed sites. Furthermore, taxa from floodplains showed no response to stressors for either pristine or impacted sites. These taxa, by virtue of their exposure to intermittent flooding, may be somewhat tolerant to low levels of sedimentation.

Mahaney et al. (2004b) also examined community responses to sedimentation and nutrient (N) enrichment stressors in simulated greenhouse experiments. As with individual plant responses, plant community responses varied by HGM subclass; however, some general trends were noted. Increased sedimentation decreased germination in most taxa, while nutrient enrichment promoted the establishment and growth of non-native and invasive taxa. Reed canary grass (*Phalaris arundinacea*), a native but highly invasive wetland grass exhibited increased establishment, growth, and biomass in response to nutrient additions, and surprisingly, in some instances, to increased sedimentation. In a community setting, it dominated all HGM subclasses. Its ability not only to survive, but also to thrive under stress, as well as its capacity to quickly colonize bare ground has implications for how we manage and restore wetlands.

Although studies of disturbance are often specific to individual stressors, most stressors act synergistically in a given environment (Chapin et al. 1987; Hobbs and Huenneke 1992). Floodwaters that stress wetland systems via inundation also deposit sediments, and sediments, in turn, are vectors for excess nutrients (Jurik et al. 1994). This interplay of stressors affects individual taxa in different ways. When sediment and nutrient co-occur, the result for some taxa (i.e., Canada thistle [*Cirsium arvensis*] and reed canary grass) is increased biomass (Mahaney et al. 2004b). This outcome, although not immediately intuitive, occurs because sedimentation decreases species density and, as a result, lowers intraspecific competition for increased available nitrogen.

Exploratory statistical techniques can also be used to examine the relationship between individual plant species and known stressors. Canonical correspondence analysis (CCA) was used to elucidate this nexus for headwater complex wetlands in the Ridge and Valley Physiographic Province of Pennsylvania (Fig. 6.2). CCA produced five distinct condition groupings (Highest, High, Moderate, Low, and Lowest) corresponding to a variety of stressors measured in the field. Higher condition categories had very few non-native and invasive plants and a greater number of perennials and cryptogams. They were associated with high soil organic matter (SOM) and coarse woody debris (CWD) and occurred in a forested context. In contrast, lower condition categories were dominated by non-native and invasive plants, and had a greater number of annuals and fewer cryptogams. These categories were associated with sedimentation and eutrophication stressors and occurred in a shrub or forb-dominated context.

The relationship between plants and stressors forms the basis for the development of plant-based bioassessment methods, which are discussed in detail in the following section.

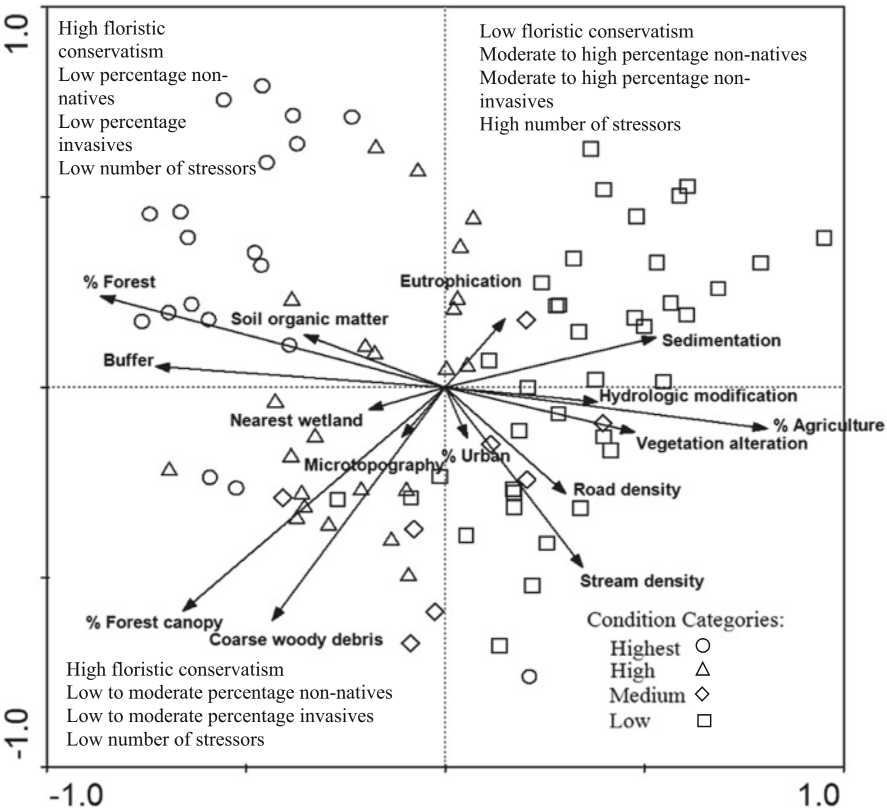

Fig. 6.2 Canonical correspondence analysis (CCA) for plant species observed in Headwater Floodplain Wetlands in the Ridge and Valley Physiographic Province. Plant associations that related to stressors in the environment (eutrophication, hydrologic modifications, and vegetation alteration) were typically comprised of generalist species and had high numbers of non-native and invasive taxa. Plant associations that related to a high percentage of forest, intact buffers, and high soil organic matter were largely comprised of perennial, native species with high conservatism values (see Sect. 6.5.1 for a description of conservatism)

6.5 Monitoring and Assessment Using Plants

Throughout history, humans and plants have shared a complex and for the most part, mutually beneficial relationship. From the earliest civilizations to the present, plants have been used for food, clothing, medicine, religious ceremonies, and to indicate the presence of water, minerals, and other substances. In the first century CE, rushes and reeds were used to search for springs and for ground that is "sweet and suitable for grain" (Ash 1941). This concept was carried over into the Middle Ages where plants were commonly used to find local mineral reserves (Brooks 1979) and identify saline soils (Mirsal 2008). In early America, pioneers used plants to locate homesteads and pinpoint water sources along migration routes (Cannon 1971). More recently, plants have been used as indicators of groundwater (Goslee et al. 1997), soil type (Kelley 1922; Gordon 1940; Bigler and Richardson 1984), water chemistry (Jeglum 1971; Matters and Bozon 1989), and for delineating wetland boundaries (Reed 1988, 1997; Scott et al. 1989; Segelquist et al. 1990; Veneman and Tiner 1990) and assessing wetland condition (Lopez and Fennessy 2002; Miller et al. 2006; Miller and Wardrop 2006; Mack et al. 2000; Veselka et al. 2010). Working in central Pennsylvania, Goslee et al. (1997) identified several plant species whose presence was correlated with either groundwater or seasonal surface water-fed wetlands (Table 6.2). Ericaceous shrubs are often associated with nutrient-poor environments such as bogs and pocosins while cattails, which are often found growing in urban and suburban wetlands, may indicate high nutrient inputs (Tiner 1999).

Miller et al. (2006) and others have identified several advantages to plants as indicators. First, they are a ubiquitous element of wetlands and, in most cases, can be identified with a modest amount of training (USEPA 2002c). Second, the plant community is immobile and, therefore, directly linked to the surrounding physical, chemical, and biological environment. Third, plant attributes are typically easy to measure and quantify. Finally, many plant community attributes, as well as individual species, have been shown to be sensitive to anthropogenic disturbances including sedimentation (van der Valk 1981, 1986; Wardrop and Brooks 1998; Mahaney et al. 2004a, b), nutrient enrichment (Pip 1984; Goldberg and Miller 1990; Kadlec and Bevis 1990; Hobbs and Huenneke 1992; Templer et al. 1998; Craft and Richardson 1998; Mahaney et al. 2004a, b; Drohan et al. 2006), and hydrologic modifications (Gosselink and Turner 1978; van der Valk 1981; Spence 1982; Squires and van der Valk 1992). According to Bedford (1996) plants are “one of the best indicators of the factors that shape wetlands within their landscape.” Likewise, plant-based bioassessment tools have emerged as one of the best indicators of human-mediated disturbances (USEPA 2002c).

A number of plant-based bioassessment tools have been developed for the MAR either at a regional or subregional scale (Table 6.3). Floristic Quality Assessment (FQA) tools were initially produced at the state level by Virginia (Virginia Department of Environmental Quality 2004), West Virginia (Rentch and Anderson 2006), and Delaware (McAvoy 2009, personal communication) and at a subregional level for Pennsylvania (Beatty et al. 2002; Bowman’s Hill Wildflower Preserve 2009). In 2009, these diverse efforts were scaled up to create the Mid-Atlantic FQA (Chamberlain and Ingram in press) and interactive Floristic Quality calculator (http://www.mawwg.psu.edu/fqaicalc/FQAICalc.html#). Vegetation IBIs have been developed for wetlands in both Pennsylvania (Miller et al. 2006) and West Virginia (Veselka et al. 2009). In addition, some rapid assessment methods have incorporated vegetation metrics as part of their protocol (Brooks et al. 1999, 2009; Jacobs 2010; Veselka et al. 2010). Plant-based rapid assessment metrics are primarily based on invasive species (Brooks et al. 1999, 2009; Jacobs 2010), but Veselka et al. (2010) also incorporated the presence of sediment or nutrient tolerant plant taxa in their West Virginia Wetland Rapid Assessment Protocol (WVWRAP). All of these tools provide a strong foundation for the development of monitoring and assessment programs and strategies within the MAR.

Table 6.3 A summary of existing plant-based bioassessment methods in the mid-Atlantic region

Bioassessment method	Description	Applicability within region	Source
FQA/FQI	Statewide coefficients of conservatism	Delaware	William McAvoy, Delware Natural Heritage Program, personal communication
FQA/FQI	Coefficients of conservatism for most common wetland plants	Virginia	Virginia Department of Environmental Quality (2004)
FQA/FQI	Coefficients of conservatism for wetland and riparian communities	West Virginia	Rentch and Anderson (2006)
FQA/FQI	Coefficients of conservatism for Penns Creek Watershed	Ridge & Valley Physiographic Province, Pennsylvania	Beatty et al. (2002)
FQA/FQI	Coefficients of conservatism by Physiographic Province	Mid-Atlantic region (PA, DE, MD, VA, WV)	Chamberlain and Ingram (in press)
PSI	Coefficients of conservatism	Coastal Plain and Piedmont Physiographic Provinces, Pennsylvania	Bowman's Hill Wildflower Preserve (2009)
IBI	IBIs based on Cowardin and HGM classification systems	West Virginia	Veselka et al. (2010)
IBI	IBI for headwater complex wetlands	Ridge & Valley Physiographic Province, Pennsylvania	Miller et al. (2006)
DECAP	HGM functional assessment for flat, riverine and depressional nontidal wetland subclasses in the Coastal Plain of Delaware and Maryland	Delaware and Maryland	Delaware Department of Natural Resources and Environmental Control (2008)
HGM	HGM functional models for five wetland subclasses	Pennsylvania (Glaciated Plateau, Piedmont, Glaciated Poconos, Allegheny Plateau, Ridge & Valley)	Wardrop et al. (2007a)

6.5.1 Floristic Quality Assessment

FQA is a bioassessment method that uses characteristics of the plant community to derive an estimate of nativity or habitat quality (Swink and Wilhelm 1979, 1994). Implicit in its application is the premise that areas with species assemblages closer to those of European presettlement times are more reflective of truly native, non-disturbed habitat and, therefore, of higher quality (Wilhelm and Ladd 1988; Swink and Wilhelm 1994; Nichols 1999). Disturbance represents a mode of introduction for invasive or cosmopolitan species, and thus, when habitats are disturbed, quality is diminished. It is important to note that disturbance is in itself a relative term that could also be used to describe the types of disturbances known to occur during presettlement times, such as incendiary fires set by Native Americans to clear patches of ground (Noss 1985). The concept of disturbance as it relates to FQA, however, is purely contemporaneous with postsettlement; that is, anthropogenic disturbance following European occupation of the North American continent (Wilhelm and Ladd 1988).

A key element of FQA is the idea that individual plant species have evolved varying degrees of tolerance to disturbance or environmental stress (Odum 1985; Chapin 1991; Hobbs and Huenneke 1992), and exhibit varying degrees of fidelity to specific habitat integrity (Herman et al. 1997; Mushet et al. 2002). This combination of tolerance and fidelity is expressed as a coefficient of conservatism or *C*-value (Swink and Wilhelm 1979, 1994). The *C*-value is a numerical assignment between 0 and 10 applied to plant species by a panel of experts with knowledge of the native flora of a particular region (Andreas and Lichvar 1995; Alix and Scribailo 1998; Nichols 2001). Plant species with high *C*-values typically occur in high quality habitats, while species with low *C*-values occur in a wide variety of conditions and generally are highly tolerant of disturbance (Wilhelm and Ladd 1988; Matthews 2003).

By its broadest definition, FQA encompasses a number of plant-based metrics (Taft et al. 1997). Most applications, however, have focused on the Floristic Quality Index (FQI), a metric that uses both the aggregate conservatism and native species richness of the plant community to derive a measure of condition or quality. As originally conceived by Swink and Wilhelm (1979, 1994), the index is calculated according to the following equation:

$$\mathrm{FQI} = \bar{C}(\sqrt{N})$$

where $\bar{C}$ represents the average coefficient of conservatism for native species, and N is native species richness.

Studies evaluating the efficacy of FQI as a tool for assessing wetlands typically rank sites according to some anthropogenic disturbance criterion and test for linear correlations between FQI and disturbance rank (Fennessy et al. 1998; Mack et al. 2000; USEPA 2002b; Wilcox et al. 2002). The resultant “dose–response curves” are graphically depicted as an *X*–*Y* coordinate scatterplot with FQI on the *Y*-axis and disturbance gradient on the *X*-axis (USEPA 2002b; Stevenson and Hauer 2002). Using this approach in the MAR, researchers have found that FQI effectively differentiates wetlands across disturbance gradients in Pennsylvania (Miller and Wardrop 2006) and Virginia (Nichols et al. 2006; DeBerry 2006) (Fig. 6.3).

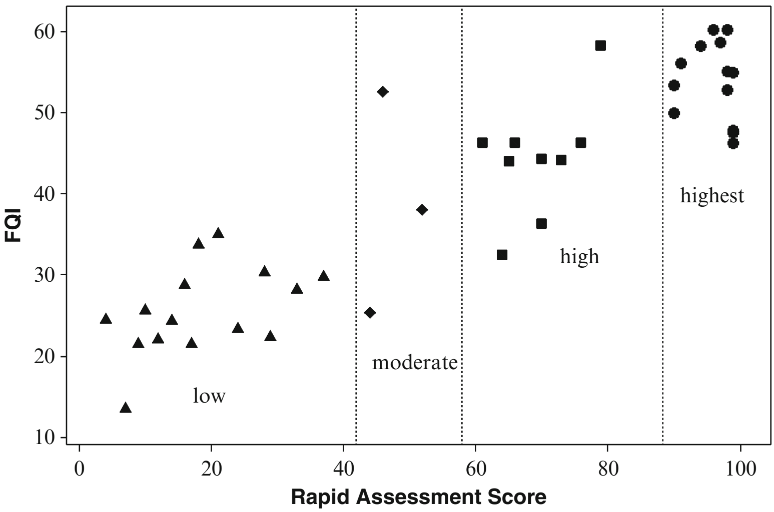

Fig. 6.3 Dose–response curve for Headwater Floodplain Wetlands in the Ridge & Valley of Central Pennsylvania. Floristic quality index (FQI) is strongly related to Rapid Assessment Score ($r=0.9$, $P<0.001$), a measure of landscape and site-specific stressors (Wardrop et al. 2007a). Condition categories are shown as low, moderate, high, and highest based on the rapid assessment score breakpoints of Wardrop et al. (2007b)

While the FQI has proven to be a valuable tool, modifications in the index have been undertaken to increase its efficacy for wetland assessments (Rooney and Rogers 2002; Miller and Wardrop 2006; DeBerry 2006). In their study of 40 headwater wetland sites in central Pennsylvania, Miller and Wardrop (2006) noted that performance was improved by modification of the index to account for over-sensitivity to species richness, as well as the presence of non-native species. Nichols et al. (2006) and DeBerry (2006) found that FQI was sensitive to vegetation layer in forested Virginia wetlands, noting that FQIs calculated separately for herbaceous and woody understory strata provided better correlation with wetland condition than did a canopy or overall composite index across all strata. The diminished response of trees has been attributed to *ecological inertia*—the idea that trees are characterized by longer disturbance response times relative to herbaceous and/or woody understory species; therefore, their presence may not be indicative of existing ecological condition (Chapin 1991; Lopez and Fennessy 2002). To address variability of the strata at forested wetland sites, Nichols et al. (2006) and DeBerry (2006) recommended that a stratum-based approach to the floristic quality concept should be considered when applying the method for forested wetlands in the MAR. DeBerry (2006) also found that an abundance-weighted version of the index was significantly correlated with site condition, and concluded that such an index may be useful in conferring additional information about the ecology of the system (as reflected in relative species abundance) without losing relative site ranks based on conservatism alone.

In some Midwestern states, FQI has recently been implemented as an assessment tool for monitoring wetland mitigation success (Herman et al. 2001; Bernthal 2003; Mack et al. 2004). However, there is some concern that FQI may have limited application in created wetland assessment because mass grading during site construction results in disturbance regimes that are indistinguishable from other anthropogenic stressors (Atkinson et al. 1993; DeBerry and Perry 2004; DeBerry 2006). The lack of a clear disturbance gradient makes it difficult if not impossible to define meaningful disturbance ranks for young sites and, therefore, difficult to test the efficacy of the FQA concept in mitigated wetlands using traditional dose–response analyses.

Though the application of FQA to mitigation assessment in the MAR has been limited, two studies deserve note. Balcombe et al. (2005) applied the FQA concept in a post hoc analysis of vegetation data across several wetland mitigation and reference sites in West Virginia, concluding that *C*-values were higher in reference wetlands and in older mitigation sites. By contrast, DeBerry (2006) evaluated a chronosequence of 15 wetland mitigation sites in Virginia and found no correlation with site age, but noted that FQIs calculated for the herbaceous layer were positively correlated with site quality as reflected by soil physiochemical parameters and other floristic quality indicators (e.g., diversity, percent native species, etc.).

6.5.2 Mid-Atlantic Regional Floristic Quality Assessment

Recent emphasis on advancing floristic assessment at the regional scale has led to collaborative efforts to develop comprehensive *C*-values that can be applied by physiographic region rather than political boundary. Regional efforts have been undertaken in North Dakota (Northern Great Plains Floristic Quality Assessment Panel 2001), South Florida (Mortellaro et al. 2009) and the Northeastern United States (Bried et al. 2012). In 2009, the Mid-Atlantic Regional FQA Project was initiated as a joint effort between USEPA Region 3 and its member states to advance floristic assessment in the region (Chamberlain and Ingram in press). A botanical panel of experts familiar with the regional flora ranked 2,822 native taxa (non-native taxa were not evaluated). Ranking was done by ecoregion, although most taxa were given a single regional coefficient value. Not surprisingly, most taxa had coefficient values falling within the middle part of the scale, ranging from 4 to 8. Only 41 taxa including native invasives (e.g., reed canary grass) and common oldfield and garden weeds were given a rank of 0. The highest ranks (9 or 10) were given to graminoids (grasses, sedges, and rushes) and members of the Orchidaceae and Asteraceae. Figure 6.4 shows a comparison among regional ranks and those assigned on a statewide or subregional level.

Taxa with lower conservatism (0–3 and 4–6) ranks occurred throughout the MAR. These generalist species are found in a wide variety of habitats and their ubiquitous presence is not unexpected. Chamberlain and Ingram (in press) also noted that conservative species (those with ranks of 7–10) were found in all physiographic provinces within the MAR in numbers similar to generalist taxa (rank of

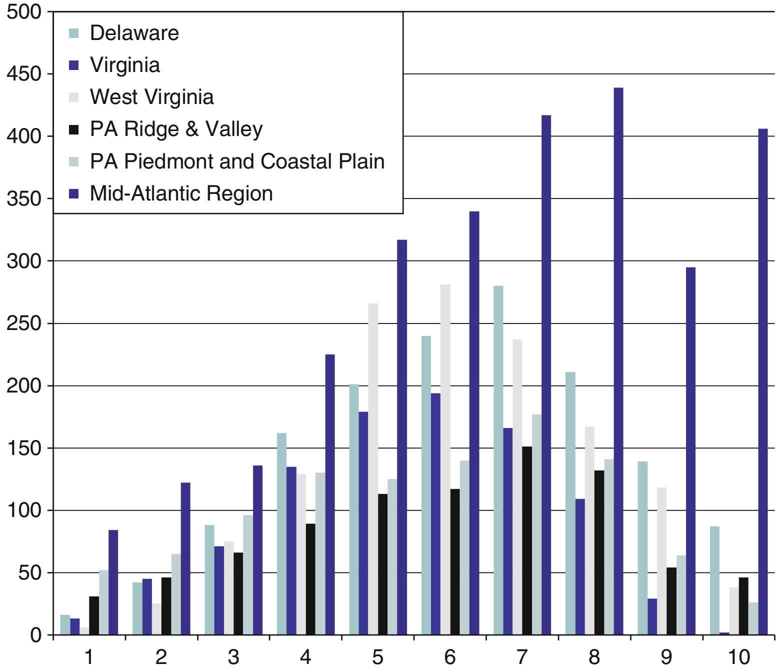

Fig. 6.4 Distribution of coefficient of conservatism values (1 lowest, 10 highest) for statewide, subregion and regional floristic quality assessments. Plants assigned a coefficient of 0 were not included because some methods gave non-native species this rank, while others assigned coefficient ranks only to native species. Pennsylvania values are reported by region because different methods were used to assign coefficients. For most treatments, rankings approximate a normal distribution, while regional coefficients are skewed toward higher levels of conservatism

0–6). The presence of conservative taxa in the Piedmont and Coastal Plain where significant development has taken place was both unexpected and encouraging. Their presence suggests that areas with high concentrations of conservative species should be identified and given priority for protection and/or acquisition.

6.5.3 Plant-Based Indices of Biotic Integrity

Although FQA is robust and effective as a stand alone bioassessment tool, components of FQA have been incorporated into multi-metric assessment tools including HGM functional models (Wardrop et al. 2007a) and vegetation-based Indices of Biotic Integrity (IBIs) (Miller et al. 2006; Veselka et al. 2010). While HGM functional models are addressed elsewhere, this section will focus on the development and use of vegetation-based IBIs in the MAR.

Karr and Chu (1999) first coined the term *biotic integrity* as the premise that healthy ecosystems support and maintain a balanced, adaptive community of organisms with species diversity, composition, and functional organization comparable to that of natural habitats within a given region. Originally developed for streams using fish as an indicator assemblage (Karr and Dudley 1981), IBIs have been developed for several habitat types and taxonomic groups. Over the past few decades six major biological assemblages have been used to develop IBIs for freshwater wetlands: plants, macroinvertebrates, fish, algae, amphibians, and birds (USEPA 2002a). Although each group displays attributes that are applicable to the bioassessment framework, a growing body of literature indicates that plants have been the most widely used assemblage for wetland IBI development (Adamus et al. 2001; Andreas and Lichvar 1995) and that plant-based IBIs are effective tools for assessing the condition of wetlands and other habitats.

In the MAR, vegetation IBIs have been developed for headwater complex (floodplain, depression, and slope) wetlands within the Ridge & Valley Physiographic Province of central Pennsylvania (Miller et al. 2006) and for six wetland classes in West Virginia (Veselka et al. 2010).

Miller et al. (2006) selected eight metrics for inclusion in their IBI. Of these, two were community-based (FQAI, % cover of tolerant species), five were functional groups (% annuals, % non-native, % invasives, % trees, and % cryptogams), and one was species-specific (% cover of reed canary grass). All eight metrics were significantly correlated to disturbance ($P<0.001$), although measures of floristic quality (FQI, % cover of tolerant plant species, and % non-native species) showed the strongest relationships. They found standard plant community measures including species richness, diversity, and evenness did not correlate strongly with disturbance as has been similarly reported from other studies (Wardrop and Brooks 1998).

Miller et al. (2006) further tested their index for application in other Physiographic Provinces and to other HGM subclasses. The index was readily transferrable to most other provinces, although the lack of sites within the Piedmont Province made it difficult to state any definitive conclusions about its efficacy in this region.

Many of the same metrics used to build the headwater complex IBI were also incorporated into indices for three other HGM classes: riverine lower perennial (mainstem floodplain), lacustrine (fringing), and seasonal (isolated) depressions. Although all three indices were highly correlated with disturbance, some individual metrics were extremely variable in their response. This was especially evident with riverine and lacustrine metrics. Many plant taxa found in disturbance-oriented environments like floodplains have evolved a degree of tolerance to low levels of stress associated with flooding. Exposure to these stressors, therefore, results in little or no response (Mahaney et al. 2004a).

In West Virginia, Veselka et al. (2010) developed six plant-based IBIs based on the Cowardin (Cowardin et al. 1979) and HGM (Brinson 1993) classification systems. Three Cowardin vegetation classes (emergent, scrub-shrub, and forested) and three HGM management classes (depression, riverine, and impoundment) were selected for IBI development. Indices were built using one to five metrics chosen for their ability to distinguish between reference and stressed sites. Of the six indices, all but the impoundment subclass were significantly correlated with disturbance. However, sensitivity to disturbance was greatly increased when Cowardin metric scores were combined with their corresponding HGM subclass metric scores.

6.6 Invasive Plant Species of the MAR

Invasive plants are those that colonize, grow, and spread rapidly altering plant community composition and disrupting natural processes (Mid-Atlantic Exotic Pest Plant Council 2011). It is estimated that invasive species cost the United States $34.7 billion each year in control efforts and agricultural losses. Invasive species may be native or non-native and it is useful to examine these two groups separately as the mode of introduction, factors influencing their spread, and control methods are quite different.

Plant taxa that occurred on the continent prior to European settlement are generally considered native to North America. These species have evolved under a set of conditions that moderate their growth and distribution, and have existed alongside the herbivores and other plants that share their community. Perturbations in the surrounding environment can interrupt this balance resulting in the loss of individuals or whole populations. In some cases, stress can facilitate the growth and colonization of native taxa. Cattail (*Typha* spp.) can rapidly grow and colonize wetland areas under elevated nutrient levels leading to monoculture stands. One of the worst native invasives in the MAR is reed canary grass (*Phalaris arundinacea*). In a study of Pennsylvania wetlands, Rubbo (2004) found *Phalaris* suppressed seed germination in box elder (*Acer negundo*) and may decrease seedling survival in green ash (*Fraxinus pennsylvanica*). Control methods, particularly shading and herbicide application, were effective in reducing the aboveground biomass of *Phalaris* and treated plots experienced increased seedling growth of native riparian trees. The results of Rubbo's study provide insight into the long-term management of *Phalaris* in floodplain communities. Management plans that focus on promoting seedling germination and survival will likely have greater success as shading from mature trees and eventual canopy closure will inhibit the growth and spread of this invasive grass.

Non-native plants are those taxa introduced to an area where they did not previously occur naturally. Non-native species come from other continents, ecosystems, states, or habitats. In their new environment, they have few, if any, natural controls thus they have the potential to spread rapidly to the detriment of the native flora. Not all non-native plants are invasive. Many such as Queen Anne's lace (*Daucus carota*) coexist with native flora and pose little threat to the natural ecosystem. Others, including common reed (*Phragmites australis*) and purple loosestrife (*Lythrum salicaria*) have successfully invaded and colonized entire wetland areas. The effect of invasive species on native biological communities can be dramatic. For instance, purple loosestrife makes up more than 50% of the biomass production in some wetlands, where it outcompetes such hardy native species cattails (*T. latifolia*) and bulrushes (*Scirpus* spp.). Generally, the establishment of invasive species is linked to decreased biodiversity (including a reduction of some endangered species), and lower habitat quality.

Abiotic conditions in wetlands are also altered by invasive native and non-native plants. For example, floating mats of vegetation can substantially reduce dissolved oxygen concentrations in the water both through the oxygen demand created by their decomposition upon senescence, and the shade they create in the water column, which reduces oxygen production by phytoplankton and submerged plants (Howard and Harley 1998). Floating species can also alter the normal succession of a wetland. *Salvinia molesta* (giant salvinia), which has been recorded in Virginia, forms floating mats on which herbaceous plants grow, eventually giving way to woody shrubs and small trees (Cook 1993).

Whether native or introduced, invasive plants typically share several common traits including rapid growth rates, short life cycle, high reproductive output, deep rooting zones, and pollination by wind or generalist pollinators (D'Antonio 1993; Burke and Grime 1996; Anderson et al. 1996; Green and Galatowitsch 2001). They are also highly competitive. Once established in an area, they can quickly take over, decreasing the productivity of other wetland species and forming dense, monotypic stands (Morrison and Molofsky 1998; Wetzel and van der Valk 1998; Barnes 1999; Green and Galatowitsch 2001) with little wildlife value. Invasive species can also alter the structure and function of invaded habitats, thereby hindering the establishment of other taxa (Thompson 1991; D'Antonio 1993; Templer et al. 1998). In extreme situations, invasives can obstruct water flows, clog boat motors, block water intake pipes and other installations and greatly reduce the recreational value of wetlands and other waters by curtailing accessibility, decreasing fish production, and promoting habitat for hosts of parasitic diseases.

Invasions may be spontaneous or the result of deliberate or inadvertent introductions of plants (Galatowitsch et al. 1999). Many of our invasive plants were intentionally introduced by the European colonists for use as culinary herbs or for gardening, livestock forage, or erosion control. Some invasive species were inadvertently introduced through use as packing material, in ship ballast, or via imported products. The Mid-Atlantic Exotic Pest Plant Council (MA-EPPC) currently tracks 253 invasive plant taxa in the region (Mid-Atlantic Exotic Pest Plant Council 2011) (Table 6.4).

6.6.1 Prominent Invasive Species in the Mid-Atlantic Region

Wetlands in the MAR, as around the world, have been drastically altered by invasive species (Meffe and Carroll 1994; Zedler and Rea 1998). The most common invasive species in the MAR are typically also found through the United States as they have expanded their range with increased human habitat alteration. Species such as common reed, reed canary grass, purple loosestrife, and hybrid cattail (*Typha* × *glauca*) are found throughout North America. While the total number of invasive species in

Table 6.4 Invasive plant taxa in the Mid-Atlantic region; occurrences are reported by state or designated as "all" if considered invasive in all states within the MAR

Common name	Scientific name	State reported invasive
Velvetleaf	*Abutilon theophrasti*	VA
Japanese maple	*Acer palmatum*	VA
Norway maple	*Acer platanoides*	All
Sycamore maple	*Acer pseudoplatanus*	PA
Bishop's goutweed	*Aegopodium podagraria*	PA
Common horse chestnut	*Aesculus hippocastanum*	PA
Colonial bentgrass	*Agrostis capillaris*	VA
Redtop	*Agrostis gigantea*	VA
Tree-of-heaven	*Ailanthus altissima*	All
Carpet bugle	*Ajuga reptans*	VA
Chocolate vine	*Akebia quinata*	MD, PA, VA
Mimosa	*Albizia julibrissin*	VA, WV
Garlic mustard	*Alliaria petiolata*	All
Wild garlic	*Allium vineale*	MD, PA, VA, WV
European alder	*Alnus glutinosa*	PA
Alligatorweed	*Alternanthera philoxeroides*	VA
Amur peppervine	*Ampelopsis brevipedunculata*	All
Sweet vernalgrass	*Anthoxanthum odoratum*	MD
Japanese angelica tree	*Aralia elata*	PA
Common burdock	*Arctium minus*	MD, PA, VA, WV
Tall oatgrass	*Arrhenatherum elatius*	VA
Mugwort	*Artemisia vulgaris*	MD, PA, VA
Small carpetgrass	*Arthraxon hispidus*	MD, PA, VA, WV
Giant reed	*Arundo donax*	MD, VA
Common bamboo	*Bambusa vulgaris*	DC
Japanese barberry	*Berberis thunbergii*	All
European barberry	*Berberis vulgaris*	PA
European white birch	*Betula pendula*	MD
Birdsrape mustard	*Brassica rapa*	VA
Bald brome	*Bromus racemosus*	WV
Poverty brome	*Bromus sterilis*	MD
Cheatgrass	*Bromus tectorum*	PA, WV
Paper mulberry	*Broussonetia papyrifera*	MD, PA, VA
Butterfly bush	*Buddleja davidii*	PA, WV
Shepard's purse	*Capsella bursa-pastoris*	VA
Balloonvine	*Cardiospermum halicacabum*	VA
Spiny plumeless thistle	*Carduus acanthoides*	VA
Musk thistle	*Carduus nutans*	MD, PA, VA
Japanese sedge	*Carex kobomugi*	MD, VA
Chinese catalpa	*Catalpa ovata*	MD, PA
Northern catalpa	*Catalpa speciosa*	MD, VA
Oriental bittersweet	*Celastrus orbiculatus*	All
Cornflower	*Centaurea cyanus*	MD

(continued)

Table 6.4 (continued)

Common name	Scientific name	State reported invasive
Tyrol knapweed	*Centaurea dubia* ssp. *vochinensis*	VA
Brown knapweed	*Centaurea jacea*	VA
Spotted knapweed	*Centaurea stoebe* ssp. *micranthos*	All
Common mouse-ear chickweed	*Cerastium fontanum*	MD, VA
Greater celedine	*Chelidonium majus*	MD
Chicory	*Cichorium intybus*	MD, VA, WV
Canada thistle	*Cirsium arvense*	All
Bull thistle	*Cirsium vulgare*	MD, PA, VA, WV
Sweet autumn virgin's bower	*Clematis terniflora*	DE, MD, VA
Asiatic dayflower	*Commelina communis*	MD, PA, VA
Poison hemlock	*Conium maculatum*	PA, VA, WV
Field bindweed	*Convolvulus arvensis*	DE, PA, VA
Smooth bedstraw	*Cruciata laevipes*	VA
Bermuda grass	*Cynodon dactylon*	VA
Scotch broom	*Cytisus scoparius*	DE, VA
Orchard grass	*Dactylis glomerata*	MD, VA
Jimsonweed	*Datura stramonium*	MD, PA, WV
Queen Anne's lace	*Daucus carota*	MD, VA, WV
Fuzzy pride-of-Rochester	*Deutzia scabra*	MD, PA, VA
Deptford pink	*Dianthus armeria*	MD
Chinese yam	*Dioscorea polystachya*	MD, VA, WV
Common teasel	*Dipsacus fullonum*	VA
Cutleaf teasel	*Dipsacus laciniatus*	MD
Indian mock-strawberry	*Duchesnea indica*	MD, PA
Mexican tea	*Dysphania ambrosioides*	MD
Blueweed	*Echium vulgare*	MD
Eclipta	*Eclipta prostrata*	MD
Brazilian egeria	*Egeria densa*	DE, VA
Water hyacinth	*Eichhornia crassipes*	DE
Russian olive	*Elaeagnus angustifolia*	DE, MD, PA, VA
Thorny olive	*Elaeagnus pungens*	VA
Autumn olive	*Elaeagnus umbellata*	DE, MD, PA, VA
Quack grass	*Elymus repens*	MD, VA
Weeping lovegrass	*Eragrostis curvula*	MD, VA
Winged burning bush	*Euonymus alatus*	All
European spindletree	*Euonymus europaeus*	VA
Winter creeper	*Euonymus fortunei*	MD, VA
Leafy spurge	*Euphorbia esula* L.	VA
Japanese knotweed	*Fallopia japonica*	All
Sakhalin knotweed	*Fallopia sachalinensis*	MD
Meadow fescue	*Festuca pratensis*	MD, VA
Fig buttercup	*Ficaria verna*	All
Fennel	*Foeniculum vulgare*	VA
Glossy buckthorn	*Frangula alnus*	MD, PA, VA

(continued)

Table 6.4 (continued)

Common name	Scientific name	State reported invasive
Goatsrue	*Galega officinalis*	PA
Smallflower galinsoga	*Galinsoga parviflora*	MD
Hairy galinsoga	*Galinsoga quadriradiata*	MD
Smooth bedstraw	*Galium mollugo*	VA
Longstalk cranesbill	*Geranium columbinum*	MD
Ground ivy	*Glechoma hederacea*	MD, PA, VA, WV
English ivy	*Hedera helix*	All
Tawny daylily	*Hemerocallis fulva*	All
Yellow daylily	*Hemerocallis lilioasphodelus*	MD, VA
Gaint hogweed	*Heracleum mantegazzianum*	PA
Dames rocket	*Hesperis matronalis*	MD, PA, VA, WV
Rose of Sharon	*Hibiscus syriacus*	PA, VA
Common velvetgrass	*Holcus lanatus*	VA
Japanese hop	*Humulus japonicus*	DE, MD, PA, VA
Hydrilla	*Hydrilla verticillata*	DE, MD, VA
Hairy catsear	*Hypochaeris radicata*	VA
English holly	*Ilex aquifolium*	MD
Japanese holly	*Ilex crenata*	VA
Cogongrass	*Imperata cylindrica*	VA
Ivyleaf morning glory	*Ipomoea hederacea*	VA
Pitted morning glory	*Ipomoea lacunosa*	MD
Tall morning glory	*Ipomoea purpurea*	VA
Yellowflag iris	*Iris pseudacorus*	DE, MD, VA, WV
Dyer's woad	*Isatis tinctoria*	VA
Japanese rose	*Kerria japonica*	VA
Crape myrtle	*Lagerstroemia indica*	VA
Henbit	*Lamium amplexicaule*	MD, WV
Spotted deadnettle	*Lamium maculatum*	MD
Purple deadnettle	*Lamium purpureum*	MD
Weeping lantana	*Lantana montevidensis*	VA
Nipplewort	*Lapsana communis*	VA
Shrubby lespedeza	*Lespedeza bicolor*	VA
Sericea lespedeza	*Lespedeza cuneata*	MD, VA, WV
Oxeye daisy	*Leucanthemum vulgare*	MD
Amur privet	*Ligustrum amurense*	VA
Border privet	*Ligustrum obtusifolium*	PA, VA
California privet	*Ligustrum ovalifolium*	PA, VA
Chinese privet	*Ligustrum sinense*	MD, VA
European privet	*Ligustrum vulgare*	All
Yellow toadflax	*Linaria vulgaris*	VA, WV
Creeping liriope	*Liriope spicata*	MD
Sweet breathe of spring	*Lonicera fragrantissima*	VA
Japanese honeysuckle	*Lonicera japonica*	All
Amur honeysuckle	*Lonicera maackii*	DE, MD, PA, VA

(continued)

Table 6.4 (continued)

Common name	Scientific name	State reported invasive
Morrow's honeysuckle	*Lonicera morrowii*	All
Standish's honeysuckle	*Lonicera standishii*	PA, VA
Tatarian honeysuckle	*Lonicera tatarica*	DE, MD, PA, VA
Bell's honeysuckle	*Lonicera × bella*	MD, PA, VA
Birdsfoot trefoil	*Lotus corniculatus*	VA
Moneywort	*Lysimachia nummularia*	MD, PA, VA, WV
Purple loosestrife	*Lythrum salicaria*	DE, MD, PA, VA
European wand loosestrife	*Lythrum virgatum*	VA
Osage orange	*Maclura pomifera*	MD, WV
Paradise apple	*Malus pumila*	MD, PA, VA, WV
Black medic	*Medicago lupulina*	VA
Chinaberry	*Melia azedarach*	VA
Yellow sweetclover	*Melilotus officinalis*	MD, VA, WV
Nepalese browntop	*Microstegium vimineum*	All
Chinese silvergrass	*Miscanthus sinensis*	MD, PA, VA, WV
White mulberry	*Morus alba*	MD, PA, VA
Marsh dayflower	*Murdannia keisak*	VA
Common grape hyacinth	*Muscari botryoides*	MD, WV
Water starwort	*Myosoton aquaticum*	MD, PA, VA
Parrotfeather	*Myriophyllum aquaticum*	DE, MD, VA
Eurasian watermilfoil	*Myriophyllum spicatum*	DE, PA, VA
Catnip	*Nepeta cataria*	MD
Wavyleaf basketgrass	*Oplismenus hirtellus* ssp. *undulatifolius*	MD, VA
Star of Bethlehem	*Ornithogalum umbellatum*	MD, PA
Japanese pachysandra	*Pachysandra terminalis*	VA
Dallisgrass	*Paspalum dilatatum*	MD
Wild parsnip	*Pastinaca sativa*	PA, VA
Princess tree	*Paulownia tomentosa*	MD, PA, VA, WV
Perilla mint	*Perilla frutescens*	MD, PA, VA, WV
Oriental lady's thumb	*Persicaria longiseta*	VA, WV
Lady's thumb	*Persicaria maculosa*	MD, VA
Mile-a-minute weed	*Persicaria perfoliata*	All
Reed canary grass	*Phalaris arundinacea*	DE, MD, PA, VA
Amur corktree	*Phellodendron amurense*	PA, VA
Timothy	*Phleum pratense*	MD, VA
Common reed	*Phragmites australis*	DE, MD, PA, VA
Golden bamboo	*Phyllostachys aurea*	MD, PA, VA, WV
Norway spruce	*Picea abies*	MD
Scots pine	*Pinus sylvestris*	PA
Japanese black pine	*Pinus thunbergiana*	DE, VA
Water lettuce	*Pistia stratiotes*	DE
Buckhorn plantain	*Plantago lanceolata*	MD, VA
Broadleaf plantain	*Plantago major*	MD, VA
Canada bluegrass	*Poa compressa*	PA, VA, WV

(continued)

Table 6.4 (continued)

Common name	Scientific name	State reported invasive
Roughstalk bluegrass	*Poa trivialis*	VA
Trifoliate orange	*Poncirus trifoliata*	WV
White popular	*Populus alba*	MD, PA, VA
Sulfur cinquefoil	*Potentilla recta*	MD
Sweet cherry	*Prunus avium*	DE, MD, PA, VA
Sour cherry	*Prunus cerasus*	MD
Mahaleb cherry	*Prunus mahaleb*	PA
European bird cherry	*Prunus padus*	PA
Peach	*Prunus persica*	MD, VA
Arrow bamboo	*Pseudosasa japonica*	MD, PA, WV
Kudzu	*Pueraria montana* var. *lobata*	All
Callery pear (Bradford pear)	*Pyrus calleryana*	MD, PA
Sawtooth oak	*Quercus acutissima*	MD, VA
Tall buttercup	*Ranunculus acris*	MD
Bulbous buttercup	*Ranunculus bulbosus*	MD
Wild radish	*Raphanus raphanistrum*	VA
European buckthorn	*Rhamnus cathartica*	MD, PA, VA
Jetbead	*Rhodotypos scandens*	DE, PA, VA
Bristly locust	*Robinia hispida*	PA
Black locust	*Robinia pseudoacacia*	MD, PA, VA
Maccartney rose	*Rosa bracteata*	VA
Dog rose	*Rosa canina*	MD, PA, VA
Sweetbriar rose	*Rosa eglanteria*	PA, VA
French rose	*Rosa gallica*	PA, VA
Smallflower sweetbriar	*Rosa micrantha*	VA
Multiflora rose	*Rosa multiflora*	All
Rugosa rose	*Rosa rugosa*	PA
Memorial rose	*Rosa wichuraiana*	VA
Himalayan berry	*Rubus bifrons*	VA
Strawberry raspberry	*Rubus illecebrosus*	MD, VA
Cutleaf blackberry	*Rubus laciniatus*	PA, WV
Wine raspberry	*Rubus phoenicolasius*	All
Red sorrel	*Rumex acetosella*	VA, WV
Curly dock	*Rumex crispus* ssp. *Crispus*	VA, WV
White willow	*Salix alba*	VA
Goat willow	*Salix caprea*	PA
Large gray willow	*Salix cinerea*	MD
Crack willow	*Salix fragilis*	PA
Laurel willow	*Salix pentandra*	MD, PA
Purpleosier willow	*Salix purpurea*	PA
Weeping willow	*Salix* × *sepulcralis* [*alba* × *babylonica*]	WV
Russian thistle	*Salsola kali*	VA
Bouncing bet	*Saponaria officinalis*	WV
Crown vetch	*Securigera varia*	MD, VA

(continued)

Table 6.4 (continued)

Common name	Scientific name	State reported invasive
Giant foxtail	*Setaria faberi* Herrm.	PA, VA
Bittersweet nightshade	*Solanum dulcamara*	MD, PA
Black nightshade	*Solanum nigrum*	VA
Sudan grass	*Sorghum bicolor* ssp. *drummondii*	PA
Johnson grass	*Sorghum halepense*	DE, MD, PA
Japanese spiraea	*Spiraea japonica*	MD, PA, VA
Common chickweed	*Stellaria pallida*	MD, PA, VA, WV
Common tansy	*Tanacetum vulgare*	MD
Dandelion	*Taraxacum officinale*	MD, PA, VA
Japanese yew	*Taxus cuspidata*	VA
Meadow salsify	*Tragopogon lamottei*	MD
Water chestnut	*Trapa natans*	DE, PA, VA
Chinese tallowtree	*Triadica sebifera*	VA
Hop clover	*Trifolium aureum*	VA
Large hop clover	*Trifolium campestre*	MD
Small hop clover	*Trifolium dubium*	MD
Red clover	*Trifolium pratense*	MD, VA
White clover	*Trifolium repens*	MD
Chinese elm	*Ulmus parvifolia*	VA
Siberian elm	*Ulmus pumila*	MD, PA, VA, WV
Common mullein	*Verbascum thapsus*	PA, VA, WV
Ivyleaf speedwell	*Veronica hederifolia*	MD, VA, WV
Thymeleaf speedwell	*Veronica serpyllifolia*	MD
Linden viburnum	*Viburnum dilatatum*	VA
European cranberrybush	*Viburnum opulus* var. *opulus*	PA
Siebold's arrowwood	*Viburnum sieboldii*	PA
Big periwinkle	*Vinca major*	MD
Common periwinkle	*Vinca minor*	All
Japanese wisteria	*Wisteria floribunda*	MD, PA, VA
Chinese wisteria	*Wisteria sinensis*	MD, PA, VA
Asiatic hawksbeard	*Youngia japonica*	VA

Data from Mid-Atlantic Exotic Pest Plant Council (2011)

the MAR is high, three of the most noxious wetland invaders are purple loosestrife, water thyme (*Hydrilla verticillata*), and common reed. Each has become widely established and has resulted in acute changes in the region's wetlands. As such they are generally representative of the effects of invasive species in general. A brief description of each species and its ecological impacts follows.

6.6.1.1 Purple Loosestrife (*Lythrum salicaria*)

Purple loosestrife is a Eurasian emergent with bright purple flowers that has formed dense monocultures in many freshwater wetlands of the MAR and other eastern and

Midwestern states. It was introduced to the United States in the early 1800s as a medicinal/horticultural plant, and through contaminated ballast water. It is now found in nearly all of the contiguous United States (USDA, NRCS 2011).

The pattern of invasion for purple loosestrife often involves a period of latency between its introduction to a site and the time when it becomes a troublesome weed. It can grow interspersed among other vegetation for 20–40 years and then proliferate and spread so rapidly that it outcompetes native species (Cronk and Fennessy 2001). This pattern has been seen in major geographic regions, including the Delaware River watershed, Hudson River Valley, and Finger Lakes Region (Stuckey 1980). Its prolific seed production ensures that a seed bank is formed. If a disturbance such as drought or drawdown occurs, seeds are able to germinate and quickly colonize the newly opened area outcompeting other species (Mal et al. 1992). In the MAR, its occurrence is widespread, including the marshes of the Chesapeake and Delaware Bays, and inland into Pennsylvania, Maryland, northern Virginia, and portions of West Virginia. In response, a watershed regional management plan has been developed to aid in its control and minimize its spread (Chesapeake Bay Program Office 2004).

Establishment of purple loosestrife is correlated with decreased biodiversity and habitat quality, and declines in the abundance of some endangered species such as bog turtle and dwarf spikerush (*Eleocharis parvula*) (Mullin 1998). The decline in abundance of some waterfowl, such as the black tern, least bittern, American bittern, and Virginia rail, is also attributed to its spread as dense stands of loosestrife reduce or exclude other emergent wetland species whose seeds serve as important food sources for these marsh birds (Chesapeake Bay Program Office 2004).

6.6.1.2 Water Thyme (*Hydrilla verticillata*)

Although the origins of water thyme are not clear, it is believed to be a native of Southeast Asia or Africa. Its current distribution is essentially global with populations on all continents except Antarctica. Water thyme is a noxious invasive in the Atlantic coast states, and is considered an invasive in Europe, Central America, and elsewhere (Steward 1993). Its leafy, branching stems commonly reach lengths of 2 m and can be as long as 7.5 m (Langeland 1996). In the MAR, it is concentrated in the Potomac River Basin, the Nanticoke, and much of eastern Pennsylvania. Two distinct biotypes exist, a dioecious form that dominates southern states and a monoecious form that dominates in the MAR. The first reports of the monoecious type were in the Delaware River in 1976 and Potomac River near Washington, DC in 1982 (Madeira et al. 2000). Rates of sexual reproduction are higher in the monoecious biotype, allowing for genetic diversification and adaptation to different habitats (Steward et al. 1984). Vegetative reproduction can be prolific through specialized axillary buds known as turions.

Hydrilla grows rapidly and can swiftly fill a wetland basin, displacing native species as it does so. This can lead to a host of effects including changes in water chemistry (including oxygen depletion as it decomposes), declines in zooplankton populations, and where fish are present, altered fish community structure (Gu 2006). Its dense stands also interfere with human uses of water such as drainage, irrigation, navigation, and recreation (Langeland 1996). Surprisingly, benefits of invasion have also been reported, including its ability to provide spawning habitat for some fish species.

6.6.1.3 Common Reed (*Phragmites australis*)

Common reed is found throughout the United States and north to Canada. The categorization of it as an invasive species is relatively recent, and stems from genetic evidence that a more aggressive biotype was introduced to the Atlantic coast sometime last century. The expansion of *Phragmites* in coastal areas of the MAR is thought to be by this invasive biotype (as reviewed in Rice et al. 2000).

Phragmites tolerates salt levels up to about half the concentration of seawater, so unless the salt-tolerant haplotype is involved, its distribution is restricted to freshwater and brackish environments (Vasquez et al. 2005). Hydrologic alterations in salt marshes that lead to tidal restrictions and salinity decreases allow the less salt-tolerant *Phragmites* to compete with true halophytes. In some locations, such as the Hackensack Meadowlands and the salt marshes of Delaware Bay, *Phragmites* forms dense monospecific stands, displacing the once dominant saltmarsh cordgrass (*Spartina alterniflora*).

Once *Phragmites* dominates a marsh, there are notable differences in the physical environment. It has a high evapotranspiration rate, which, in turn, can have a profound effect on the water table. In many marshes where *Phragmites* is dominant, lower water tables lead to peat compaction and lowered marsh surface elevations (Cronk and Fennessy 2001). *Phragmites* can be controlled in tidally restricted salt marshes by restoring the natural hydrology (Sinicrope et al. 1990). In some instances, increasing salinity levels by restoring tidal exchange has been shown to be an effective control measure.

6.7 Threatened and Endangered Plants in the MAR

Since the early part of the seventeenth century, the MAR has been subjected to intensive human influence. In the 400 years since Europeans first colonized the region, many habitats have been isolated, reduced in size, degraded, and destroyed and as a result many species dependent upon these habitats are now rare, endangered, or extirpated. Inland freshwater wetlands have been especially vulnerable, as these wetlands were easily converted to agriculture and urban development. In fact, the MAR supports one of the fastest growing regions in the country. Called the urban crescent, this area of concentrated development stretches from Baltimore south to Richmond and east to Norfolk (Watts and Bradshaw 2005).

In the mid-1970s, the US Fish and Wildlife Service (FWS) and the National Oceanic and Atmospheric Association (NOAA) began listing plants as either threatened or endangered under the Endangered Species Act. State Natural Heritage

Programs compile a similar list of statewide conservation species with ranks following those developed by NatureServe (Faber-Langendoen et al. 2009). State conservation ranks are divided into three main categories:

S1: *Critically imperiled*—Critically imperiled in the jurisdiction because of extreme rarity or because of some factor(s) such as very steep declines making it especially vulnerable to extirpation from the jurisdiction

S2: *Imperiled*—Imperiled in the jurisdiction because of rarity due to very restricted range, very few populations or occurrences, steep declines, or other factors making it very vulnerable to extirpation from the jurisdiction

S3: *Vulnerable*—Vulnerable in the jurisdiction due to a restricted range, relatively few populations or occurrences, recent and widespread declines, or other factors making it vulnerable to extirpation

The most recent conservation inventories by state Natural Heritage programs indicate there are 1,712 unique taxa in the MAR that are designated as rare, threatened, or endangered (Appendix). When examined by state, Maryland has the most listed species and West Virginia the least (Table 6.5). Virginia, however, has the highest number of critically imperiled (S1) species of all states in the region.

Most ranked species in the MAR (1,009 or 59%) are wetland plants. The high proportion of wetland to non-wetland species is also mirrored by conservation rankings by individual states (Table 6.5). In West Virginia alone, wetlands cover only 1% of the land surface yet they provide essential habitat for 44% of the state's rare flora (Byers et al. 2007). The coastal states of Maryland, Delaware, and Virginia have the highest numbers of wetland species with conservation status, respectively, and overwhelmingly, these species associate with freshwater, nontidal wetlands. In Delaware over 50% of all listed wetland species occur in nontidal freshwater wetlands while only 7% are associated with brackish or saline tidal marshes (McAvoy 2010). Sea-level fens, nutrient poor, groundwater-fed depressions that occur in coastal areas from Virginia to Massachusetts, are one example of freshwater nontidal freshwater wetlands that support several rare plant species including beaked spikerush (*Eleocharis rostellata*), ten-angled pipewort (*Eriocaulon decangulare* var. *decangulare*), brown-fruited rush (*Juncus pelocarpus*), and white beakrush (*Rhynchospora alba*). These unique habitats are threatened by salt water intrusion due to chronic sea level rise (Virginia Department of Conservation and Recreation 2012).

In the western MAR, bogs, seeps, calcareous fens, and other high elevation wetlands are scattered throughout the Ridge and Valley from Pennsylvania to Virginia. In a study of these wetland communities in the Allegheny Mountains of West Virginia, Byers et al. (2007) identified 145 state rare plant species representing 31% of West Virginia's rare flora, including 60 ranked as critically imperiled, 56 as imperiled and 29 as vulnerable. Many of these wetlands, like the Cranberry Glades, are home to more northern-affiliated species occurring at the southern end of their range like the critically imperiled bog rosemary (*Andromeda polifolia* var. *glaucophylla*) and buckbean (*Menyanthes trifoliata*), as well as more southern-affiliated species at the northernmost extent of their range including glade spurge (*Euphorbia*

Table 6.5 Summary of state listed vascular wetland (FAC, FACW, or OBL) plant species for the mid-Atlantic region[a]

	State rank[b]			
	S1	S2	S3	% Wetland
DE	139	119	80	63
MD	205	69	98	59
PA	186	74	49	57
VA	229	93	0	61
WV	127	80	31	53

The percentage of the total number of listed species that are also wetland plants is indicated by state

[a]Where indicator status was not available, species were designated as wetland plants using habitat information; where two rankings were given, the species was tallied under the most conservative rank

[b]Ranking categories are from Faber-Langendoen et al. (2009)

purpurea), Monongahela Barbara's buttons (*Marshallia grandiflora*), and long-stalked holly (*Ilex collina*).

As with wetlands in the Coastal Plain, high elevation wetlands of the MAR are subject to a host of threats (Byers et al. 2007). Road construction and logging have fragmented the forest. Mining activities have altered hydrology and urban and suburban developments have introduced non-native and invasive species. In addition to local stressors, these wetlands are threatened by acid deposition and climate change. With so much floral diversity, genetic wealth and natural heritage at stake, we would do well to heed the words of Byers et al. (2007) who unequivocally state, "We cannot afford to be complacent about the survival of our beautiful and ecologically rich…wetlands." Indeed, to lose these species would not only be devastating to those states entrusted to protect this natural legacy, but an irreplaceable loss to the region as a whole.

6.8 Appendix: Conservation Status Ranks for Vascular Plant Species Occurring in States within the Mid-Atlantic Region

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Abies balsamea	Balsam fir	Tree		S1		S1	S3
Abies fraseri	Fraser fir	Tree				S1	
Acer saccharum var. *saccharum*	Sugar maple	Tree	S3				
Aconitum reclinatum	White monkshood	Forb			S1		S3
Aconitum uncinatum	Blue monkshood	Forb		S1	S2		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Acorus americanus	Sweet flag	Forb			S1		
Actaea pachypoda	White baneberry	Forb	S1				
Actaea podocarpa	American bugbane	Forb		S2	S3		
Adiantum pedatum	Maidenhair fern	Cryptogam	S3				
Adlumia fungosa	Climbing fumitory	Vine		S2			
Aeschynomene virginica	Sensitive joint-vetch	Forb		S1		S2	
Agalinis acuta	Sandplain gerardia	Forb		S1			
Agalinis auriculata	Auricled gerardia	Forb		S1	S1	S1	
Agalinis maritima var. *maritima*	Salt marsh false foxglove	Forb	S1				
Agalinis obtusifolia	Blunt-leaved gerardia	Forb		S1			
Agalinis paupercula	Small-flowered false foxglove	Forb			S1	S1	
Agalinis setacea	Thread-leaved false foxglove	Forb	S1	S1			
Agalinis skinneriana	Midwestern gerardia	Forb		S1			
Agalinis tenuifolia var. *tenuifolia*	Slender false foxglove	Forb	S1				
Agastache nepetoides	Yellow giant hyssop	Forb	S2				
Agastache scrophulariifolia	Purple giant hyssop	Forb		S1S2			
Ageratina aromatica (var. *aromatica*)	Lesser snakeroot	Forb			S3		S1
Agrimonia gryposepala	Tall hairy agrimony	Forb	S3				
Agrimonia microcarpa	Small-fruited agrimony	Forb					S1
Agrimonia pubescens	Downy agrimony	Forb	S3				
Agrimonia rostellata	Woodland agrimony	Forb	S3				
Agrimonia striata	Woodland agrimony	Forb	S2	S1			
Agrostis mertensii	Arctic bentgrass	Graminoid					S1
Aletris aurea	Golden colic root	Forb				S1	
Aletris farinosa	Colic root	Forb	S3		S1		
Alisma triviale	Northern water plantain	Forb			S1		
Allium oxyphilum	Nodding onion	Forb					S2

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Allium tricoccum	Wild leek	Forb	S3				
Alnus incana subsp. *rugosa*	Speckled alder	Tree/Shrub				S2	
Alnus maritima (subsp. *maritima*)	Seaside alder	Tree/Shrub	S3	S3.1			
Alnus viridis	Mountain alder	Tree/Shrub			S1		
Alopecurus aequalis	Short-awn foxtail	Graminoid			S3		
Amaranthus cannabinus	Waterhemp ragweed	Forb			S3		
Amaranthus pumilus	Seabeach amaranth	Forb	S1	S1		S1	
Amelanchier bartramiana	Oblong-fruited serviceberry	Tree			S1		S2
Amelanchier canadensis	Serviceberry	Tree			S1		
Amelanchier humilis	Running serviceberry	Tree		S1	S1		
Amelanchier nantucketensis	Nantucket shadbush	Tree		S1		S1	
Amelanchier obovalis	Coastal juneberry	Tree			S1		
Amelanchier sanguinea	Round-leaf serviceberry	Tree		S1	S2		
Amelanchier stolonifera	Running juneberry	Tree		S2			
Amianthium muscitoxicum	Fly-poison	Forb	S2				
Ammannia coccinea	Scarlet ammannia	Forb			S2		
Ammannia latifolia	Koehne's ammannia	Forb		S2			
Ammophila breviligulata	American beachgrass	Graminoid			S2		
Amorpha fruticosa	False indigo-bush	Shrub					S2
Ampelopsis cordata	Heartleaf peppervine	Vine					S1
Amphicarpum amphicarpon	Peanut grass	Graminoid	S2			S1	
Amphicarpum purshii	Pursh's amphicarpum	Graminoid		S3			
Anaphalis margaritacea	Pearly everlasting	Forb		S3		S1	
Andromeda polifolia	Bog rosemary	Shrub			S3		
Andromeda polifolia var. *glaucophylla*	Bog rosemary	Shrub					S1
Andropogon glomeratus (var. *glomeratus*)	Broomsedge	Graminoid			S3		S2

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Andropogon glomeratus var. *hirsutior*	Southern bushy broom-sedge	Graminoid	S1.1				
Andropogon gyrans	Elliott's beardgrass	Graminoid			S3		
Anemone americana	Roundlobed hepatica	Forb	S3				
Anemone berlandieri	Eastern prairie anemone	Forb				S1	
Anemone canadensis	Canada anemone	Forb				S1	S1
Anemone cylindrica	Long-fruited anemone	Forb			S1		
Anemone quinquefolia var. *minima*	Dwarf anemone	Forb					S2
Angelica atropurpurea	Great angelica	Forb	S1.1				
Angelica triquinata	Filmy angelica	Forb		S1			
Angelica venenosa	hairy angelica	Forb	S2				
Antennaria solitaria	Single-headed pussytoes	Forb		S2	S1		
Antennaria virginica	Shale barren pussytoes	Forb			S3		
Anthoxanthum hirtum	Vanilla grass	Graminoid	S1				
Aplectrum hyemale	Puttyroot	Forb	S1		S3		
Apocynum androsaemifolium	Spreading dogbane	Forb	S1				
Aquilegia canadensis	Wild columbine	Forb	S2				
Arabidopsis lyrata	Lyre-leaf rockcress	Forb	S1				
Arabis glabra	Tower-mustard	Forb				S1	
Arabis hirsuta	Western hairy rockcress	Forb			S1		
Arabis hirsuta var. *adpressipilis*	Hairy rockcress	Forb				S1S2	
Arabis hirsuta var. *pycnocarpa*	Hairy rockcress	Forb					S2
Arabis missouriensis	Missouri rockcress	Forb		S1	S1		
Arabis patens	Spreading rockcress	Forb		S3	S2	S2	S2
Arabis serotina	Shale barren rockcress	Forb				S2	S2
Arabis shortii	Short's rockcress	Forb		S3		S2	S1
Aralia hispida	Bristly sarsaparilla	#N/A		S1		S2	
Aralia racemosa	American spikenard	#N/A	S3				
Arceuthobium pusillum	Dwarf mistletoe	#N/A			S2		
Arctostaphylos uva-ursi	Bearberry	Shrub		S1		S1	
Arethusa bulbosa	Swamp pink	Forb			S1	S1	

(continued)

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Arisaema dracontium	green dragon	Forb	S2				
Aristida dichotoma var. *curtissii*	Three-awned grass	Graminoid			S1S2		
Aristida lanosa	Woolly three-awn	Graminoid		S1			
Aristida longespica var. *longespica*	Slender three-awn	Graminoid			S3S4		
Aristida purpurascens (var. *purpurascens*)	Purple needlegrass	Graminoid			S2		S1
Aristida tuberculosa	Sea-beach three-awn	Graminoid		S1			
Aristida virgata	Wand-like three-awn grass	Graminoid	S1	S1			
Aristolochia macrophylla	Pipevine	Vine		S1			
Armoracia lacustris	Lake cress	Forb		S1			
Arnica acaulis	Leopard's bane	Forb		S1	S1		
Arnoglossum atriplicifolium	Pale Indian-plantain	Forb	S2				
Arnoglossum muehlenbergii	Great Indian-plantain	Forb				S2	
Arnoglossum reniforme	Great Indian-plantain	Forb			S1		
Artemisia campestris ssp. *caudata*	Beach wormwood	Forb			S1		
Arundinaria gigantea (ssp. *gigantea*)	Giant cane	Graminoid		S2			S2
Asclepias amplexicaulis	Clasping milkweed	Forb	S3				
Asclepias exaltata	Poke milkweed	Forb	S2				
Asclepias hirtella	Green milkweed	Forb					S2
Asclepias lanceolata	Lance-leaf orange milkweed	Forb	S1				
Asclepias longifolia	Long-leaf milkweed	Forb				S1	
Asclepias purpurascens	Purple milkweed	Forb	S2			S2	
Asclepias rubra	Red milkweed	Forb	S1	S1		S2	
Asclepias tuberosa ssp. *rolfsii*	Sandhills butterflyweed	Forb				S1	
Asclepias variegata	White milkweed	Forb	S2		S1		
Asclepias verticillata	Whorled milkweed	Forb		S3			
Asclepias viridiflora	Green milkweed	Forb	S3				
Asclepias viridis	Spider milkweed	Forb					S1

(continued)

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Asplenium bradleyi	Bradley's spleenwort	Cryptogam			S1	S2	
Asplenium pinnatifidum	Lobed spleenwort	Cryptogam		S1	S3		
Asplenium resiliens	Black-stem spleenwort	Cryptogam		S1	S1		
Asplenium ruta-muraria	Wall-rue	Cryptogam		S3			
Asplenium septentrionale	Northern spleenwort	Cryptogam					S2
Asplenium trichomanes subsp. *trichomanes*	Maidenhair spleenwort	Cryptogam	S1.1				
Asplenium x alternifolium	Spleenwort	Cryptogam					S1
Astragalus canadensis	Canada milkvetch	Forb		S1	S2		
Astragalus distortus	Bent milkvetch	Forb		S2			S2
Astragalus neglectus	Cooper's milkvetch	Forb			S1	S2	S1
Astranthium integrifolium ssp. *integrifolium*	Western daisy	Forb					S1
Atriplex arenaria	Sea-beach orach	Forb		S3			
Atriplex mucronata	Crested saltbush	Shrub	S3				
Aureolaria flava (var. *flava*)	Yellow false-foxglove	Forb	S1	S3			
Aureolaria pedicularia var. *pedicularia*	Fernleaf yellow false foxglove	Forb	S1				
Baccharis halimifolia	Eastern baccharis	Shrub			S3		
Bacopa innominata	Tropical waterhyssop	#N/A				S2	
Bacopa rotundifolia	Roundleaf waterhyssop	#N/A				S1	
Baptisia albescens	Narrowpod white wild indigo	Forb				S1	
Baptisia australis (var. *australis*)	Wild blue indigo	Forb		S2	S2		S3
Bartonia paniculata (subsp. *paniculata*)	Twining bartonia	Vine	S2	S3	S3		
Bartonia verna	Spring bartonia	Forb				S1	
Berberis canadensis	American barberry	Shrub					S1
Betula cordifolia	Mountain paper birch	Tree				S2	
Betula papyrifera	Paper birch	Tree					S2
Betula populifolia	Gray birch	Tree	S2			S1	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Betula uber	Virginia round-leaf birch	Tree				S1	
Bidens bidentoides	Swamp beggar-ticks	Forb			S1		
Bidens bidentoides var. *mariana*	Maryland bur-marigold	Forb		S3.1			
Bidens coronata	Tickseed sunflower	Forb		S2S3			
Bidens discoidea	Small beggar-ticks	Forb			S3		
Bidens laevis	Beggar-ticks	Forb			S1		
Bidens mitis	Small-fruited beggar-ticks	Forb	S2	S1			
Bidens trichosperma	Northern tick-seed sunflower	Forb	S3				
Blephilia ciliata	Downy woodmint	Forb		S3			
Blephilia hirsuta	Hairy woodmint	Forb		S2			
Boechera canadensis	Sicklepod	Forb	S2				
Boechera laevigata	Smooth rockcress	Forb	S1				
Bolboschoenus novae-angliae	Brackish bulrush	Graminoid	S1				
Boltonia asteroides	Aster-like boltonia	Forb		S1	S1		
Boltonia asteroides var. *glastifolia*	Aster-like boltonia	Forb	S2				
Boltonia montana	Valley doll's-daisy	Forb				S1	
Botrychium jenmanii	Alabama grapefern	Cryptogam				S1	
Botrychium lanceolatum var. *angustisegmentum*	Lance-leaf grape-fern	Cryptogam				S1	S1
Botrychium matricariifolium	Daisy-leaf moonwort	Cryptogam	S1				S2
Botrychium multifidum	Leathery grapefern	Cryptogam				S1	
Botrychium oneidense	Blunt-lobe grape-fern	Cryptogam		S1		S2	S3
Botrychium simplex	Least grape-fern	Cryptogam				S1	
Bouteloua curtipendula (var. *curtipendula*)	Side-oats grama	Graminoid		S2	S2		S3
Brachyelytrum erectum	Bearded short-husk grass	Graminoid	S3				

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Bromus ciliatus	Fringed brome grass	Graminoid				S1	
Bromus kalmii	Wild chess	Graminoid		S1	S3	S1	
Bromus latiglumis	Broad-glumed brome	Graminoid	S1.1	S1			
Bromus nottowayanus	Nottoway's brome	Graminoid		S1S2			
Bromus pubescens	Hairy wood brome	Graminoid	S2				
Buchnera americana	Blue-hearts	Forb				S1S2	
Buckleya distichophylla	Piratebush	Shrub				S2	
Cabomba caroliniana	Carolina fanwort	Forb				S2	
Cakile edentula	American Sea-rocket	Forb			S3		
Calamagrostis canadensis var. *canadensis*	Blue-joint reedgrass	Graminoid	S1				
Calamagrostis porteri (ssp. *porteri*)	Porter's reedgrass	Graminoid		S1			S2
Calamagrostis stricta ssp. *stricta* var. *stricta*	Northern reedgrass	Graminoid					S1
Calamovilfa brevipilis	Pine barrens reedgrass	Graminoid				S1	
Calla palustris	Wild calla	Forb		S1			
Callitriche terrestris	Pond water-starwort	Forb	S3				
Calopogon pallidus	Pale grass-pink	Forb				S1	
Calopogon tuberosus (var. *tuberosus*)	Tuberous grasspink	Forb	S1	S1		S2	S1
Caltha palustris	Marsh marigold	Forb	S2				
Calystegia spithamaea ssp. *purshiana*	Shale barren bindweed	Forb					S3
Calystegia spithamaea (subsp. *spithamaea*)	Low bindweed	Forb	S1	S2			
Camassia scilloides	Wild hyacinth	Forb			S1	S2	
Campanula aparinoides	Marsh bellflower	Forb	S2				
Campanula rotundifolia	Harebell	Forb		S2		S1	S3
Cardamine angustata	Slender toothwort	Forb	S2				
Cardamine clematitis	Mountain bittercress	Forb				S1	
Cardamine dissecta	Divided-leaved toothwort	Forb				S1	
Cardamine douglassii	Purple cress	Forb		S3			
Cardamine flagellifera	Bitter cress	Forb					S2
Cardamine longii	Long's bittercress	Forb	S2	S1			
Cardamine maxima	Large toothwort	Forb			S2		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Cardamine micranthera	Small-anthered bittercress	Forb				S1	
Cardamine parviflora var. *arenicola*	Small-flower bittercress	Forb	S1				
Cardamine pratensis	Cuckooflower	Forb		S1		S1	
Cardamine pratensis var. *palustris*	Cuckooflower	Forb			S1		
Cardamine rotundifolia	Roundleaf bittercress	Forb	S1.1	S3			
Carex aestivalis	Summer sedge	Graminoid		S1			S2
Carex aggregata	Glomerate sedge	Graminoid					S2
Carex alata	Broad-winged sedge	Graminoid			S2		
Carex albursina	Sedge	Graminoid		S3			
Carex appalachica	Appalachian sedge	Graminoid					S2
Carex aquatilis (var. *aquatilis*)	Water sedge	Graminoid		S1	S2	S1	S1
Carex arctata	Black sedge	Graminoid				S1	S1
Carex argyrantha	Hay sedge	Graminoid		S3			
Carex atherodes	Awned sedge	Graminoid			S1	S1	S1
Carex aurea	Golden-fruited sedge	Graminoid			S1		
Carex barrattii	Barratt's sedge	Graminoid	S3	S3		S2	
Carex bebbii	Bebb's sedge	Graminoid			S1	S1	
Carex bicknellii	Bicknell's sedge	Graminoid			S1		
Carex bromoides (subsp. *bromoides*)	Brome-like sedge	Graminoid	S2				
Carex brunnescens	Brownish sedge	Graminoid		S3			
Carex bullata	Bull sedge	Graminoid	S3	S3	S1		
Carex bushii	Bush's sedge	Graminoid	S2				S2S3
Carex buxbaumii	Buxbaum's sedge	Graminoid	S1.1	S2	S3	S2	S2
Carex canescens	Hoary sedge	Graminoid					S3
Carex careyana	Carey's sedge	Graminoid		S1	S1		S1
Carex collinsii	Collin's sedge	Graminoid	S3		S2		
Carex communis (var. *communis*)	Fibrous-root sedge	Graminoid	S2				
Carex comosa	Bearded sedge	Graminoid					S2
Carex conjuncta	Soft fox sedge	Graminoid	S3				
Carex conoidea	Field sedge	Graminoid	S1.1	S1		S1S2	S1
Carex crawei	Craw's sedge	Graminoid				S2	
Carex crawfordii	Crawford's sedge	Graminoid			S1		
Carex crinita var. *brevicrinis*	Short Hair Sedge	Graminoid			S1		
Carex cristatella	Crested sedge	Graminoid	S1.1			S2	
Carex crus-corvi	Ravenfoot sedge	Graminoid				S1S2	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Carex cryptolepis	Northeastern sedge	Graminoid			S1		
Carex cumberlandensis	Sedge	Graminoid					S2
Carex davisii	Davis' sedge	Graminoid	S1	S1		S1	S1
Carex debilis var. *pubera*	Downy white-edge sedge	Graminoid					S1
Carex decomposita	Cypress-knee sedge	Graminoid	S1	S1		S2	
Carex deflexa	Short-stemmed sedge	Graminoid					S1
Carex diandra	Lesser panicled sedge	Graminoid		S1	S2		
Carex disperma	Soft-leaved sedge	Graminoid			S3		
Carex eburnea	Ebony sedge	Graminoid		S1	S1		S3
Carex echinata	Little prickly sedge	Graminoid		S3			
Carex emoryi	Emory's sedge	Graminoid	S1	S3			S2
Carex exilis	Coast sedge	Graminoid	S1	S1			
Carex flava	Yellow sedge	Graminoid			S2	S1	
Carex foenea	Sedge	Graminoid			S1		
Carex formosa	Handsome sedge	Graminoid			S1		
Carex garberi	Elk sedge	Graminoid			S1		
Carex geyeri	Geyer's sedge	Graminoid			S1		
Carex gigantea	Giant sedge	Graminoid	S3	S3			
Carex glaucescens	Sedge	Graminoid		S1			
Carex gracilescens	Slender sedge	Graminoid	S2				
Carex gracillima	Graceful sedge	Graminoid	S3				
Carex granularis	Meadow sedge	Graminoid	S3				
Carex grisea	Inflated narrowleaf sedge	Graminoid	S1.1				
Carex gynandra	Nodding sedge	Graminoid	S2				
Carex haydenii	Cloud sedge	Graminoid		S1	S1S2		S1
Carex hirtifolia	Pubescent sedge	Graminoid	S3	S3			S2
Carex hitchcockiana	Hitchcock's sedge	Graminoid		S1			
Carex hormathodes	marsh straw sedge	Graminoid	S3				
Carex hyalinolepis	Shoreline sedge	Graminoid		S2S3			
Carex hystericina	Porcupine sedge	Graminoid		S1			
Carex interior	Inland sedge	Graminoid		S1		S1	S1
Carex jamesii	James's sedge	Graminoid	S1.1				
Carex joorii	Cypress-swamp sedge	Graminoid	S2	S3			
Carex juniperorum	Juniper sedge	Graminoid				S1	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Carex lacustris	River-bank sedge	Graminoid	S1	S2		S1	S2
Carex lasiocarpa	Hairy-fruited sedge	Graminoid		S1	S3		
Carex lasiocarpa var. *americana*	Slender sedge	Graminoid				S1	S1
Carex laxiculmis var. *copulata*	Spreading sedge	Graminoid					S1
Carex leptalea subsp. *harperi*	Coastal Plain bristly-stalk sedge	Graminoid	S3				
Carex leptalea (subsp. *leptalea*)	Piedmont bristly-stalk sedge	Graminoid	S1				
Carex limosa	Mud sedge	Graminoid			S2		
Carex longii	Long's sedge	Graminoid			S2S3		
Carex louisianica	Louisiana sedge	Graminoid		S3			
Carex lucorum	Sedge	Graminoid		S1			
Carex lucorum var. *austrolucorum*	Sedge	Graminoid					S1
Carex lupuliformis	False hop sedge	Graminoid	S2	S2	S1	S2	S1
Carex manhartii	Manhart sedge	Graminoid				S1	S1
Carex meadii	Mead's Sedge	Graminoid		S1	S1		S1
Carex mesochorea	Midland sedge	Graminoid					S2
Carex mitchelliana	Mitchell's sedge	Graminoid	S2	S2	S1		
Carex molesta	Troublesome sedge	Graminoid					S3
Carex nigromarginata	Black-edge sedge	Graminoid					S3
Carex normalis	Larger straw sedge	Graminoid					S3
Carex oblita	dark green sedge	Graminoid	S2				
Carex oklahomensis	Sooner sedge	Graminoid				S1	
Carex oligocarpa	Eastern few-fruit sedge	Graminoid	S1.1				
Carex oligosperma (var. *oligosperma*)	Few-seed sedge	Graminoid			S2		S1
Carex ormostachya	Spike sedge	Graminoid			S2	S1	
Carex pallescens	Pale sedge	Graminoid				S1	
Carex pauciflora	Few-flowered sedge	Graminoid			S1		S1
Carex paupercula	Bog sedge	Graminoid			S3		
Carex pedunculata	Long-stalked sedge	Graminoid		S1			S2
Carex pellita	Woolly sedge	Graminoid	S2				S2
Carex planispicata	Sedge	Graminoid	S2	S1S2			
Carex polymorpha	Variable sedge	Graminoid			S2	S2	S1
Carex prairea	Prairie sedge	Graminoid			S2	S1	S1

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Carex projecta	Necklace sedge	Graminoid		S2			S3
Carex pseudocyperus	Cyperus-like sedge	Graminoid			S1		
Carex purpurifera	Purple sedge	Graminoid					S1
Carex retroflexa	Reflexed sedge	Graminoid	S1				
Carex retrorsa	Backward sedge	Graminoid			S1		
Carex reznicekii	Reznicek's sedge	Graminoid	S3				
Carex richardsonii	Richardson's sedge	Graminoid		S1	S1		
Carex roanensis	Roan Mountain sedge	Graminoid			S1	S2	S1
Carex scabrata	Rough sedge	Graminoid	S1				
Carex schweinitzii	Schweinitz's sedge	Graminoid			S1	S1	
Carex seorsa	Weak stellate sedge	Graminoid					S1
Carex shortiana	Short's Sedge	Graminoid		S2	S3		
Carex siccata	Sedge	Graminoid			S2		
Carex silicea	Sea-beach sedge	Graminoid	S2	S1		S1	
Carex sparganioides	Burr-reed sedge	Graminoid	S2	S1S2			
Carex sprengelii	Sedge	Graminoid			S3		
Carex squarrosa	Squarrose sedge	Graminoid	S3				
Carex sterilis	Sterile sedge	Graminoid			S1	S1	
Carex straminea	Straw sedge	Graminoid	S1	S1S2		S1	
Carex striatula	Lined sedge	Graminoid	S3	S3			
Carex styloflexa	Bent sedge	Graminoid					S1
Carex suberecta	Prairie straw sedge	Graminoid					S1
Carex tenera	Quill sedge	Graminoid					S1
Carex tetanica	Rigid sedge	Graminoid			S2		S1
Carex tetanica var. *canbyi*	Sedge	Graminoid	S1.1				
Carex tonsa var. *rugosperma*	Shaved sedge	Graminoid					S2S3
Carex torta	Twisted sedge	Graminoid	S2				
Carex trichocarpa	Hairy-fruit sedge	Graminoid	S2	S2			S1
Carex tuckermanii	Tuckerman sedge	Graminoid		S1			S1
Carex typhina	Cattail sedge	Graminoid	S3		S2		S2
Carex utriculata	Sedge	Graminoid				S1	S3
Carex venusta	Dark green sedge	Graminoid		S2			
Carex vesicaria	Inflated sedge	Graminoid	S1	S1		S1S2	S2
Carex vestita	Velvety sedge	Graminoid	S2	S2		S2	
Carex viridula	Green sedge	Graminoid			S1		
Carex wiegandii	Wiegands sedge	Graminoid			S1		
Carex willdenowii	Willdenow's sedge	Graminoid	S1				
Carex woodii	Pretty sedge	Graminoid					S2
Carex x aestivaliformis	Sedge	Graminoid				S1	S1

(continued)

Scientific name	Common name	Life form	State conservation status rank[a] DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Carphephorus bellidifolius	Sandy-woods chaffhead	#N/A				S1	
Carphephorus tomentosus	Woolly chaffhead	#N/A				S1	
Carya carolinae-septentrionalis	Southern shagbark hickory	Tree				S1	
Carya laciniosa	Big shellbark hickory	Tree		S1	S3S4		
Carya ovata	Shagbark hickory	Tree	S3				
Cassia marilandica	Maryland senna	Shrub		S3			
Castanea dentata	American chestnut	Tree		S2S3			
Castanea pumila	Allegheny chinquapin	Tree	S3				
Castilleja coccinea	Indian paintbrush	Forb		S1	S2		
Caulophyllum thalictroides	Blue cohosh	Forb	S3				
Ceanothus americanus	New Jersey tea	Shrub	S2				
Ceanothus herbaceus	Prairie redroot	Shrub					S1
Centella erecta	Erect coinleaf	Forb	S2	S3			
Centrosema virginianum	Spurred butterfly-pea	Forb		S2			
Cerastium nutans	Nodding chickweed	Forb	S1				
Cerastium velutinum (var. *velutinum*)	Field chickweed	Forb	S1.1		S3		
Cerastium velutinum var. *villosissimum*	Goat hill chickweed	Forb			S1		
Ceratophyllum echinatum	Prickly hornwort	Forb		S1			S1
Chaerophyllum procumbens (var. *procumbens*)	Spreading chervil	Forb	S2				
Chamaecrista fasciculata var. *macrosperma*	Marsh wild senna	#N/A		S1			
Chamaecyparis thyoides	Atlantic white cedar	Tree	S3	S3			
Chamaedaphne calyculata	Leatherleaf	Shrub		S1			
Chamaelirium luteum	Devil's-bit	Forb	S1	S3			
Chamaesyce bombensis	Southern beach spurge	Forb				S2	
Chamaesyce polygonifolia	Small sea-side spurge	Forb			S2		
Chamaesyce vermiculata	Worm seeded spurge	Forb					S2

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Chasmanthium latifolium	Wild oat	Graminoid			S1		
Chasmanthium laxum	Slender sea-oats	Graminoid			S1		
Cheilanthes alabamensis	Alabama lipfern	Cryptogam				S1	
Cheilanthes eatonii	Chestnut lipfern	Cryptogam				S2	S2
Cheilanthes feei	Fee's lipfern	Cryptogam				S1	
Cheilanthes tomentosa	Woolly lipfern	Cryptogam					S1
Chelone cuthbertii	Cuthbert turtlehead	Forb				S2	
Chelone obliqua	Red turtlehead	Forb		S1		S1	
Chenopodium foggii	Fogg's goosefoot	Forb			S1		
Chenopodium gigantospermum	Maple-leaved goosefoot	Forb		S1			
Chenopodium standleyanum	Standley's goosefoot	Forb		S1			S2
Chimaphila umbellata	Prince's pine	Forb		S3			
Chimaphila umbellata var. *cisatlantica*	Pipsissewa	Forb	S2				
Chionanthus virginicus	Fringe tree	Tree			S3		
Chrysogonum virginianum	Green-and-gold	Forb		S3	S1		
Chrysopsis gossypina	Cottony goldenaster	Forb				S1	
Chrysopsis mariana	Maryland golden-aster	Forb			S1		
Cicuta bulbifera	Bulb-bearing water-hemlock	Forb	S1	S1			S1
Cimicifuga rubifolia	Appalachian bugbane	Forb				S2	
Cinna latifolia	Slender wood reedgrass	Graminoid		S3			
Cirsium altissimum	Tall thistle	Forb				S1	
Cirsium carolinianum	Carolina thistle	Forb				S1	
Cirsium horridulum	Horrible thistle	Forb		S3	S1		
Cirsium muticum	Swamp thistle	Forb		S3			
Cirsium pumilum	Pasture thistle	Forb	S3				
Cirsium virginianum	Virginia thistle	Forb				S2	
Cladium jamaicense	Sawgrass	Graminoid				S2	
Cladium mariscoides	Twig rush	Graminoid			S2		
Claytonia caroliniana	Carolina spring-beauty	Forb		S3			
Cleistes bifaria	Spreading pogonia	Forb				S2	S1
Cleistes divaricata	Spreading pogonia	Forb		S1		S1	
Clematis addisonii	Addison's leatherflower	Vine				S2	

(continued)

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Clematis albicoma	White-haired leatherflower	Vine					S3
Clematis catesbyana	Satin curls	Vine				S1	
Clematis occidentalis (var. *occidentalis*)	Purple clematis	Vine		S1		S2	S2
Clematis viorna	Vase-vine leather flower	Vine	S1.1	S3	S1		
Clematis viticaulis	Millboro leatherflower	Vine				S2	
Clethra acuminata	Mountain pepper-bush	Shrub			S1		
Clintonia alleghaniensis	Harned's swamp clintonia	Forb		S1			
Clintonia borealis	Yellow clintonia	Forb		S2			
Clitoria mariana (var. *mariana*)	Maryland butterfly-pea	Forb	S2		S1		
Cocculus carolinus	Red-berried moonseed	#N/A				S1	
Coeloglossum viride	Long-bracted orchis	Forb		S1			
Coeloglossum viride var. *virescens*	Long-bract green orchis	Forb					S1
Coelorachis rugosa	Wrinkled jointgrass	#N/A	S1	S1		S1	
Collinsia verna	Eastern blue-eyed Mary	Forb				S2	
Collinsonia verticillata	Whorled horsebalm	Forb				S1	
Commelina erecta	Sand dayflower	Forb	S2	S3			
Commelina erecta var. *angustifolia*	Slender dayflower	Forb					S2
Commelina virginica	Virginia dayflower	Forb	S1				
Comptonia peregrina	Sweet-fern	Shrub	S2				
Conioselinum chinense	Hemlock parsley	Forb			S1	S1	
Conopholis americana	Squaw-root	Forb	S2				
Coptis trifolia	Goldthread	Forb		S1			S3
Corallorhiza bentleyi	Bentley's coralroot	Forb				S1	S1
Corallorhiza maculata var. *occidentalis*	Spotted coralroot	Forb				S1	S1
Corallorhiza odontorhiza	Autumn coralroot	Forb	S1				
Corallorhiza trifida	Early coralroot	Forb		S1			S1
Corallorhiza wisteriana	Wister's coralroot	Forb		S1	S1		S2
Coreopsis falcata	Pool coreopsis	Forb				S1	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Coreopsis linifolia	Atlantic tickseed	Forb				S1	
Coreopsis pubescens var. *robusta*	Star tickseed	Forb					S2
Coreopsis rosea	Rose coreopsis	Forb	S1	S1			
Coreopsis tripteris	Tall tickseed	Forb		S1			
Coreopsis verticillata	Whorled coreopsis	Forb		S3			S1
Cornus amomum ssp. *obliqua*	Silky dogwood	Shrub				S2	
Cornus canadensis	Bunchberry	Subshrub		S1		S1	S2
Cornus rugosa	Round-leaved dogwood	Forb		S1		S1	S1
Cornus stricta	Marsh dogwood	Shrub	S2				
Corydalis aurea	Golden corydalis	Forb			S1		
Corydalis flavula	Yellow corydalis	Forb	S1.1				
Corydalis sempervirens	Pale corydalis	Forb		S3			
Corylus cornuta	Beaked hazelnut	Shrub	S1.1	S3			
Crataegus calpodendron	Pear hawthorn	Tree				S1	
Crataegus mollis	Hawthorn	Tree				S1	
Crataegus pennsylvanica	Red-fruited hawthorn	Tree			S2S3		
Crataegus spathulata	Littlehip hawthorn	Tree					S1
Crataegus succulenta	Fleshy hawthorn	Tree				S1	
Crocanthemum bicknellii	Plains frostweed	Forb				S1	
Crocanthemum canadense	Canada frostweed	Forb	S3				
Crocanthemum propinquum	Low frostweed	Forb	S3			S1	
Crotalaria purshii	Rattle-box	Forb				S2	
Crotalaria rotundifolia	Rabbit-bells	#N/A		S1			
Croton glandulosus var. *septentrionalis*	Northern croton	Forb					S3
Cryptogramma stelleri	Slender rock-brake	Cryptogam			S1		S1
Ctenium aromaticum	Toothache grass	#N/A				S1	
Cunila origanoides	Dittany	#N/A	S2				
Cuphea viscosissima	Blue waxweed	Forb	S2				
Cuscuta campestris	Dodder	Forb			S2		
Cuscuta cephalanthi	Button-bush dodder	Forb			S2		
Cuscuta compacta	Dodder	Forb			S2		
Cuscuta indecora var. *neuropetala*	Pretty dodder	Forb					S1
Cuscuta pentagona	Field dodder	Forb			S2		
Cuscuta polygonorum	Smartweed dodder	Forb		S1	S2		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Cuscuta rostrata	Beaked dodder	Forb		S1		S2	S2
Cuthbertia graminea	Grass-like roselings	#N/A				S1	
Cymophyllus fraserianus	Fraser's sedge	Graminoid		S1	S1		S3
Cynanchum laeve	Smooth swallow-wort	#N/A			S1		
Cynoglossum virginianum (var. *virginianum*)	Wild comfrey	Forb	S3				
Cyperus dentatus	Toothed flatsedge	Graminoid				S1	
Cyperus diandrus	Umbrella flatsedge	Graminoid	S1		S2	S1	
Cyperus engelmannii	Engelmann's flatsedge	Graminoid				S1	
Cyperus granitophilus	Granite-loving flatsedge	Graminoid				S1	
Cyperus houghtonii	Houghton's umbrella-sedge	Graminoid		S1	S1		
Cyperus hystricinus	Flatsedge	Graminoid	S1				
Cyperus lancastriensis	Lancaster's flatsedge	Graminoid	S1		S2		
Cyperus odoratus var. *engelmannii*	Engelman's rusty flatsedge	Graminoid	S3				
Cyperus plukenetii	Plukenet's flatsedge	Graminoid				S2	
Cyperus refractus	Reflexed flatsedge	Graminoid	S1		S1		S3
Cyperus retrofractus	Rough cyperus	Graminoid		S2			
Cyperus schweinitzii	Schweinitz's flatsedge	Graminoid			S2		
Cyperus squarrosus	Awned cyperus	Graminoid					S3
Cypripedium calceolus var. *parviflorum*	Small Yellow lady's-slipper	Forb			S1		
Cypripedium candidum	Small White lady's Slipper	Forb		S1		S1	
Cypripedium kentuckiense	Kentucky lady's-slipper	Forb				S1	
Cypripedium parviflorum var. *pubescens*	Large yellow lady's-slipper	Forb	S1				
Cypripedium reginae	Showy lady's-slipper	Forb			S2	S1	S1
Cystopteris bulbifera	Bulblet fern	Cryptogam		S3			
Cystopteris laurentiana	Laurentian bladder-fern	Cryptogam			S1		
Cystopteris protrusa	Lowland fragile fern	Cryptogam	S2				

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Cystopteris tennesseensis	Tennessee bladder-fern	Cryptogam		S1	S1	S1	
Cystopteris tenuis	Bladderfern	Cryptogam	S1				
Dalibarda repens	Star violet	Forb				S1	S3
Danthonia compressa	Flattened oatgrass	Graminoid	S2				
Danthonia sericea	Silky oatgrass	Graminoid	S1				
Dasistoma macrophylla	Mullein foxglove	Forb				S1	S2
Decodon verticillatus	Hairy swamp loosestrife	Forb					S1
Delphinium exaltatum	Tall larkspur	Forb		S1	S1		S2
Delphinium tricorne	Dwarf larkspur	Forb		S3			
Deschampsia cespitosa	Tufted hairgrass	Graminoid		S1	S3	S1	
Desmodium canadense	Showy tick-trefoil	Forb		S3		S1	
Desmodium canescens	hoary tick-trefoil	Forb	S1				
Desmodium cuspidatum (var. *cuspidatum*)	Toothed tick-trefoil	Forb		S1		S2	
Desmodium fernaldii	Fernald's tick-trefoil	Forb	S1.1				
Desmodium glabellum	Dillen's tick-trefoil	Forb	S3				
Desmodium laevigatum	Smooth tick-trefoil	Forb		S3S4			
Desmodium lineatum	Linear-leaved tick-trefoil	Forb		S1			S1
Desmodium marilandicum	Maryland tick-trefoil	Forb	S3				
Desmodium nuttallii	Nuttalls' tick-trefoil	Forb			S2		
Desmodium obtusum	Stiff tick-trefoil	Forb	S1				
Desmodium ochroleucum	Cream-flowered tick-trefoil	Forb		S1			
Desmodium pauciflorum	Few-flowered tick-trefoil	Forb		S1			S1
Desmodium rigidum	Rigid tick-trefoil	Forb		S1			
Desmodium rotundifolium	Prostrate tick-trefoil	Forb	S3				
Desmodium sessilifolium	Sessile-leaf tick-trefoil	Forb				S2	
Desmodium strictum	Pineland tick-trefoil	Forb	S2	S1		S2	
Desmodium tenuifolium	Slim-leaf tick-trefoil	Forb				S1	
Desmodium viridiflorum	Velvety tick-trefoil	Forb		S3S4			
Diamorpha smallii	Small's stonecrop	Forb				S1	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Diarrhena americana	American beakgrain	Graminoid			S1		
Diarrhena obovata	Beak sedge	Graminoid				S1	S1
Dicentra cucullaria	Dutchman's breeches	Forb	S3				
Dicentra eximia	Wild bleeding-hearts	Forb		S2	S1		
Dichanthelium aciculare	Needle-leaf witch grass	Graminoid	S1				
Dichanthelium acuminatum (var. *acuminatum*)	Panic grass	Graminoid					S1
Dichanthelium annulum	Serpentine panic grass	Graminoid			S2	S2	
Dichanthelium boreale	Northern witch grass	Graminoid					S1
Dichanthelium caerulescens	Blue witch grass	Graminoid				S1	
Dichanthelium dichotomum var. *roanokense*	Roanoke witch grass	Graminoid	S2				
Dichanthelium ensifolium	Witch grass	Graminoid	S1				
Dichanthelium hirstii	Hirst Brothers' panic grass	Graminoid	S1.1				
Dichanthelium laxiflorum	Lax-flower witch grass	Graminoid			S1		
Dichanthelium longiligulatum	Long-ligule witch grass	Graminoid	S1				
Dichanthelium lucidum	Shining panic grass	Graminoid			S1		
Dichanthelium meridionale	Matting witch grass	Graminoid					S3
Dichanthelium oligosanthes (var. *oligosanthes*)	Few-flower witch grass	Graminoid	S1	S2S3	S3		
Dichanthelium oligosanthes var. *scribnerianum*	Scribner's witch grass	Graminoid	S2				
Dichanthelium ravenelii	Ravenel's witch grass	Graminoid	S2	S3			
Dichanthelium scabriusculum	Tall swamp witch grass	Graminoid	S1	S1			
Dichanthelium scoparium	Velvety panic grass	Graminoid			S1		
Dichanthelium sphaerocarpon	Roundfruit witch grass	Graminoid	S3				
Dichanthelium spretum	Eaton's witch grass	Graminoid			S1		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Dichanthelium strigosum	Rough-hair panic grass	Graminoid				S1	
Dichanthelium wrightianum	Wright's panic grass	Graminoid	S2	S1			
Dichanthelium yadkinense	Yadkin's witch grass	Graminoid	S1		S1		
Didiplis diandra	Water-purslane	Forb				S1	
Diervilla lonicera	Northern bush-honeysuckle	Shrub	S1				
Digitaria cognata	Fall witch grass	Graminoid				S2	
Digitaria filiformis	Slender crabgrass	Graminoid					S1
Digitaria serotina	Dwarf crabgrass	Graminoid				S1	
Diphasiastrum tristachyum	Ground-cedar clubmoss	Bryophyte	S2				
Diplazium pycnocarpon	Glade fern	Cryptogam	S1.1	S2			
Dirca palustris	Eastern leatherwood	Shrub	S1.1	S2			
Dodecatheon meadia	Common shooting-star	Forb		S3	S1		
Dodecatheon radicatum	Jeweled shooting-star	Forb			S2		
Doellingeria infirma	Cornel-leaf aster	Forb	S1	S3			
Doellingeria umbellata	Flat-top white aster	Forb	S3				
Drosera capillaris	Pink sundew	Forb		S1			
Drosera rotundifolia (var. *rotundifolia*)	Roundleaf sundew	Forb	S2	S3			S3
Dryopteris campyloptera	Mountain wood-fern	Cryptogam		S1	S1		
Dryopteris celsa	Log fern	Cryptogam	S2	S3	S1		
Dryopteris clintoniana	Clinton's wood-fern	Cryptogam	S1	S1	S2		
Dryopteris goldiana	Goldie's wood-fern	Cryptogam	S1	S2			
Dryopteris marginalis	Marginal wood-fern	Cryptogam	S3				
Echinacea laevigata	Smooth coneflower	Forb				S2	
Echinochloa walteri	Walter's barnyard-grass	Graminoid			S1		
Echinodorus cordifolius	Upright burhead	Forb		S1			
Echinodorus tenellus	Dwarf burhead	Forb				S1	
Elatine americana	American waterwort	Forb	S2	S3			
Elatine minima	Small waterwort	Forb	S2	S1		S1	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Eleocharis albida	White spikerush	Graminoid		S2			
Eleocharis baldwinii	Baldwin spikerush	Graminoid				S1	
Eleocharis caribaea	Capitate spikerush	Graminoid			S1		
Eleocharis compressa	Flattened spikerush	Graminoid		S1	S1	S2	S2
Eleocharis elliptica	Slender spikerush	Graminoid			S2		S1
Eleocharis engelmannii	Engelmann spikerush	Graminoid		S3			S1
Eleocharis equisetoides	Horsetail spikerush	Graminoid	S1	S1		S1	
Eleocharis erythropoda	Bald spikerush	Graminoid	S1				
Eleocharis fallax	Creeping spikerush	Graminoid	S3				
Eleocharis halophila	Salt-marsh spikerush	Graminoid	S1.1	S1			
Eleocharis intermedia	Matted spikerush	Graminoid		S1	S2	S1	S1
Eleocharis melanocarpa	Black-fruited spikerush	Graminoid	S2	S1		S2	
Eleocharis obtusa var. *peasei*	Wrights spikerush	Graminoid			S1		
Eleocharis palustris	Creeping spikerush	Graminoid					S3
Eleocharis parvula	Little-spike spikerush	Graminoid			S1		
Eleocharis pauciflora var. fernaldii	Few-flowered spikerush	Graminoid			S1		
Eleocharis quadrangulata	Four-angled spikerush	Graminoid			S1		S2
Eleocharis robbinsii	Robbins' spikerush	Graminoid	S3	S1	S2	S1	
Eleocharis rostellata	Beaked spikerush	Graminoid	S2		S1		S1
Eleocharis tenuis var. *verrucosa*	Slender spikerush	Graminoid			S1		
Eleocharis tortilis	Twisted spikerush	Graminoid		S3			
Eleocharis tricostata	Three-angle spikerush	Graminoid	S1	S1		S1	
Eleocharis tuberculosa	Long-tubercled spikerush	Graminoid			S1		
Eleocharis uniglumis	Creeping spikerush	Graminoid				S1	
Eleocharis vivipara	Viviparous spikerush	Graminoid				S1	
Eleocharis wolfii	Wolf's spikerush	Graminoid				S1	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Elephantopus carolinianus	Elephant's foot	Forb			S3		
Elephantopus nudatus	Smooth elephant's foot	Forb	S2				
Ellisia nyctelea	Ellisia	Forb			S2		
Elodea nuttallii	Nuttall waterweed	Forb					S3
Elymus hystrix	Bottlebrush wild rye	Graminoid	S2				
Elymus riparius	River wild rye	Graminoid	S3				
Elymus trachycaulus (ssp. *Trachycaulus*)	Slender wheatgrass	Graminoid			S3	S2	S2
Elymus villosus	Hairy wild rye	Graminoid	S3				
Elymus virginicus var. *halophilus*	Salt-loving Virginia wild rye	Graminoid	S3				
Endodeca serpentaria	Virginia snakeroot	#N/A	S3				
Enemion biternatum	False rue-anemone	Forb				S1	S1
Epigaea repens	Trailing arbutus	#N/A	S3				
Epilobium ciliatum	Northern willowherb	Forb		S1			
Epilobium leptophyllum	Linear-leaved willowherb	Forb		S2S3		S2	
Epilobium palustre	Marsh willowherb	Forb			S1		
Epilobium strictum	Downy willowherb	Forb		S1	S3		
Equisetum fluviatile	Water horsetail	Cryptogam	S1	S1		S1	S2
Equisetum hyemale subsp. *affine*	Scouring-rush horsetail	Cryptogam	S1				
Equisetum sylvaticum	Woodland horsetail	Cryptogam	S1	S1		S1	S1
Equisetum variegatum	Variegated horsetail	Cryptogam			S1		
Equisetum x ferrissii	Scouring rush	Cryptogam			S1		
Eragrostis hypnoides	Teal lovegrass	Graminoid	S2				
Eragrostis refracta	Meadow lovegrass	Graminoid	S2	S3S4			
Erigenia bulbosa	Harbinger-of-spring	Forb		S3			
Erigeron pulchellus var. *brauniae*	Lucy Braun's robin plantain	Forb		S1			
Erigeron vernus	White-top fleabane	Forb				S2	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Eriocaulon aquaticum	Seven-angled pipewort	#N/A	S2	S1		S1	
Eriocaulon compressum	Flattened pipewort	#N/A	S2	S2			
Eriocaulon decangulare	Ten-angle pipewort	#N/A	S1	S2		S2	
Eriocaulon parkeri	Parker's pipewort	#N/A	S2	S2		S2	
Eriogonum allenii	Yellow buckwheat	Forb					S2
Eriophorum gracile	Slender cottongrass	Graminoid		S1	S1		
Eriophorum tenellum	Rough cottongrass	Graminoid			S1		
Eriophorum virginicum	Tawny cottongrass	Graminoid	S1	S3			
Eriophorum viridicarinatum	Thin-leaved cottongrass	Graminoid			S2		
Eryngium aquaticum (var. *aquaticum*)	Rattlesnake-master	Forb	S2				
Eryngium integrifolium	Savanna eryngo	Forb				S1	
Eryngium yuccifolium (var. *yuccifolium*)	Rattlesnake-master	Forb				S2	
Erysimum capitatum (var. *capitatum*)	Prairie rocket	Forb				S2	S1
Erythronium albidum	White trout lily	Forb		S2	S3	S2	
Euonymus americanus	Bursting heart	Forb			S3S4		
Euonymus atropurpureus	Wahoo	Shrub	S1				
Eupatorium album var. *vaseyi*	Vasey's white thoroughwort	Forb	S3				
Eupatorium altissimum	Tall boneset	Forb		S3			
Eupatorium anomalum	Anomalous eupatorium	Forb				S1	
Eupatorium godfreyanum	Vasey's eupatorium	Forb			S1S2		S2S3
Eupatorium hyssopifolium (var. *hyssopifolium*)	Hyssopleaf thoroughwort	Forb					S1
Eupatorium hyssopifolium var. *laciniatum*	Fringed boneset	Forb					S1
Eupatorium incarnatum	Pink thoroughwort	Forb				S2	
Eupatorium leucolepis	White-bracted boneset	Forb	S3	S2S3			
Eupatorium maculatum (var. *maculatum*)	Mottle joe-pye weed	Forb				S2	S1
Eupatorium pilosum	Vervain thoroughwort	Forb					S2
Eupatorium rotundifolium	Eupatorium	Forb			S3		

(continued)

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Eupatorium serotinum	Late-flowering thoroughwort	Forb	S3				
Eupatorium sessilifolium (var. *sessilifolium*)	Sessile-leaf thoroughwort	Forb	S1				
Euphorbia exserta	Coastal-sand spurge	Forb				S1	
Euphorbia ipecacuanhae	Wild ipecac	Forb			S1		
Euphorbia obtusata	Blunt-leaved spurge	Forb		S1	S1		
Euphorbia pubentissima	Flowering spurge	Forb					S1
Euphorbia purpurea	Darlington's spurge	Forb	S1.1	S1	S1	S2	S2
Eurybia radula	Rough-leaved aster	Forb		S1	S2	S1	
Eurybia schreberi	Schreber's aster	Forb	S2				
Eurybia spectabilis	Showy aster	Forb	S1	S1	S1		
Eurybia surculosa	Creeping aster	Forb				S1	
Euthamia tenuifolia	Grass-leaved goldenrod	Forb			S1		
Eutrochium purpureum (var. *purpureum*)	Purple-node Joe-pye weed	Forb	S3				
Festuca paradoxa	Cluster fescue	Graminoid			S1		
Filipendula rubra	Queen-of-the-prairie	Forb		S1	S1S2	S2	
Fimbristylis annua	Annual fimbry	Graminoid		S3	S2		S1
Fimbristylis caroliniana	Carolina fimbry	Graminoid	S2	S1S2			
Fimbristylis perpusilla	Harper's fimbristylis	Graminoid	S1	S2		S1	
Fimbristylis puberula (var. *puberula*)	Hairy fimbry	Graminoid				S1	
Floerkea proserpinacoides	False mermaid-weed	Forb	S3				
Fraxinus nigra	Black ash	Tree	S2	S3			S3
Fraxinus profunda	Pumpkin ash	Tree			S1		
Fraxinus quadrangulata	Blue ash	Tree			S1		S1
Fuirena pumila	Smooth fuirena	#N/A	S3	S2S3			
Fuirena squarrosa	Hairy umbrella sedge	#N/A	S3				
Galactia volubilis	Downy milkpea	Vine		S3			S2
Galearis spectabilis	Showy orchis	Forb	S3				
Galium asprellum	Rough bedstraw	Forb	S2				
Galium boreale	Northern bedstraw	Forb		S1			
Galium concinnum	Shining bedstraw	Forb		S3			
Galium hispidulum	Coast bedstraw	Forb		S1			

(continued)

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Galium labradoricum	Labrador marsh bedstraw	Forb			S1		
Galium lanceolatum	lance-leaf bedstraw	Forb	S3				
Galium latifolium	Purple bedstraw	Forb		S3	S3		
Galium trifidum	Marsh bedstraw	Forb			S2		
Gaultheria hispidula	Creeping snowberry	#N/A		S1	S3		S3
Gaylussacia brachycera	Box huckleberry	#N/A	S1	S1	S1	S2	S2
Gaylussacia dumosa	Dwarf huckleberry	Shrub					S1
Gentiana alba	Yellow gentian	Forb					S1
Gentiana andrewsii (var. *andrewsii*)	Closed bottle gentian	Forb	S1.1	S2			
Gentiana austromontana	Appalachian gentian	Forb					S1
Gentiana autumnalis	Pine-barren gentian	Forb				S1	
Gentiana catesbaei	Elliott's gentian	Forb	S3				
Gentiana linearis	Narrow-leaved gentian	Forb		S3			
Gentiana saponaria	Soapwort gentian	Forb	S3		S1S2		
Gentiana villosa	Striped gentian	Forb		S1	S1		
Gentianella quinquefolia	Stiff gentian	Forb		S1			
Gentianopsis crinita	Fringed gentian	Forb		S1		S1	
Geranium bicknellii	Cranesbill	Forb			S1		
Geranium robertianum	Herb robert	Forb		S1			
Geum aleppicum	Yellow avens	Forb		S1			S1
Geum laciniatum	Rough avens	Forb					
Geum rivale	Purple avens	Forb					S1
Geum virginianum	Pale avens	Forb	S1				
Glyceria acutiflora	Sharp-scaled mannagrass	Graminoid	S2	S1			S2
Glyceria canadensis	Canada mannagrass	Graminoid	S1				
Glyceria grandis (var. *grandis*)	American mannagrass	Graminoid	S1.1	S1		S1	S2
Glyceria laxa	Northern mannagrass	Graminoid					S2S3
Glyceria obtusa	Blunt mannagrass	Graminoid			S1		
Gnaphalium uliginosum	Low cudweed	Forb				S1	
Goodyera repens	Dwarf rattlesnake-plantain	Forb			S2		S1S2

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Goodyera tesselata	Checkered rattlesnake-plantain	Forb			S1		
Gratiola aurea	Golden hedge hyssop	Forb	S3		S1		
Gratiola brevifolia	Branching hedge hyssop	Forb	S1.1				
Gratiola ramosa	Branched hedge hyssop	Forb				S1	
Gratiola viscidula	Short's hedge hyssop	Forb		S1			
Gymnocarpium appalachianum	Appalachian oak fern	Cryptogam			S1		S2
Gymnocarpium dryopteris	Oak fern	Cryptogam		S1			S1
Gymnocladus dioicus	Kentucky coffee-tree	Tree		S1			
Gymnopogon ambiguus	Eastern beardgrass	Graminoid	S1				S1
Gymnopogon brevifolius	Broad-leaved beardgrass	Graminoid		S1			
Hasteola suaveolens	Sweet-scented Indian-plantain	Forb		S1		S2	S3
Hedyotis nigricans	Barren bluets	Forb				S1	
Helenium brevifolium	Shortleaf sneezeweed	Forb				S2	
Helenium virginicum	Virginia sneezeweed	Forb				S2	
Helianthemum bicknellii	Hoary frostweed	Forb		S1	S2		S1
Helianthemum canadense	Canada frostweed	Forb					S2
Helianthemum propinquum	Frostweed	Forb			S1S3		S1
Helianthus angustifolius	Swamp sunflower	Forb	S3				
Helianthus decapetalus	Thinleaf sunflower	Forb	S3				
Helianthus divaricatus	Woodland sunflower	Forb	S1				
Helianthus hirsutus	Sunflower	Forb			S2		
Helianthus laevigatus	Smooth sunflower	Forb		S1			S2
Helianthus microcephalus	Small-headed sunflower	Forb		S1			
Helianthus occidentalis (ssp. *Occidentalis*)	McDowell sunflower	Forb		S1		S1	S2
Heliopsis helianthoides (var. *helianthoides*)	Ox-eye	Forb	S1				

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Heliotropium curassavicum	Seaside heliotrope	Forb				S1	
Helonias bullata	Swamp-pink	Forb	S2	S2		S2S3	
Hemicarpha micrantha	Dwarf bulrush	Graminoid				S1	
Heracleum lanatum	Cow-parsnip	Forb		S3			
Heracleum maximum	Cow-parsnip	Forb	S3				
Heteranthera dubia	Grassleaf mud-plantain	Forb	S2				
Heteranthera multiflora	Multiflowered mud-plantain	Forb			S1	S1	
Heteranthera reniformis	Kidneyleaf mud-plantain	Forb					S1
Heuchera alba	White-flowered alumroot	Forb					S2
Heuchera americana	American alumroot	Forb	S3				
Heuchera americana var. *hispida*	Rough alumroot	Forb					S2
Heuchera caroliniana	Carolina alumroot	Forb				S1	
Heuchera longiflora	Long-flowered alumroot	Forb					S2
Heuchera pubescens	Downy heuchera	Forb		S3			
Hexalectris spicata (var. *spicata*)	Crested coralroot	Forb					S1
Hexastylis contracta	Mountain heartleaf	Forb				S1	
Hexastylis virginica	Virginia heartleaf	Forb		S1			
Hibiscus laevis	Halberd-leaved mallow	Forb		S3			S2
Hieracium traillii	Maryland hawkweed	Forb			S1		
Hieracium umbellatum	Umbellate hawkweed	Forb			S1		
Hierochloe hirta ssp. *arctica*	Common northern sweet grass	Graminoid			S1		S1
Hierochloe odorata (ssp. *odorata*)	Vanilla grass	Graminoid		S1		S1	
Honckenya peploides	Sea-beach sandwort	Forb		S1			
Hordeum jubatum	Fox-tail barley	Graminoid				S1	
Hottonia inflata	Featherfoil	forb	S2	S1			S1
Houstonia canadensis	Canada bluets	forb				S2	
Houstonia purpurea (var. *purpurea*)	Purple bluets	forb	S2		S1		
Houstonia serpyllifolia	Creeping bluets	forb		S3	S1		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Houstonia tenuifolia	Slender-leaved bluets	forb		S1			
Hudsonia ericoides	Golden heather	forb	S1	S1			
Hudsonia tomentosa (var. *tomentosa*)	False heather	forb					S1
Huperzia appalachiana	Appalachian fir-clubmoss	Bryophyte				S2	
Huperzia porophila	Rock clubmoss	Bryophyte			S1		S1
Huperzia porophila	Rock clubmoss	Bryophyte				S1	
Hybanthus concolor	Green violet	#N/A	S1	S3			
Hydrastis canadensis	Goldenseal	Forb	S3	S2			
Hydrocotyle americana	American water-pennywort	Forb	S3				
Hydrocotyle ranunculoides	Floating pennywort	Forb					S2
Hydrophyllum macrophyllum	Large-leaved waterleaf	Forb		S2			
Hydrophyllum virginianum	Eastern waterleaf	Forb	S3				
Hylodesmum glutinosum	Large tick-trefoil	Forb	S2				
Hypericum adpressum	Creeping St. John's-wort	Forb	S2	S1		S1	
Hypericum boreale	Northern St. John's-wort	Forb	S1			S2	
Hypericum crux-andreae	St. Peter's-wort	Subshrub	S3				
Hypericum densiflorum	Bushy St. John's-wort	Shrub	S2		S3		
Hypericum denticulatum	Coppery St. John's-wort	Forb	S2	S2			
Hypericum dissimulatum	Disguised St. John's-wort	Forb			S2S4		
Hypericum drummondii	Drummond St. John's-wort	Forb	S1.1				S1
Hypericum gymnanthum	Clasping-leaved St. John's-wort	Forb	S2	S3	S1		
Hypericum hypericoides	Erect St. Andrew's cross	Subshrub	S3				
Hypericum majus	Larger Canadian St. John's-wort	Forb			S2		
Hypericum mitchellianum	Blue Ridge St. John's-wort	Forb					S1
Hypericum prolificum	Shrubby St. John's-wort	Shrub	S1.1				
Hypericum setosum	St. John's-wort	Forb				S1S2	

(continued)

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Hypericum stragulum	St Andrew's-cross	Forb			S2		
Hypericum virgatum	Coppery St. John's-wort	Forb					S1
Hypopitys monotropa	American pinesap	#N/A	S2				
Hypoxis hirsuta	Eastern yellow stargrass	#N/A	S3				
Ilex collina	Long-stalked holly	Shrub				S2	S2
Ilex coriacea	Bay-gall holly	Shrub				S2	
Ilex decidua	Deciduous holly	Shrub		S2			
Ilex opaca	American holly	Shrub			S2		
Iliamna corei	Peter’s mountain mallow	#N/A				S1	
Iliamna remota	Kankakee globemallow	#N/A				S1	
Impatiens pallida	Pale jewel-weed	Forb	S3				
Iodanthus pinnatifidus	Purple rocket	Forb			S1		
Ionactis linariifolia	Flaxleaf aster	Forb	S2				
Iresine rhizomatosa	Bloodleaf	Forb		S1			
Iris cristata	Crested iris	Forb		S1	S1		
Iris prismatica	Slender blue flag	Forb	S2	S1	S1		
Iris verna	Dwarf iris	Forb		S1	S1		
Iris virginica	Virginia blue flag	Forb		S3	S2		
Isoetes engelmannii	Engelmann's quillwort	#N/A	S2	S3			
Isoetes hyemalis	Winter quillwort	#N/A				S2	
Isoetes riparia	Riverbank quillwort	#N/A	S1				
Isoetes tenella	Spiny-spored quillwort	#N/A	S1.1				
Isoetes valida	True quillwort	#N/A			S1S3		S1
Isoetes x brittonii	Quillwort	#N/A			S1S2		
Isotria medeoloides	Small-whorled pogonia	forb	S1.1		S1	S2	S1
Isotria verticillata	Large whorled pogonia	forb	S3				
Itea virginica	Virginia willow	#N/A			S1		
Iva imbricata	Sea-coast marsh elder	Shrub				S1S2	
Juglans cinerea	Butternut	Tree	S3	S2S3			S3
Juncus abortivus	Pine-barren rush	Graminoid				S1	
Juncus alpinoarticulatus ssp. *nodulosus*	Richardson's rush	Graminoid			S2		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Juncus arcticus var. *littoralis*	Baltic rush	Graminoid			S2		
Juncus articulatus	Jointed rush	Graminoid		S1		S1S2	S2
Juncus balticus var. *littoralis*	Baltic rush	Graminoid				S1	S1
Juncus biflorus	Grass-leaved rush	Graminoid			S2		S2
Juncus brachycarpus	Short-fruited rush	Graminoid			S1		
Juncus brachycephalus	Small-headed rush	Graminoid			S2	S2	
Juncus brevicaudatus	Narrow-panicled rush	Graminoid		S2		S2	
Juncus caesariensis	New Jersey rush	Graminoid		S1		S2	
Juncus coriaceus	leathery rush	Graminoid	S2				
Juncus debilis	Weak rush	Graminoid			S3		
Juncus dichotomus	Forked rush	Graminoid			S1		S1
Juncus elliottii	Bog rush	Graminoid				S1S2	
Juncus filiformis	Thread rush	Graminoid			S3		S2
Juncus longii	Long's rush	Graminoid		S1			
Juncus megacephalus	Big-headed rush	Graminoid				S2	
Juncus militaris	Bayonet rush	Graminoid	S2		S1		
Juncus nodosus (var. *nodosus*)	Knotted rush	Graminoid				S1	S1S2
Juncus pelocarpus	Brown-fruited rush	Graminoid	S2	S1		S1	
Juncus scirpoides	Scirpus-like rush	Graminoid			S1		S2
Juncus subcaudatus	short-tailed rush	Graminoid	S1				
Juncus torreyi	Torrey's rush	Graminoid		S1	S3	S2	S2
Juncus trifidus	Highland rush	Graminoid		S1		S1	S1
Juniperus communis	Common juniper	Tree			S2		
Juniperus communis var. *depressa*	Ground juniper	#N/A				S1	
Kalmia angustifolia	Sheep laurel	Shrub	S2	S3S4		S2	
Kalmia carolina	Carolina sheep laurel	Shrub				S2	
Krigia biflora	Two-flowered cynthia	#N/A		S3			
Krigia dandelion	Potato dandelion	#N/A		S1			
Kyllinga pumila	Thin-leaved flatsedge	#N/A		S1			
Lachnanthes caroliana (*Lachnanthes caroliniana*)		#N/A	S1	S1			
Lachnocaulon anceps	Bog-buttons	#N/A				S2	
Lactuca floridana	Woodland lettuce	Forb	S2				
Lactuca hirsuta	Downy lettuce	Forb			S3		
Larix laricina	Larch	Tree		S1			S1

(continued)

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Lathyrus japonicus	Beach peavine	#N/A			S2		
Lathyrus ochroleucus	Wild-pea	#N/A			S1		
Lathyrus palustris	Vetchling peavine	#N/A		S1	S1	S1	
Lathyrus venosus	Veiny pea	#N/A		S3	S2		
Lechea maritima	Beach pinweed	#N/A		S3			
Lechea minor	Thyme-leaved pinweed	#N/A			S1S3		
Lechea mucronata	hairy pinweed	#N/A	S2				
Lechea tenuifolia	Slender pinweed	#N/A					S1
Lechea villosa	Bog-buttons	#N/A			S3S4		
Ledum groenlandicum	Common Labrador-tea	#N/A			S3		
Leersia hexandra	Club-headed cutgrass	#N/A		S1			
Leersia lenticularis	Catchfly-grass	#N/A		S1			
Lemna perpusilla	Minute duckweed	Forb			S1S3		
Lemna trisulca	Star duckweed	Forb		S1		S1	
Lemna turionifera	Duckweed	Forb			S1S3		
Lemna valdiviana	Pale duckweed	Forb					S3
Lespedeza angustifolia	Narrowleaf bushclover	#N/A	S3		S1		
Lespedeza frutescens	Violet bushclover	#N/A		S3			
Lespedeza hirta (var. *hirta*)	Hairy bushclover	#N/A	S3				
Lespedeza stuevei	Tall bushclover	#N/A	S2	S3			
Leucothoe fontanesiana	Highland doghobble	Shrub				S1S2	
Leucothoe racemosa	Swamp doghobble	Shrub			S2S3		
Leucothoe recurva	Recurved fetterbush	Shrub					S1
Liatris pilosa	Grassleaf blazingstar	Forb	S2				
Liatris scariosa	Round-head gayfeather	Forb			S2		
Liatris scariosa var. *nieuwlandii*	Northern blazing-star	Forb					S1
Liatris spicata	Spiked blazing-star	Forb		S1			
Liatris squarrosa	Scaly blazing-star	Forb		S1			
Liatris squarrulosa	Appalachian gay-feather	Forb					S1
Liatris turgida	Turgid gay-feather	Forb					S2

(continued)

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Lilaeopsis carolinensis	Carolina lilaeopsis	Forb				S1	
Lilium canadense (subsp. *canadense*)	Canada lily	Forb	S2				
Lilium catesbaei	Southern red lily	Forb				S1	
Lilium grayi	Gray's lily	Forb				S2	
Lilium michauxii	Carolina lily	Forb					S1
Lilium philadelphicum (var. *philadelphicum*)	Wood lily	Forb					S2S3
Lilium pyrophilum	Sandhills lily	Forb				S1	
Limnobium spongia	American frog's-bit	Forb	S2	S1			
Limosella australis	Mudwort	Forb		S2			
Lindernia dubia var. *anagallidea*	False pimpernel	Forb					S2
Linnaea borealis	Twinflower	Forb			S1		
Linnaea borealis ssp. *americana*	Twinflower	Forb					S1
Linum intercursum	Sandplain flax	Forb	S1	S2	S1		
Linum lewisii var. *lewisii*	Prairie flax	Forb					S2
Linum sulcatum (var. *sulcatum*)	Grooved yellow flax	Forb		S1	S1		S1
Liparis liliifolia	Large twayblade	Forb	S2	S2S3			
Liparis loeselii	Loesel's twayblade	Forb		S1S2		S2	S3
Lipocarpha maculata	American lipocarpha	Graminoid				S1	
Lipocarpha micrantha	Small-flowered hemicarpha	Graminoid		S1	S1		
Listera australis	Southern twayblade	Forb	S3	S3	S1		
Listera cordata (var. *cordata*)	Heartleaf twayblade	Forb			S1		S2
Listera smallii	Appalachian twayblade	Forb		S1	S1		S2
Lithospermum canescens	Hoary puccoon	Forb			S2		
Lithospermum caroliniense	Hispid gromwell	Forb			S1	S1	
Lithospermum latifolium	American gromwell	Forb		S1			
Litsea aestivalis	Pondspice	Forb		S1		S1	
Lobelia canbyi	Canby's lobelia	Forb	S2	S1			
Lobelia dortmanna	Water lobelia	Forb			S2		
Lobelia elongata	Southern blue lobelia	Forb	S1.1	S3		S1	
Lobelia kalmii	Brook lobelia	Forb			S1		S1
Lobelia puberula	Downy lobelia	Forb			S1		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Lobelia spicata (var. *spicata)*	Pale-spike lobelia	Forb	S2				
Lonicera canadensis	Canada honeysuckle	Shrub		S1			S2
Lonicera hirsuta	Hairy honeysuckle	Vine			S1		
Lonicera oblongifolia	Swamp Fly honeysuckle	Shrub			S1		
Lonicera villosa	Mountain Fly honeysuckle	Shrub			S1		
Lotus helleri	Carolina prairie-trefoil	Forb				S1	
Ludwigia alata	Winged seedbox	Forb				S1	
Ludwigia brevipes	Long beach seedbox	Forb				S2	
Ludwigia decurrens	Upright primrose-willow	Forb		S2S3	S1		
Ludwigia glandulosa	Cylindric-fruited seedbox	Forb		S1			
Ludwigia hirtella	Hairy seedbox	Forb	S1	S1		S1	
Ludwigia leptocarpa	River seedbox	Forb					S2
Ludwigia linearis (var. *linearis*)	Narrow-leaf seedbox	Forb	S3				
Ludwigia pilosa	Hairy seedbox	Forb				S1	
Ludwigia polycarpa	False loosestrife seedbox	Forb			S1		S1
Ludwigia ravenii	Raven's seedbox	Forb				S1	
Ludwigia repens	Creeping seedbox	Forb				S1	
Lupinus perennis ssp. *perennis*	Sundial lupine	Forb	S1	S2	S3		S1
Luzula acuminata (var. *acuminata*)	Northern hairy woodrush	Graminoid	S1				
Luzula acuminata var. *carolinae*	Southern hairy woodrush	Graminoid	S1				
Luzula bulbosa	Southern wood-rush	Graminoid			S1		S1
Lycopodiella alopecuroides	Foxtail clubmoss	Bryophyte	S3		S1		
Lycopodiella appressa	Southern bog clubmoss	Bryophyte	S3		S2		
Lycopodiella caroliniana	Carolina clubmoss	Bryophyte		S1			
Lycopodiella inundata	Bog clubmoss	Bryophyte		S2		S1	S2
Lycopodiella margueritae	Clubmoss	Bryophyte			S1		
Lycopodiella x copelandii	Clubmoss	Bryophyte			S1		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Lycopodium clavatum	Staghorn clubmoss	Bryophyte	S3				
Lycopodium lagopus	One-cone ground-pine	Bryophyte					S1
Lycopodium tristachyum	Ground-cedar	Bryophyte		S3			
Lycopus amplectens	Sessile-leaved water-horehound	Forb	S2	S1			
Lycopus rubellus	Bugleweed	Forb			S1		
Lygodium palmatum	Climbing fern	Cryptogam	S1.1	S2			S3
Lyonia mariana	Stagger-bush	Shrub	S2		S1		
Lysimachia hybrida	Lance-leaf loosestrife	Forb	S2	S2	S1	S2	S1
Lysimachia lanceolata	Lance-leaved loosestrife	Forb		S3			
Lysimachia quadriflora	Four-flowered loosestrife	Forb				S1	S1
Lysimachia thyrsiflora	Tufted loosestrife	Forb		S1			S1
Lythrum alatum (var. *alatum*)	Winged loosestrife	Forb		S1	S1	S2	S2
Magnolia macrophylla	Bigleaf magnolia	Tree				S1	
Magnolia tripetala	Umbrella magnolia	Tree		S3	S2		
Magnolia virginiana	Sweet bay magnolia	Tree			S2		
Maianthemum stellatum	Starflower false solomon's-seal	Forb					S2
Malaxis bayardii	Bayard's malaxis	Forb			S1		
Malaxis monophyllos var. *brachypoda*	White adder's-mouth	Forb			S1		
Malaxis unifolia	Green adder's-mouth	Forb	S1				
Malus angustifolia	Narrow-leaved wild crab	Tree		S3			
Malvastrum hispidum	Hispid falsemallow	Forb				S1	
Manfreda virginica	False aloe	Subshrub				S2	S1
Marshallia grandiflora	Large-flowered marshallia	Forb			S1		S2
Marshallia obovata	Obovate marshallia	Forb				S2	
Matelea carolinensis	Anglepod	Forb	S2	S1			
Matelea decipiens	Old-field milkvine	Vine				S1	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Matelea obliqua	Climbing milkweed	Vine		S1	S1		
Matteuccia struthiopteris	Ostrich fern	Cryptogam		S2			S2
Matteuccia struthiopteris var. *pensylvanica*	Ostrich fern	Cryptogam				S1	
Mecardonia acuminata (var. *acuminata*)	Purple mecardonia	#N/A	S3	S1			
Meehania cordata	Heartleaf meehania	#N/A			S1		
Megalodonta beckii	Beck's water-marigold	Forb			S1		
Melanthium latifolium	Broad-leaved bunchflower	Forb		S1			
Melanthium virginicum	Virginia bunchflower	Forb		S3			
Melica mutica	Two-flowered melic grass	Graminoid		S1			S2
Melica nitens	Three-flower melic grass	Graminoid		S2	S2	S1S2	S1
Melothria pendula	Creeping cucumber	#N/A		S1			S1
Menyanthes trifoliata	Buckbean	#N/A		S1		S1	S1
Menziesia pilosa	Minniebush	#N/A			S3		
Mertensia virginica	Virginia bluebells	Forb	S3				
Micranthemum umbrosum	Shade mudflower	Forb				S1	
Micranthes pensylvanica	Swamp saxifrage	Forb	S1.1				
Micranthes virginiensis	Virginia saxifrage	Forb	S3				
Milium effusum	Millet grass	Graminoid		S3			
Mimosa quadrivalvis var. *angustata*	Little-leaf sensitivebriar	#N/A				S2	
Mimulus moschatus	Muskflower	Forb				S1	
Minuartia caroliniana	Carolina sandwort	Forb	S1.1	S1			
Minuartia glabra	Mountain sandwort	Forb		S1	S2		
Minuartia groenlandica	Mountain sandwort	Forb				S1	S1
Minuartia michauxii	Rock sandwort	Forb		S2			
Mitella diphylla	two-leaf Bishop's-cap	Forb	S2				
Mitella nuda	Naked Bishop's-cap	Forb			S1		
Mitreola petiolata	Lax hornpod	Forb				S1	
Mitreola sessilifolia	Sessile-leaved hornpod	Forb				S1	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Moehringia lateriflora	Grove sandwort	Forb		S1		S1	
Monarda clinopodia	Basil beebalm	Forb	S2	S3			
Monarda fistulosa ssp. *brevis*	Smoke hole bergamot	Forb					S1
Monarda fistulosa (var. *fistulosa*)	Wild bergamot beebalm	Forb	S1.1				
Monotropsis odorata	Sweet pinesap	#N/A		S1			S1
Montia chamissoi	Chamisso's miner's-lettuce	Forb			S1		
Morella caroliniensis	Evergreen bayberry	Shrub	S2	S1			
Morus rubra	Red mulberry	Tree	S3				
Muhlenbergia bushii	Bush’s muhly	Graminoid				S1	
Muhlenbergia capillaris (var. *capillaris*)	Long-awn hairgrass	Graminoid		S1			S1
Muhlenbergia cuspidata	Plains muhly	Graminoid				S2	
Muhlenbergia glomerata	Marsh muhly	Graminoid				S2	
Muhlenbergia sobolifera	Cliff muhly	Graminoid	S2				
Muhlenbergia sylvatica	Woodland dropseed	Graminoid		S3			
Muhlenbergia tenuiflora	Slender muhly	Graminoid	S2				
Muhlenbergia torreyana	Torrey's dropseed	Graminoid		S1			
Muhlenbergia uniflora	Fall dropseed muhly	Graminoid			S2		
Myosotis laxa (subsp. *laxa*)	Small forget-me-not	Forb	S3				
Myosotis macrosperma	Large-seed forget-me-not	Forb	S1	S2S3			S2
Myosotis verna	Spring forget-me-not	Forb		S3			
Myrica cerifera var. *pumila*	Dwarf southern bayberry	Shrub				S1	
Myrica gale	Sweet-gale	Shrub			S2		
Myriophyllum farwellii	Farwell's water-milfoil	Forb			S3		
Myriophyllum heterophyllum	Broadleaf water-milfoil	Forb		S1			
Myriophyllum humile	Low water-milfoil	Forb	S3			S1	
Myriophyllum laxum	Loose water-milfoil	Forb				S1	
Myriophyllum pinnatum	Cutleaf water-milfoil	Forb	S2				S1
Myriophyllum sibiricum	Northern water-milfoil	Forb			S1		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Myriophyllum tenellum	Slender water-milfoil	Forb			S2	S1	
Myriophyllum verticillatum	Whorled water-milfoil	Forb			S1		
Najas flexilis	Slender naiad	Forb		S3			
Najas gracillima	thread-like naiad	Forb	S1				S2
Najas guadalupensis	Southern naiad	Forb		S3			
Napaea dioica	Glade mallow	Forb		S1			
Nelumbo lutea	American lotus	Forb	S1.1	S2			
Nemopanthus mucronatus	Mountain holly	Shrub		S3			
Nemophila aphylla	Small-flowered baby-blue-eyes	Forb		S1			
Nestronia umbellula	Nestronia	Forb				S1	
Nuphar microphylla	Yellow cowlily	Forb			S1		
Nuphar sagittifolia	Narrow-leaved spatterdock	Forb				S1	
Nuttallanthus canadensis	Old-field toadflax	Forb					S2
Nymphoides aquatica	Larger floating-heart	Forb		S1		S1	
Nymphoides cordata	little floating-heart	Forb	S1	S1	S2		
Obolaria virginica	Virginia pennywort	Forb	S3				
Oclemena nemoralis	Bog aster	Forb			S1		
Oenothera argillicola	Shale-barren evening-primrose	Forb		S3	S2		S3
Oenothera fruticosa (var. *fruticosa*)	Narrow-leaf evening-primrose	Forb	S2				
Oenothera humifusa	Sea-beach evening-primrose	Forb	S3				
Oenothera oakesiana	Evening-primrose	Forb			S2		
Oenothera pilosella ssp. *pilosella*	Evening-primrose	Forb					S2
Oenothera tetragona (var. *tetragona*)	Shrubby evening-primrose	Forb	S2				
Oldenlandia boscii	Bosc's bluets	Forb				S1	
Oldenlandia uniflora	Clustered bluets	Forb		S3	S1		
Oligoneuron rigidum var. *glabratum*	Southeastern stiff goldenrod	Forb				S1	
Oligoneuron rigidum (var. *rigidum*)	Stiff goldenrod	Forb				S2	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Onosmodium molle	Shaggy false-gromwell	Forb		S1			
Onosmodium molle var. *hispidissimum*	False gromwell	Forb			S1		
Onosmodium virginianum	Virginia false-gromwell	Forb		S1		S2	
Ophioglossum engelmannii	Limestone Adder's-tongue	Cryptogam			S1		S1
Ophioglossum petiolatum	Longstem adder's-tongue	Cryptogam				S1	
Ophioglossum pusillum	Northern adder's tongue	Cryptogam				S1	
Ophioglossum vulgatum	Southern adder's-tongue	Cryptogam	S3				
Opuntia humifusa	Prickly-pear cactus	Forb			S3		
Orbexilum onobrychis	Lanceleaf scurfpea	Forb				S1	
Orobanche uniflora	One-flowered broomrape	Forb	S3				
Oryzopsis asperifolia	White-grained mountain-ricegrass	Graminoid		S2		S1	S1
Oryzopsis pungens	Slender mountain-ricegrass	Graminoid			S2		
Osmanthus americanus	Wild olive	#N/A				S1	
Osmunda cinnamomea var. *glandulosa*	Glandular cinnamon fern	Cryptogam				S1	
Ostrya virginiana	Eastern hop-hornbeam	Tree	S2				
Oxydendrum arboreum	Sourwood	Tree		S1	S3S4		
Oxypolis canbyi	Canby's dropwort	Forb		S1			
Oxypolis rigidior	Stiff cowbane	Forb			S2		
Packera anonyma	Small's ragwort	Forb	S1.1		S2		
Packera antennariifolia	Cat's-paw ragwort	Forb		S3	S1		S3
Packera millefolium	Yarrow-leaved ragwort	Forb				S2	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Packera paupercula (var. *paupercula*)	Balsam ragwort	Forb	S1	S3			S2
Packera plattensis	Prairie ragwort	Forb					S1
Panax quinquefolius	American ginseng	Forb	S2	S3			
Panicum commonsianum var. *euchlamydeum*	Cloaked panic grass	Graminoid			S2		
Panicum flexile	Wiry witch grass	Graminoid		S1	S2S3		
Panicum hemitomon	Maidencane panic grass	Graminoid	S2	S3		S2	
Panicum longifolium	Long-leaf panic grass	Graminoid			S1		
Panicum philadelphicum	Philadelphia panic grass	Graminoid	S1.1				
Panicum tuckermanii	Tuckerman's panic grass	Graminoid			S2		
Panicum verrucosum	Warty panic grass	Graminoid					S1
Panicum xanthophysum	Slender panic grass	Graminoid			S1		
Parnassia asarifolia	Kidneyleaf grass-of-parnassus	Forb		S1			S2
Parnassia glauca	Carolina grass-of-parnassus	Forb			S2		
Parnassia grandifolia	Largeleaf grass-of-parnassus	Forb				S2	S1
Paronychia argyrocoma	Silver nailwort	Forb					S3
Paronychia canadensis	Forked nailwort	Forb	S3				
Paronychia fastigiata (var. *fastigiata*)	Cluster-stemmed nailwort	Forb	S2				
Paronychia fastigiata var. *nuttallii*	Forked-chickweed	Forb			S1S2		
Paronychia fastigiata var. *paleacea*	Cluster-stemmed nailwort	Forb	S2				
Paronychia virginica (var. *virginica*)	Yellow nailwort	Forb		S1		S1	S2
Parthenium integrifolium	American feverfew	Forb		S1	S1		
Paspalum dissectum	Walter's paspalum	Graminoid	S3	S2		S2	
Paspalum distichum	Joint paspalum	Graminoid				S2	
Paspalum fluitans	Floating paspalum	Graminoid		S1			
Paspalum praecox	Early paspalum	Graminoid				S1	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Paspalum pubiflorum	Hairy-seed paspalum	Graminoid					S1
Paspalum setaceum (var. *setaceum*)	slender paspalum	Graminoid	S3				
Passiflora lutea	Passion-flower	Vine			S2		
Paxistima canbyi	Canby's mountain lover	#N/A		S1	S1	S2	S2
Pedicularis canadensis	Early wood lousewort	Forb	S2				
Pedicularis lanceolata	Swamp lousewort	Forb		S1	S1S2		S2
Pediomelum canescens	Hoary scurfpea	Forb				S1	
Pellaea glabella (ssp. *glabella*)	Smooth cliffbrake	Cryptogam		S1			S2
Peltandra virginica	Arrow-arum	Forb					S2
Penstemon australis	Southern beard-tongue	Forb				S2	
Penstemon calycosus	Long-sepal beard-tongue	Forb				S1	
Penstemon canescens	Beard-tongue	Forb			S3		
Penstemon laevigatus	Smooth beardtongue	Forb	S1	S3	S3		
Persea palustris	Red bay	Shrub		S1			
Persicaria setacea	Bog smartweed	Forb			S2		
Phacelia covillei	Blue scorpion-weed	Forb		S2		S1	S1
Phacelia fimbriata	Fringed scorpionweed	Forb				S2	
Phacelia purshii	Miami-mist	Forb		S3			
Phanopyrum gymnocarpon	Clustered panic grass	Graminoid				S1	
Phaseolus polystachios	Wild kidney bean	#N/A		S3	S1S2		
Phegopteris connectilis	Northern beech fern	Cryptogam		S2			
Phemeranthus teretifolius	Round-leaved fame-flower	Forb			S2		
Phlox amplifolia	Large-leaved phlox	Forb				S2	
Phlox buckleyi	Swordleaf phlox	Forb				S2	S2
Phlox glaberrima	Smooth phlox	Forb		S1			
Phlox ovata	Mountain phlox	Forb			S1		
Phlox pilosa	Downy phlox	Forb		S1	S1S2	S2	
Phlox subulata ssp. *brittonii*	Moss pink	Forb			S1		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Photinia floribunda	Purple chokeberry	Shrub		S3			
Phragmites australis subsp. *americanus*	North American reed	Graminoid	S2		S1		
Phyla lanceolata	Fog-fruit	Forb	S2				
Phyla nodiflora	Common frog-fruit	Forb				S1	
Phyllanthus caroliniensis	Carolina leaf-flower	Forb		S3	S1		
Physalis virginiana	Virginia ground-cherry	Forb		S3	S1S2		
Physalis walteri	Dune ground-cherry	Forb				S2	
Picea rubens	Red spruce	Tree		S3			
Pieris floribunda	Mountain fetter-bush	Shrub					S2
Pilea fontana	Coolwort	Forb		S3			
Pinus echinata	Short-leaf pine	Tree	S3		S1S2		
Pinus palustris	Long-leaf pine	Tree				S1	
Pinus resinosa	Red pine	Tree					S1
Pinus rigida	pitch pine	Tree	S3				
Piptatherum canadense	Canada mountain-ricegrass	Graminoid					S1
Piptatherum racemosum	Black-fruited mountainrice	Graminoid		S2			S2
Piptochaetium avenaceum	Blackseed needlegrass	Graminoid	S3		S1		S2
Pityopsis graminifolia	Grass-leaved golden-aster	Forb				S1	
Plantago maritima var. *juncoides*	Seaside plantain	Forb				S1	
Platanthera aquilonis	Northern green orchid	Forb			S1		
Platanthera blephariglottis	White-fringe orchis	Forb	S1	S2	S2S3	S1	
Platanthera ciliaris	Yellow fringed orchid	Forb		S2	S2		S3
Platanthera cristata	Yellow-crested orchis	Forb	S2	S3			
Platanthera dilatata	Leafy white orchid	Forb			S1		
Platanthera flava	Pale green orchid	Forb		S2			
Platanthera flava var. *herbiola*	Southern rein orchid	Forb	S1				

(continued)

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Platanthera grandiflora	Large purple fringed orchid	Forb		S2		S1	
Platanthera hookeri	Hooker's orchid	Forb			S1		
Platanthera huronensis	Huron green orchid	Forb			S1		
Platanthera lacera	Green-fringe orchis	Forb	S3				
Platanthera leucophaea	Prairie fringed orchid	Forb				S1	
Platanthera peramoena	Purple fringeless orchid	Forb	S1	S1	S2	S2	
Platanthera psycodes	Small purple-fringe orchid	Forb					S1
Platanthera shriveri	Shriver's frilly orchid	Forb				S1	
Pleopeltis polypodioides	Resurrection fern	Cryptogam		S3			
Pluchea camphorata	Marsh fleabane	Forb		S1			
Pluchea foetida (var. *foetida*)	Stinking camphorweed	Forb	S3				
Pluchea odorata	Shrubby camphorweed	Forb			S1		
Poa alsodes	Grove meadowgrass	Graminoid		S2			
Poa autumnalis	Autumn bluegrass	Graminoid	S3		S1		
Poa cuspidata	Early bluegrass	Graminoid	S2				
Poa languida	Drooping bluegrass	Graminoid			S2		
Poa paludigena	Bog bluegrass	Graminoid	S1.1		S3	S2	S1
Poa palustris	Fowl bluegrass	Graminoid				S1S2	
Poa saltuensis	Drooping bluegrass	Graminoid		S1		S2	S1
Podostemum ceratophyllum	Threadfoot	Forb	S1	S3			
Pogonia ophioglossoides	Rose pogonia	Forb	S2	S3			S2
Polanisia dodecandra (ssp. *dodecandra*)	Common clammyweed	Forb		S1		S2	
Polemonium reptans (var. *reptans*)	Greek valerian	Forb	S3				
Polemonium vanbruntiae	Jacob's-ladder	Forb		S2	S1		S2
Polygala cruciata	Cross-leaved milkwort	Forb		S2	S1		
Polygala cruciata var. *aquilonia*	Crossleaf milkwort	Forb	S2				S1
Polygala curtissii	Curtis's milkwort	Forb			S1		S2
Polygala incarnata	Pink milkwort	Forb	S1	S2S3			

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Polygala nuttallii	Nuttall's milkwort	Forb			S3		
Polygala polygama	Racemed milkwort	Forb		S1	S1S2		
Polygala senega	Seneca snakeroot	Forb		S2			
Polygonatum biflorum var. *commutatum*	Giant Solomon's-seal	Forb	S3				
Polygonella articulata	Eastern Jointweed	Forb			S1		
Polygonella polygama	October-flower	Forb				S1	
Polygonum amphibium var. *emersum*	Water smartweed	Forb					S2S3
Polygonum amphibium var. *stipulaceum*	Water smartweed	Forb					S1
Polygonum careyi	Carey's smartweed	Forb			S1		
Polygonum cilinode	Fringed bindweed	Forb		S3			
Polygonum glaucum	Seabeach knotweed	Forb	S1	S1		S1S2	
Polygonum ramosissimum	Bushy knotweed	Forb	S2				
Polypodium virginianum	Virginia polypody	Cryptogam	S3				
Polystichum braunii	Braun's holly fern	Cryptogam			S1		
Populus balsamifera (ssp. *balsamifera*)	Balsam poplar	Tree			S1		S1
Populus tremuloides	Quaking aspen	Tree				S2	
Porteranthus stipulatus	American ipecac	Forb				S1	
Portulaca smallii	Small's purslane	Forb				S1	
Potamogeton amplifolius	Large-leaf pondweed	Forb				S1S2	
Potamogeton confervoides	Tuckerman's pondweed	Forb			S2		
Potamogeton foliosus	Leafy pondweed	Forb		S1			
Potamogeton friesii	Fries' pondweed	Forb			S1		
Potamogeton gramineus	Grassy pondweed	Forb			S1		
Potamogeton hillii	Hill's pondweed	Forb			S1	S1	
Potamogeton illinoensis	Illinois pondweed	Forb		S1			S2
Potamogeton natans	Floating pondweed	Forb	S1				
Potamogeton oakesianus	Oakes' pondweed	Forb			S1S2	S2	
Potamogeton obtusifolius	Blunt-leaved pondweed	Forb			S1		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Potamogeton perfoliatus	Clasping-leaved pondweed	Forb		S2			
Potamogeton praelongus	White-stemmed pondweed	Forb			S1		
Potamogeton pulcher	Spotted pondweed	Forb			S1		S1
Potamogeton pusillus	Slender pondweed	Forb		S1			
Potamogeton pusillus var. *tenuissimus*	Slender pondweed	Forb					S1
Potamogeton richardsonii	Red-head pondweed	Forb			S3		
Potamogeton spirillus	Spiral pondweed	Forb		S1		S1	S2
Potamogeton strictifolius	Straight-leaf pondweed	Forb				S1	
Potamogeton tennesseensis	Tennessee pondweed	Forb			S1	S1	S2
Potamogeton vaseyi	Vasey's pondweed	Forb			S1		
Potamogeton zosteriformis	Flatstem pondweed	Forb		S1	S2S3	S1	
Potentilla anserina	Silverweed	Forb			S3		
Potentilla arguta	Tall cinquefoil	Forb				S1	
Potentilla fruticosa	Shrubby cinquefoil	Forb			S1		
Potentilla paradoxa	Bushy cinquefoil	Forb			S1		
Potentilla tridentata	Three-toothed cinquefoil	Forb			S1		
Prenanthes autumnalis	Slender rattlesnake-root	Forb		S1		S2	
Prenanthes crepidinea	Nodding rattlesnake-root	Forb					S1
Prenanthes serpentaria	Lion's-foot	Forb			S3		
Prosartes maculata	Nodding mandarin	Forb					S1
Prunus alleghaniensis (var. *alleghaniensis*)	Alleghany plum	Tree		S2	S2S3		S3
Prunus angustifolia (var. *angustifolia*)	Chickasaw plum	Tree					S1
Prunus maritima	Beach plum	Tree	S3	S1	S1	S1	
Prunus nigra	Canada plum	Tree				S1	
Prunus pumila var. *depressa*	Dwarf sand cherry	Tree			S1		S1
Prunus pumila var. *susquehanae*	Susquehanna cherry	Tree			S2	S1	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a] DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Pseudognaphalium helleri	Catfoot	Forb				S1	
Pseudognaphalium macounii	Winged cudweed	Forb				S1	
Ptelea trifoliata	Common hop-tree	Tree		S3	S2		
Ptilimnium fluviatile	Harperella	Forb					S1
Ptilimnium nodosum	Harperella	Forb		S1		S1	
Puccinellia fasciculata	Salt marsh goosegrass	Graminoid				S1	
Pycnanthemum beadlei	Beadle's mountain mint	Forb					S1
Pycnanthemum clinopodioides	Basil mountain-mint	Forb				S1	
Pycnanthemum incanum (var. *incanum*)	Hoary mountain-mint	Forb	S1				
Pycnanthemum loomisii	Loomis' mountain-mint	Forb					S2
Pycnanthemum montanum	Appalachian mountain-mint	Forb				S2	
Pycnanthemum muticum	Blunt mountain-mint	Forb	S3				S1
Pycnanthemum setosum	Awned mountain-mint	Forb	S3			S1	
Pycnanthemum torrei	Torrey's Mountain-mint	Forb	S1	S1	S1		S1
Pycnanthemum verticillatum (var. *verticillatum*)	Whorled mountain-mint	Forb	S3	S1			
Pycnanthemum virginianum	Virginia mountain-mint	Forb		S2			
Pyrola americana	Round-leaf shinleaf	Forb	S2				
Pyrola chlorantha		Forb			S1		
Pyrola elliptica	Elliptic shinleaf	Forb	S2			S2	
Pyrularia pubera	Buffalo nut	Forb			S3		
Pyxidanthera barbulata	Flowering pixiemoss	Forb				S1	
Quercus falcata	Southern red oak	Tree			S1		

(continued)

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Quercus hemisphaerica	Darlington's oak	Tree				S1	
Quercus incana	Bluejack oak	Tree				S2	
Quercus lyrata	Overcup oak	Tree	S2				
Quercus macrocarpa	Mossy-cup oak	Tree		S1		S1	
Quercus marilandica (var. *marilandica*)	Blackjack oak	Tree	S3				
Quercus michauxii	Swamp chestnut oak	Tree			S1		
Quercus phellos	Willow oak	Tree			S2		
Quercus prinoides	Dwarf chinqua-pin oak	Tree	S1	S3		S1	
Quercus shumardii	Shumard's oak	Tree		S2	S2		S2
Ranunculus allegheniensis	Mountain crowfoot	Forb		S3			
Ranunculus ambigens	Water-plantain buttercup	Forb	S1.1		S3	S1	
Ranunculus aquatilis var. *diffusus*	White water-crowfoot	Forb			S3	S1	
Ranunculus caricetorum	Hispid buttercup	Forb	S1				
Ranunculus fascicularis	Early buttercup	Forb		S1	S1S2		
Ranunculus flabellaris	Yellow water-crowfoot	Forb	S1.1	S1	S2		
Ranunculus flammula var. *filiformis*	Creeping spearwort	Forb					S1
Ranunculus hederaceus	Long-stalked crowfoot	Forb		S1	S1		
Ranunculus hispidus	Bristly buttercup	Forb	S1.1				
Ranunculus laxicaulis	Mississippi buttercup	Forb				S1	
Ranunculus macounii	Macoun buttercup	Forb					S1
Ranunculus pensylvanicus	Bristly crowfoot	Forb					S1
Ranunculus pusillus (var. *pusillus*)	Low spearwort	Forb	S3		S1		S1
Ranunculus sceleratus (var. *sceleratus*)	Crused crowfoot	Forb					S2
Ranunculus septentrionalis	Shining hispid buttercup	Forb	S1.1				
Ranunculus trichophyllus	White water-crowfoot	Forb		S1			
Ratibida pinnata	Gray-headed prairie coneflower	Forb			S1		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Rhamnus alnifolia	Alder-leaved buckthorn	Tree				S1	S1S2
Rhamnus lanceolata	Lance-leaved buckthorn	Tree			S1		
Rhamnus lanceolata var. *glabrata*	Smooth lanceleaved buckthorn	Tree				S1	
Rhamnus lanceolata (ssp. *lanceolata*)	Lance-leaved buckthorn	Tree					S1
Rhexia aristosa	Awned meadow beauty	Forb	S1				
Rhexia mariana (var. *mariana*)	Maryland meadow beauty	Forb			S1		S1
Rhexia petiolata	Ciliate meadow beauty	Forb				S1	
Rhododendron arborescens	Smooth azalea	Shrub		S3		S2	
Rhododendron atlanticum	Dwarf azalea	Shrub	S3		S1		
Rhododendron calendulaceum	Flame azalea	Shrub		S1			
Rhododendron viscosum	Swamp azalea	Shrub					S1
Rhus michauxii	Michaux's sumac	Tree				S1	
Rhynchosia tomentosa	Hairy snoutbean	Forb	S1	S2			
Rhynchospora alba	White beakrush	Graminoid	S2	S3		S2	
Rhynchospora capillacea	Capillary beaked-rush	Graminoid			S1	S1	
Rhynchospora cephalantha var. *attenuata*	Small capitate beakrush	Graminoid				S2	
Rhynchospora cephalantha (var. *cephalantha*)	Capitate beakrush	Graminoid	S2	S1			
Rhynchospora colorata	White-topped sedge	Graminoid				S1	
Rhynchospora corniculata	Short-bristle beakrush	Graminoid	S2				
Rhynchospora debilis	Savannah beakrush	Graminoid				S1	
Rhynchospora fascicularis	Fasciculate beakrush	Graminoid				S2	
Rhynchospora filifolia	Thread-leaved beakrush	Graminoid	S1.1				
Rhynchospora fusca	Brown beakrush	Graminoid	S2				S1
Rhynchospora globularis	Grass-like beakrush	Graminoid		S1			
Rhynchospora glomerata	Clustered beakrush	Graminoid	S2	S3			
Rhynchospora harperi	Harper's beakrush	Graminoid	S1	S1			

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Rhynchospora harveyi	Harvey's beakrush	Graminoid				S1	
Rhynchospora inundata	Drowned beakrush	Graminoid	S1	S1			
Rhynchospora macro-stachya (var. *macrostachya*)	Tall horned rush	Graminoid				S2	
Rhynchospora microcephala	Tiny-headed beakrush	Graminoid	S2	S2			
Rhynchospora nitens	Short-beaked beakrush	Graminoid	S1	S1		S1	
Rhynchospora oligantha	Few-flowered beakrush	Graminoid				S1	
Rhynchospora rariflora	Few-flowered beakrush	Graminoid	S1.1				
Rhynchospora recognita	Tall horned rush	Graminoid	S2	S2	S1		S2
Rhynchospora scirpoides	Long-beaked beakrush	Graminoid	S2	S2		S1	
Rhynchospora stenophylla	Coastal bog beaksedge	Graminoid				S1	
Rhynchospora torreyana	Torrey's beakrush	Graminoid	S2	S2			
Rhynchospora wrightiana	Wright's beakrush	Graminoid				S1	
Ribes americanum	Wild black currant	Shrub				S1	
Ribes cynosbati	Prickly gooseberry	Shrub		S3			
Ribes glandulosum	Skunk currant	Shrub		S3			
Ribes hirtellum	Low wild gooseberry	Shrub		S1			S1
Ribes lacustre	Swamp currant	Shrub			S1		S2
Ribes missouriense	Missouri gooseberry	Shrub			S1		S1
Ribes triste	Swamp red currant	Shrub			S2		S1
Rorippa sessiliflora	Stalkless yellowcress	Forb				S1	S1
Rosa acicularis ssp. *sayi*	Prickly rose	Shrub					S1
Rosa blanda	Smooth rose	Shrub		S1			S2
Rosa carolina	Carolina rose	Shrub	S3				
Rosa setigera	Prairie rose	Shrub			S1	S1	
Rosa virginiana	Virginia rose	Shrub			S1		
Rotala ramosior	Tooth-cup	Forb			S3		
Rubus cuneifolius	Sand blackberry	Shrub			S1		
Rubus idaeus ssp. *strigosus*	Red raspberry	Shrub				S2	
Rubus odoratus	Purple-flowering raspberry	Shrub	S1				

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Rubus pubescens (var. *pubescens*)	Dwarf red raspberry	Shrub					S1
Rudbeckia fulgida	Orange coneflower	Forb		S3	S3		S2
Rudbeckia heliopsidis	Sun-facing coneflower	Forb				S1	
Rudbeckia triloba	Thin-leaved coneflower	Forb		S3			
Rudbeckia triloba var. *pinnatiloba*	Pinnate-lobed coneflower	Forb				S1	
Ruellia caroliniensis	Carolina wild petunia	Forb	S3				
Ruellia humilis	Hairy wild-petunia	Forb		S1	S1		S1
Ruellia pedunculata	Stalked wild-petunia	Forb			S1		
Ruellia purshiana	Pursh's ruellia	Forb		S1			
Ruellia strepens	Rustling wild-petunia	Forb		S1	S2		
Rumex altissimus	Tall dock	Forb	S1	S1			
Sabatia angularis	Square-stemmed rose pink	Forb	S3				
Sabatia campanulata	Slender marsh pink	Forb	S1	S1		S2	
Sabatia difformis	Two-formed pink	Forb	S1	S1		S1	
Sabatia dodecandra	Large marsh pink	Forb	S1	S3			
Sabatia stellaris	Sea pink	Forb	S3				
Saccharum baldwinii	Narrow plumegrass	Graminoid		S1			
Saccharum brevibarbe var. *contortum*	Bent-awn plumegrass	Graminoid	S2				
Saccharum coarctatum	Bunched plumegrass	Graminoid	S2				
Saccharum contortum	Bent-awn plumegrass	Graminoid		S3S4			
Sacciolepis striata	Gibbous grass	Graminoid	S1	S1			
Sagittaria calycina var. *spongiosa*	Arrowhead	Forb			S1	S1	
Sagittaria calycina (var. *calycina*)	Long-lobe arrowhead	Forb	S3	S2		S1	S2
Sagittaria cuneata	Wapatum arrowhead	Forb			S1		
Sagittaria engelmanniana	Engelmann's arrowhead	Forb	S2	S2			
Sagittaria graminea	Grassleaf arrowhead	Forb	S2				
Sagittaria lancifolia var. *media*	Bull-tongue arrowhead	Forb	S1.1				

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Sagittaria latifolia var. *pubescens*	Hairy arrowhead	Forb	S3				
Sagittaria rigida	Sessile-fruited arrowhead	Forb		S1		S1	
Sagittaria spatulata	Tidal arrowhead	Forb	S1				
Sagittaria subulata	Strap-leaf arrowhead	Forb	S2		S3		
Salicornia bigelovii	Dwarf glasswort	Forb	S1				
Salix amygdaloides	Peach-leaved willow	Tree					S1
Salix candida	Hoary willow	Shrub			S1		
Salix caroliniana	Carolina willow	Tree		S3	S1		
Salix discolor	Glaucous willow	Tree				S1	S2
Salix exigua	Sandbar willow	Tree		S1		S1	
Salix humilis var. *tristis*	Dwarf Prairie willow	Shrub		S1			
Salix lucida (ssp. *lucida*)	Shining willow	Tree					S1
Salix myricoides	Broad-leaved willow	Tree			S2		
Salix pedicellaris	Bog willow	Shrub			S1		
Salix sericea	Silky willow	Tree	S1				
Salix serissima	Autumn willow	Tree			S2		
Samolus parviflorus	Pineland pimpernel	Forb			S3		
Samolus valerandi ssp. *parviflorus*	Water pimpernel	Forb					S2
Sanguisorba canadensis	Canada burnet	Forb		S2		S2	S2S3
Sanicula marilandica	Maryland black snakeroot	Forb	S1.1	S3			
Sanicula trifoliata	Large-fruited snakeroot	Forb	S1.1	S3			
Sarcocornia pacifica	Perennial glasswort	Forb	S3				
Sarracenia flava	Yellow pitcher-plant	Forb				S1	
Sarracenia purpurea var. *venosa*	Southern purple pitcher-plant	Forb				S2	
Sarracenia purpurea (var. *purpurea*)	Northern purple pitcherplant	Forb	S2	S2			
Saxifraga careyana	Carey saxifrage	Forb					S3
Saxifraga caroliniana	Carolina saxifrage	Forb					S1
Saxifraga michauxii	Michaux saxifrage	Forb					S1
Saxifraga micranthidifolia	Lettuce-leaved saxifrage	Forb		S3			
Saxifraga pensylvanica	Swamp saxifrage	Forb					S2

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Scheuchzeria palustris	Pod-grass	Graminoid			S1		
Schizachne purpurascens	Purple oat	Graminoid		S1		S1	S1
Schizachyrium littorale	Seaside bluestem	Graminoid	S3				
Schizachyrium scoparium var. *littorale*	Seaside bluestem	Graminoid			S3		
Schizaea pusilla	Curly-grass fern	Cryptogam	S1.1				
Schoenoplectus acutus (var. *acutus*)	Hard-stemmed bulrush	Graminoid			S2	S1	S2
Schoenoplectus etuberculatus	Canby's bulrush	Graminoid	S1	S1			
Schoenoplectus fluviatilis	River bulrush	Graminoid			S3	S2	
Schoenoplectus novae-angliae	Salt-marsh bulrush	Graminoid		S2			
Schoenoplectus purshianus (var. *purshianus*)	Bristled weakstalk bulrush	Graminoid	S2				S3
Schoenoplectus purshianus var. *williamsii*	Bristleless weakstalk bulrush	Graminoid	S2				
Schoenoplectus smithii	Smith's bulrush	Graminoid			S1		
Schoenoplectus smithii var. *setosus*	Smith's bristled bulrush	Graminoid	S1.1				
Schoenoplectus subterminalis	Water clubrush	Graminoid	S2	S1	S3	S1S2	
Schoenoplectus torreyi	Torrey's bulrush	Graminoid			S1	S1	
Scirpus ancistrochaetus	Northeastern bulrush	Graminoid		S1	S3	S2	S1
Scirpus atrocinctus	Black-girdle bulrush	Graminoid					S3
Scirpus atrovirens	Bulrush	Graminoid	S1				
Scirpus expansus	Red-stem bulrush	Graminoid	S2	S3			
Scirpus flaccidifolius	Schuyler reclining bulrush	Graminoid				S1	
Scirpus microcarpus	Small-fruit bulrush	Graminoid					S3
Scirpus pedicellatus	Stalked bulrush	Graminoid			S1		
Scirpus pendulus	Pendulous bulrush	Graminoid	S2	S3			
Scleria ciliata (var. *ciliata*)	Fringed nutrush	Graminoid				S1	
Scleria minor	Slender nutrush	Graminoid		S1		S2	
Scleria muehlenbergii	Muhlenberg's nutrush	Graminoid	S1	S1S2	S1		
Scleria nitida	Shining nutrush	Graminoid		S1			
Scleria oligantha	Little-headed nutrush	Graminoid					S1

(continued)

			State conservation status rank[a]				
Scientific name	Common name	Life form	DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Scleria pauciflora var. *caroliniana*	Hairy few-flowered nutrush	Graminoid	S1			S1	
Scleria pauciflora (var. *pauciflora*)	Smooth few-flowered nutrush	Graminoid	S1	S3	S2		S1
Scleria reticularis	Reticulated nutrush	Graminoid	S3	S2S3			
Scleria triglomerata	Tall nutrush	Graminoid	S2	S1S2			S2
Scleria verticillata	Whorled nutrush	Graminoid		S1	S1	S2	
Sclerolepis uniflora	One-flowered bog button	#N/A	S2	S2		S1	
Scrophularia lanceolata	Hare figwort	Forb		S3			
Scutellaria galericulata	Hooded skullcap	Forb	S1	S1		S1	S1
Scutellaria incana	Downy skullcap	Forb		S3		S2	
Scutellaria leonardii	Leonard's skullcap	Forb		S2			
Scutellaria nervosa	Veined skullcap	Forb		S1			
Scutellaria ovata (ssp. *ovata*)	Heart-leaved skullcap	Forb		S3			S1
Scutellaria parvula (var. *parvula*)	Small skullcap	Forb				S1	
Scutellaria saxatilis	Rock skullcap	Forb		S1	S1		S2
Scutellaria serrata	Showy skullcap	Forb		S3	S1		
Sedum glaucophyllum	Cliff stonecrop	Forb		S1			
Sedum rosea	Roseroot stonecrop	Forb			S1		
Sedum telephioides	Allegheny stonecrop	Forb			S3		
Sedum ternatum	Woodland stonecrop	Forb	S2				
Selaginella apoda	Meadow spikemoss	Bryophyte	S3				
Senna marilandica	Wild senna	Subshrub			S3		
Sericocarpus linifolius	Narrow-leaved white-topped aster	Forb			S1		S1
Sesuvium maritimum	Sea-purslane	Forb		S1			
Seymeria cassioides	Seymeria	#N/A				S1S2	
Shepherdia canadensis	Canada buffalo-berry	#N/A			S1		
Sibbaldiopsis tridentata	Three-toothed cinquefoil	Forb				S2	S2
Sida elliottii	Elliott sida	Forb				S1	
Sida hermaphrodita	Virginia mallow	Forb		S1	S2	S1	S3
Sideroxylon lycioides	Buckthorn	#N/A	S1.1				
Silene caroliniana ssp. *wherryi*	Wherry's catchfly	Forb					S1

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Silene caroliniana var. *pensylvanica*	Northern wild pink	Forb	S1				
Silene nivea	Snowy campion	Forb		S1		S1	S1
Silene ovata	Ovate catchfly	Forb				S1	
Silene rotundifolia	Roundleaf catchfly	Forb				S2	S1
Silene virginica	Fire-pink	Forb	S1				
Silene virginica var. *robusta*	Robust fire pink	Forb					S1
Silphium compositum var. *reniforme*	Kidney-leaf rosin-weed	Forb					S1
Silphium perfoliatum var. *connatum*	Virginia cup-plant	Forb					S1
Silphium terebinthinaceum	Prairie rosinweed	Forb				S1	
Silphium trifoliatum	Three-leaved rosinweed	Forb		S3			
Sisyrinchium albidum	White blue-eyedgrass	Forb				S2	
Sisyrinchium atlanticum	Eastern blue-eyed Grass	Forb	S3		S1		
Sisyrinchium fuscatum (*arenicola*)	Coastal Plain blue-eyed grass	Forb		S1			
Sisyrinchium mucronatum	Michaux's blue-eyed-grass	Forb	S1.1				
Smallanthus uvedalius	Yellow-flowered leafcup	Forb	S3		S3		
Smilacina stellata	Star-flowered false Solomon's-seal	Forb		S1			
Smilax bona-nox	Saw greenbrier	Vine	S1	S3			
Smilax ecirrata	Upright greenbrier	Vine				S1	
Smilax hispida	weak-prickle greenbrier	Vine	S1				
Smilax pseudochina	long-stalk greenbrier	Vine	S2	S2			
Smilax walteri	Walter's greenbrier	Vine	S3				
Solidago arguta (var. *arguta*)	cutleaf goldenrod	Forb	S1				
Solidago arguta var. *harrisii*	Harris' golden-rod	Forb		S3	S1		S3

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Solidago curtisii	Curtis' goldenrod	Forb		S1	S1		
Solidago erecta	Slender golden-rod	Forb			S1		
Solidago faucibus	Gorge goldenrod	Forb					S1
Solidago gracillima	Southern bog goldenrod	Forb				S2	
Solidago latissimifolia	Elliott's goldenrod	Forb	S1	S3		S2	
Solidago patula var. *strictula*		Forb				S1	
Solidago patula (var. *patula*)	Roundleaf goldenrod	Forb	S3	S3			S1
Solidago racemosa	Sticky goldenrod	Forb				S1	
Solidago randii	Rand’s goldenrod	Forb				S2S3	
Solidago rigida	Hard-leaved goldenrod	Forb			S1		
Solidago roanensis	Tenessee golden-rod	Forb			S2		
Solidago rupestris	Riverbank goldenrod	Forb				S1	
Solidago simplex ssp. *randii*	Mountain goldenrod	Forb					S1
Solidago simplex ssp. *randii* var. *racemosa*	Sticky goldenrod	Forb			S1		S2
Solidago simplex var. *racemosa*	Riverbank goldenrod	Forb		S1			
Solidago speciosa (var. *speciosa*)	Showy goldenrod	Forb		S2	S2		
Solidago tarda	Late goldenrod	Forb	S1.1				
Solidago tortifolia	Leafy pinewoods goldenrod	Forb				S1	
Solidago uliginosa (var. *uliginosa*)	Bog goldenrod	Forb	S1.1	S3	S2	S2	
Solidago ulmifolia (var. *ulmifolia*)	Elm-leaf goldenrod	Forb	S2				
Sorbus americana	American mountain-ash	Tree		S3			
Sorbus decora	Showy mountain-ash	Tree			S1		
Sorghastrum elliottii	Long-bristled Indian-grass	Graminoid		S1			
Sparganium androcladum	Branching bur-reed	Forb		S3	S1		S2S3
Sparganium angustifolium	Narrow-leaf bur-reed	Forb			S2		S1S2
Sparganium chlorocarpum	Narrow-leaf bur-reed	Forb				S1	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Sparganium erectum	Green-fruited bur-reed	Forb		S3			
Sparganium eurycarpum	Broad-fruited bur-reed	Forb		S3			
Spartina pectinata	Fresh water cordgrass	Graminoid	S1.1			S2	
Spermacoce glabra	Buttonweed	#N/A		S1		S1	S1
Sphenopholis pensylvanica	Swamp wedgescale grass	Graminoid	S1	S2			
Spiraea alba (var. *alba*)	Narrowleaf white spiraea	Shrub	S1				
Spiraea alba var. *latifolia*	Broad-leaf white spiraea	Shrub	S1				
Spiraea betulifolia	Dwarf spiraea	Shrub		S3	S1		
Spiraea virginiana	Virginia spiraea	Shrub				S1	S1
Spiranthes casei	Case's ladies'-tresses	Forb			S1		
Spiranthes lacera var. *gracilis*	Southern slender ladies'-tresses	Forb	S2				
Spiranthes lacera (var. *lacera*)	Northern slender ladies'-tresses	Forb					S1
Spiranthes lucida	Wide-leaved ladies' tresses	Forb		S1	S3	S1	S1S2
Spiranthes magnicamporum	Great Plains ladies'-tresses	Forb				S1	
Spiranthes ochroleuca	Yellow Nodding ladies' tresses	Forb		S1		S1	
Spiranthes ovalis	October ladies'-tresses	Forb			S1		
Spiranthes ovalis var. *erostellata*	Oval ladies'-tresses	Forb					S1
Spiranthes praecox	Grass-leaved ladies' tresses	Forb		S1			
Spiranthes romanzoffiana	Hooded Ladies'-tresses	Forb			S1		
Spiranthes tuberosa	Little Ladies'-tresses	Forb		S3	S1		S3
Spiranthes vernalis	Spring Ladies'-tresses	Forb	S2		S1		S3
Sporobolus asper	Long-leaved rushgrass	Graminoid		S1			
Sporobolus clandestinus	Rough dropseed	Graminoid	S1	S2	S1		S1

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Sporobolus compositus (var. *compositus*)	Longleaf dropseed	Graminoid				S1S2	
Sporobolus heterolepis	Northern dropseed	Graminoid		S1	S1	S1	
Sporobolus junceus	Purple dropseed	Graminoid				S1	
Sporobolus neglectus	Small dropseed	Graminoid				S2	
Stachys arenicola	Marsh hedgenettle	Forb				S1	
Stachys aspera	Rough hedge-nettle	Forb		S1		S2	S1
Stachys cordata	Nuttall's hedge-nettle	Forb			S1		
Stachys eplingii	Epling's hedge-nettle	Forb				S1	S1
Stachys latidens	Broad-toothed hedge-nettle	Forb		S1			
Stachys nuttallii	Nuttall's hedge-nettle	Forb		S1			S3
Stachys tenuifolia	Smooth hedge-nettle	Forb	S1.1				S3
Steinchisma hians	Gaping panic grass	#N/A				S1	
Stellaria alsine	Trailing stitchwort	Forb	S2	S1			
Stellaria borealis	Mountain starwort	Forb					
Stellaria borealis (ssp. *borealis*)	Northern stitchwort	Forb			S1S2		S1
Stenanthium gramineum (var. *gramineum*)	Featherbells	Forb		S1	S1S2		S2S3
Stenanthium gramineum var. *micranthum*	Tiny-flowered featherbells	Forb					S1
Stenanthium gramineum var. *robustum*	Stout featherbells	Forb					S1S2
Stenanthium leimanthoides	Death-camas	Forb		S1			
Stewartia ovata	Mountain camellia	#N/A				S2	
Stillingia sylvatica (ssp. *sylvatica*)	Queen's delight	#N/A				S1	
Stipulicida setacea (var. *setacea*)	Pineland scaly-pink	#N/A				S1	
Streptopus amplexifolius	White twisted-stalk	#N/A			S1	S1	
Streptopus roseus	Rose twisted-stalk	#N/A		S1S2			
Strophostyles umbellata	Wild bean	#N/A			S2		
Stylophorum diphyllum	Celandine poppy	Forb				S2	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Stylosanthes biflora	Pencilflower	#N/A			S2		
Suaeda linearis	Narrow-leaf seepweed	#N/A	S3	S3			
Sullivantia sullivantii	Sullivantia	#N/A				S1	
Swertia caroliniensis	American columbo	#N/A			S1		
Symphoricarpos albus (var. *albus*)	Snowberry	Forb		S1		S2	S2
Symphyotrichum boreale	Rush aster	Forb			S1		S1
Symphyotrichum concolor	Silvery aster	Forb		S1			
Symphyotrichum cordifolium	Heart-leaf aster	Forb	S3				
Symphyotrichum depauperatum	Serpentine aster	Forb		S1	S2		
Symphyotrichum drummondii	Drummond aster	Forb		S1			
Symphyotrichum dumosum	Bushy aster	Forb			S1		
Symphyotrichum elliottii	Elliott's aster	Forb				S1	
Symphyotrichum ericoides	White heath aster	Forb			S3		
Symphyotrichum laeve var. *concinnum*	Narrow-leaved smooth blue aster	Forb	S1				S2
Symphyotrichum laeve (var. *laeve*)	Smooth blue aster	Forb	S1				
Symphyotrichum novi-belgii	New York aster	Forb			S2		S2S3
Symphyotrichum ontarionis (var. *ontarionis*)	Ontario aster	Forb				S1	
Symphyotrichum praealtum	Willow aster	Forb		S1	S3		
Symphyotrichum praealtum var. *angustior*	Willow aster	Forb				S1	
Symphyotrichum pratense	Silky aster	Forb				S1	
Symphyotrichum prenanthoides	Crooked-stem aster	Forb	S1				
Symphyotrichum shortii	Short's aster	Forb		S3		S1	
Symplocos tinctoria	Sweetleaf	#N/A		S3			
Synandra hispidula	Guyandotte beauty	#N/A				S2	S2
Syntrichia ammonsiana	Ammons's tortula						S1
Taenidia integerrima	Yellow pimpernel	Forb	S1.1				

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Taenidia montana	Mountain pimpernel	Forb		S2	S1		S3
Talinum teretifolium	Fameflower	Forb		S1			S1
Taxodium ascendens	Pond cypress	Tree				S1	
Taxodium distichum	Bald cypress	Tree	S2				
Taxus canadensis	American yew	Shrub		S2	S3S4		S2S3
Tephrosia spicata	Southern goat's rue	Forb		S1			
Tetragonotheca helianthoides	Pineland squarehead	#N/A				S1	
Teucrium canadense var. *virginicum*	American germander	Forb	S3				
Thalictrum clavatum	Mountain meadow-rue	Forb					S2
Thalictrum coriaceum	Thick-leaved meadow-rue	Forb			S2		
Thalictrum dasycarpum	Purple meadow-rue	Forb			S1		
Thalictrum dioicum	Early meadow-rue	Forb	S1				
Thalictrum macrostylum	Small-leaved meadow-rue	Forb				S1	
Thalictrum revolutum	Waxleaf meadow-rue	Forb	S1				
Thaspium barbinode	Hairy-jointed meadow-parsnip	Forb	S1				
Thaspium trifoliatum	Purple meadow-parsnip	Forb		S1			
Thelypteris simulata	Bog fern	Cryptogam	S2	S2		S1S2	S1
Thuja occidentalis	Arbor-vitae	#N/A		S1			S2
Tillandsia usneoides	Spanish moss	#N/A				S2	
Tipularia discolor	Cranefly orchid	Forb			S3		
Torreyochloa pallida var. *fernaldii*	Fernald's mannagrass	Graminoid		S1			S2
Torreyochloa pallida (var. *pallida*)	Pale mannagrass	Graminoid		S3			S1
Toxicodendron pubescens	Poison oak	Shrub	S2				
Toxicodendron rydbergii	Giant poison-ivy	Shrub			S1	S1	
Toxicodendron vernix	Poison sumac	Tree/Shrub	S3				S2
Trachelospermum difforme	Climbing dogbane	#N/A	S1	S1			
Tradescantia virginiana	Virginia spiderwort	Forb	S3				
Trautvetteria caroliniensis	Carolina tassel-rue	Forb		S3			

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Triadenum fraseri	Northern marsh St. John's-wort	Forb	S1.1			S1	
Triadenum tubulosum	Large Marsh St. John's-wort	Forb		S1		S1	S1
Triadenum walteri	Walter St. John's-wort	Forb					S1
Triantha glutinosa	Sticky false-asphodel	Forb				S1	S1
Triantha racemosa	Coastal False-asphodel	Forb				S1	
Trichomanes boschianum	Filmy fern	Cryptogam				S1	S1
Trichophorum planifolium	Bashful bulrush	Graminoid	S2	S2S3			S1
Trichostema brachiatum	False pennyroyal	Forb		S3			
Trichostema setaceum	Narrowleaf bluecurls	Forb	S1	S1	S1	S2	S2
Tridens chapmanii	Chapman's purple-top	Graminoid	S1.1				
Tridens flavus var. *chapmanii*	Chapman's redtop	Graminoid		S1			
Trientalis borealis (subsp. *borealis*)	Northern starflower	Forb	S1				
Trifolium calcaricum	Running glade clover	Forb				S1	
Trifolium reflexum	Buffalo clover	Forb				S1	S1
Trifolium stoloniferum	Running buffalo clover	Forb					S3
Trifolium virginicum	Kate's mountain clover	Forb		S2S3	S1		S3
Triglochin striata	Three-ribbed arrow-grass	Graminoid		S1			
Trillium cernuum	Nodding trillium	Forb	S2	S3	S2	S2	S1
Trillium erectum	Ill-scented trillium	Forb	S1.1				
Trillium flexipes	Drooping trillium	Forb		S1	S2	S1	S2
Trillium nivale	Snow trillium	Forb		S1	S3	S1	S2
Trillium pusillum var. *virginianum*	Least trillium	Forb		S2		S2	S1
Triosteum angustifolium (var. *angustifolium*)	Yellowleaf tinker's-weed	#N/A	S1	S1	S1		
Triosteum aurantiacum (var. *aurantiacum*)	Coffee tinker's-weed	#N/A	S1				
Triosteum perfoliatum	Perfoliate tinker's-weed	#N/A	S1.1				

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Triphora trianthophora	Nodding pogonia	Forb	S1	S1		S1	S2
Triplasis purpurea	Purple sandgrass	Graminoid			S1		
Tripsacum dactyloides	Eastern gamma-grass	Graminoid			S1		
Trisetum spicatum	Narrow false oats	Graminoid			S1	S1	
Trollius laxus	Spreading globeflower	Forb			S1		
Tsuga canadensis	Eastern hemlock	Tree	S1				
Typha domingensis	Southern cattail	Forb		S3			
Utricularia cornuta	Horned bladderwort	Forb			S2		
Utricularia geminiscapa	Hidden-fruited bladderwort	Forb	S3				S1
Utricularia gibba	Humped bladderwort	Forb	S3				S2
Utricularia inflata	Large swollen bladderwort	Forb	S1	S1			
Utricularia intermedia	Flat-leaved bladderwort	Forb			S2		
Utricularia juncea	Southern bladderwort	Forb	S2			S2	
Utricularia macrorhiza	Greater bladderwort	Forb					S1
Utricularia purpurea	Purple bladderwort	Forb	S1	S1		S2	
Utricularia resupinata	Reversed bladderwort	Forb	S2	S1			
Utricularia striata	Fibrous bladderwort	Forb	S3	S1		S1	
Utricularia subulata	Zigzag bladderwort	Forb	S2	S3			
Uvularia grandiflora	Large-flowered bellwort	Forb		S1			
Uvularia pudica	Mountain bellwort	Forb			S3		
Vaccinium crassifolium	Creeping blueberry	Shrub				S1	
Vaccinium macrocarpon	Large cranberry	Shrub	S3	S3		S2	S3
Vaccinium myrtilloides	Velvetleaf blueberry	Shrub		S3		S1S2	
Vaccinium oxycoccos	Small cranberry	Shrub		S2			S3
Valeriana pauciflora	Valerian	Forb		S1		S2	
Valerianella chenopodiifolia	Goose-foot cornsalad	Forb		S1			
Vallisneria americana	Tape-grass	Forb	S3				
Veratrum virginicum	Virginia bunchflower	Forb	S2		S1		

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Verbena scabra	Sandpaper vervain	Forb				S2	
Vernonia glauca	Tawny ironweed	Forb	S3		S1		S1
Veronica americana	American speedwell	Forb	S1				
Veronica scutellata	Marsh speedwell	Forb		S1		S1	S2
Veronicastrum virginicum	Culver's-root	Forb	S1				
Viburnum lentago	Nannyberry	Shrub	S1.1	S1			S1S2
Viburnum nudum	Possum-haw	Shrub			S1		
Viburnum opulus var. *americanum*	Highbrush cranberry	Shrub					S1
Viburnum rafinesquianum	Downy arrow-wood	Shrub	S1				S2
Viburnum rufidulum	Rusty blackhaw	Shrub					S1
Viburnum trilobum	Highbush-cranberry	Shrub			S1S2		
Vicia americana (ssp. *americana*)	American purple vetch	#N/A				S1S2	
Viola appalachiensis	Appalachian Blue violet	Forb		S2	S3S4		S3
Viola blanda	Smooth white violet	Forb	S3				
Viola blanda var. *palustriformis*	Large-leaved white violet	Forb		S1			
Viola brittoniana (var. *brittoniana*)	Coast violet	Forb	S3		S1		
Viola labradorica	American dog violet	Forb	S3				
Viola macloskeyi var. *pallens*	Smooth white violet	Forb	S1				
Viola palmata	Palmate-leaved violet	Forb	S3				
Viola pedata	Bird's-foot violet	Forb	S1				
Viola pedatifida	Crowfoot violet	Forb				S1	
Viola renifolia	Kidney-leaved white violet	Forb			S1		
Viola rostrata	Long-spurred violet	Forb		S3			
Viola rotundifolia	Roundleaf violet	Forb	S2				
Viola selkirkii	Great-spurred violet	Forb			S3S4		
Viola septentrionalis	Northern blue violet	Forb					S2
Viola striata	Striped violet	Forb	S3				
Viola tripartita	Three-parted violet	Forb					S1
Viola walteri	Prostrate blue violet	Forb				S2	

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Vitis novae-angliae	New England grape	Vine			S1		
Vitis rupestris	Sand grape	Vine		S1	S1		S2
Vittaria appalachiana	Appalachian gametophyte	#N/A			S2		S1
Wisteria frutescens	American wisteria	Vine				S2	
Wolffia columbiana	Columbia water-meal	Forb		S3		S1	S1
Wolffia papulifera	Water-meal	Forb		S2			
Wolffia punctata	Dotted water-meal	Forb		S2			
Wolffiella gladiata	Bog-mat	Forb			S2		
Woodsia appalachiana	Allegheny cliff fern	Cryptogam					S2
Woodsia ilvensis	Rusty woodsia	Cryptogam		S1			S2
Woodsia obtusa (subsp. *obtusa*)	blunt-lobe woodsia	Cryptogam	S1				
Woodwardia areolata	Netted chainfern	Cryptogam			S2		S2
Xerophyllum asphodeloides	Eastern turkeybeard	#N/A					S1
Xyris caroliniana	Carolina yelloweyed-grass	Forb				S1	
Xyris difformis var. *curtissii*	Curtiss' yelloweyed-grass	Forb				S1	
Xyris fimbriata	Fringed yelloweyed-grass	Forb	S1	S1		S1	
Xyris laxifolia var. *iridifolia*	Irisleaf yelloweyed-grass	Forb				S1	
Xyris platylepis	Tall yelloweyed-grass	Forb				S2	
Xyris smalliana	Small's yelloweyed-grass	Forb	S2	S1			
Xyris torta	Twisted yelloweyed-grass	Forb	S3		S1		S2
Zannichellia palustris	Horned pondweed	Forb	S2				S1
Zanthoxylum americanum	Northern prickly-ash	#N/A		S1			
Zenobia pulverulenta	Dusty zenobia	#N/A				S1	
Zephyranthes atamasca	Atamasco lily	Forb		S1			

(continued)

Scientific name	Common name	Life form	State conservation status rank[a]				
			DE[b]	MD[c]	PA[d]	VA[e]	WV[f]
Zigadenus densus	Dense-flowered camas	Forb				S1	
Zigadenus elegans ssp. *glaucus*	White camas	Forb					S1
Zigadenus glaberrimus	Large-flowered camas	Forb				S1	
Zigadenus glaucus	White camas	Forb			S1		
Zigadenus leimanthoides	Oceanorus	Forb				S1	S3
Zizania aquatica	Indian wild rice	Graminoid			S3		
Zizaniopsis miliacea	Southern wild rice	Graminoid		S1			
Zizia aptera	Wingless alexander	Forb	S1				
Zizia aurea	Golden alexander	Forb	S2	S3			
Zornia bracteata	Viperina	Forb				S1	
Zostera marina var. *stenophylla*	Eel-grass	Graminoid	S1				

Rankings follow those developed by NatureServe (Faber-Langendoen et al. 2009)
[a]Conservation Rankings:
[b]McAvoy 2011
[c]Maryland Department of Natural Resources 2010
[d]Pennsylvania Natural Heritage Program 2011
[e]Virginia Department of Conservation and Recreation 2009
[f]West Virginia Natural Heritage Program 2007
S1—Critically Imperiled: critically imperiled in the jurisdiction because of extreme rarity or because of some factor(s) such as very steep declines making it especially vulnerable to extirpation from the jurisdiction
S2—Imperiled: imperiled in the jurisdiction because of rarity due to very restricted range, very few populations or occurrences, steep declines, or other factors making it very vulnerable to extirpation from the jurisdiction
S3—Vulnerable: vulnerable in the jurisdiction due to a restricted range, relatively few populations or occurrences, recent and widespread declines, or other factors making it vulnerable to extirpation

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Chapter 7
Wetland-Riparian Wildlife of the Mid-Atlantic Region: An Overview

Robert P. Brooks and Thomas L. Serfass

Abstract The Mid-Atlantic Region (MAR) encompasses five major physiographic regions or ecoregions ranging from coastal habitats to mountainous terrain. It includes both unglaciated and glaciated sections. The topography is further differentiated and dissected by several major river basins draining into the Delaware, Susquehanna-Chesapeake, Ohio-Mississippi, and Great Lakes. The biological diversity of the MAR reflects this inherent habitat diversity, with wetland–riparian species well represented. Unfortunately, about 50% of the species of concern in the MAR also are dependent on wetlands, streams, rivers, and riparian areas. This array of diverse taxa is introduced here, with abundant links to prior studies and publications, and websites to guide the reader to these sources. In addition, we feature two "flagship species" that occupy and alter wetland–riparian ecosystems, respectively—river otter and beaver.

7.1 Introduction

The diversity of wildlife species found in wetland and riparian habitats of the Mid-Atlantic Region (MAR) is impressive for a temperate region of the world. A significant portion of this vertebrate biodiversity can be explained by the physiographic and hydrogeomorphic diversity that defines the MAR. Moving from east to west, the MAR spans five major physiographic regions: Coastal Plain, Piedmont, Ridge and Valley, Allegheny Plateau, and Glaciated Plateau (described in Chap. 2 of this

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book). That same gradient creates a multitude of margins for the distributions of species across northern to southern latitudes, and between eastern and midwestern biomes. The central spine of the Appalachians divides the river basins of the Delaware, Susquehanna, Ohio, and north-flowing rivers to the Great Lakes. The isolation of these rivers has contributed to their diverse faunas and the high endemism of some groups (e.g., salamanders of the Appalachians, freshwater mussels, crayfish, and fishes (Abell et al. 2000)).

The ridges, river valleys, and rivers serve as critically important migratory corridors for raptors (e.g., Hawk Mountain Sanctuary on the Kittitenny), neotropical songbirds (e.g., Louisiana waterthrush), and anadromous fishes (e.g., American shad), respectively. Throughout the freshwater wetlands of the MAR, one encounters rare, threatened, and endangered species of concern, often because this region represents the edge of their range, but also because of pressures from settlement patterns during the past 400 years. Wetland and riparian species occur on lists for species of concern in higher proportions than their overall species richness would predict. In some states of the MAR, wetland- and riparian-dependent species comprise at least 40–50% of listed species of concern. A list of fauna and flora of concern, arranged by state, can be found at www.riparia.psu.edu/MARbook.

There are a variety of excellent sources of information pertaining to wetland wildlife relevant to the MAR; consequently we will not attempt to replicate them here. For example, in 1989, the Pennsylvania Academy of Science published a book, Wetlands Ecology and Conservation: Emphasis in Pennsylvania, which contained chapters describing major biological taxa, including many wetland-dependent wildlife groups such as mammals (Kirkland and Serfass 1989), songbirds (Brauning 1989), waterbirds (Hartman 1989), raptors (Rymon 1989), reptiles and amphibians (McCoy 1989), and fishes (Boltz and Stauffer 1989). Much of that information remains relevant today for much of the MAR despite the focus on Pennsylvania. In subsequent chapters related to vertebrate and invertebrate wildlife, we will focus on a number of studies that highlight MAR's wildlife biodiversity with either an ecoregion or wetland-type focus. Most of these case studies were conducted by the faculty, staff, and students of Riparia from the 1980s through the present. There are also a variety of species lists, descriptive accounts, and range maps that are readily available in a variety of publications and websites applicable for the region (e.g., state Heritage Programs; GAP Analysis Program models of species distributions by state), which were designed to identify concentrations or "hot spots" of species diversity (e.g., Myers et al. 2000; http://gapanalysis.usgs.gov/); amphibians and reptiles of Pennsylvania and the Northeast (Hulse et al. 2001); wetland field guides (e.g., Tiner 2005), and taxonomic guides (e.g., freshwater invertebrates, (Voshell 2002), etc.).

Wetland mammals typically found in the region are described in DeGraff and Rudis (1986), Kirkland and Serfass (1989), and Serfass and Brooks (1998), as well as a variety of field guides. Consequently, we feature our discussion here on two mammalian species closely associated with wetlands and with characteristics that make them excellent wetland "flagship species," river otter and beaver. Subsequent chapters report on recent research on birds (Chap. 8), amphibians and reptiles (Chap. 9), and freshwater macroinvertebrates (Chap. 10).

7.2 Diversity of Wetland-Riparian Wildlife in the Mid-Atlantic Region

Headwater streams and their associated wetlands and floodplains are important for selected fish habitat components (e.g., salmonids and clupeids) but, in general, do not support the high biomass or species richness present in larger rivers. The distribution and habitat of fish species was documented by Cooper (1983) and others for the region, and there have been fish inventories in individual National Park units (e.g., Leonard and Orth 1986; Ross et al. 2003). The following websites provide links to the specific Inventory and Monitoring programs of the park regions within the MAR:

http://science.nature.nps.gov/im/units/ermn/
http://science.nature.nps.gov/im/units/midn/index.cfm
http://science.nature.nps.gov/im/units/ncrn/index.cfm
http://science.nature.nps.gov/im/units/ncbn/index.aspx

In high gradient headwater streams, brook trout, various minnows (Cyprinidae), and sculpins (Cottidae) are common. Boltz and Stauffer (1989) highlighted the fishes that are dependent in some manner on wetlands and their connectivity with streams. Although the richness and abundance of fishes in tributary watersheds can be a useful indicator of condition, fish penetration into the upper reaches of these ecosystems is limited (Church 2002). In coastal plain streams, however, headwater streams and associated floodplains are important for the spawning and rearing of shad and river herring (Walsh et al. 2005). In places where fish are not present in abundance, amphibians, particularly streamside salamanders, and riparian birds serve as top trophic-level predators, and thus, can serve as an alternate vertebrate indicators of condition (Brooks et al. 1998; O'Connell et al. 2003; Rocco et al. 2004).

The importance of the wetland and riparian components of watersheds as habitat for wildlife communities is reasonably well documented in the MAR. Just as rivers and lakes provide habitat for fishes, the provision of wildlife habitat is an oft-cited function of the adjacent wetlands and riparian areas. Profiles for various taxa are summarized in DeGraff and Rudis (1986), Majumdar et al. (1989), Brooks et al. (1993), Serfass and Brooks (1998), Tiner (2005), and in Chaps. 8, 9, and 10 of this book. Obligate and facultative fauna using these stream, wetland, or riparian habitats can include seasonal (e.g., aquatic insects, winter migrant birds, summer foraging bats), resident (e.g., freshwater mussels, cyprinid minnows, salmonids, streamside salamanders, beaver), wide-ranging (e.g., American mink, river otter, herons), or breeding migrant (e.g., belted kingfisher, Louisiana waterthrush, Acadian flycatcher) species (see wetland-dependency guild for wildlife in Brooks and Croonquist 1990).

Breeding waterfowl and waterbirds are found throughout the MAR, but in relatively low densities compared to other regions of North America (e.g., Prairie Potholes, Great Lakes, Everglades). Coastal bays and estuarine wetlands attract more species and greater numbers during winter months, particularly when inland rivers and wetlands are ice covered. The larger rivers and their floodplain corridors, such as the Susquehanna and Delaware, are used heavily during spring and fall migrations (Hartman 1989).

Representative species of ducks using freshwater wetlands include mallard, black duck, wood duck, and the mergansers. Beaver ponds in various successional stages are used frequently, providing isolation from other breeding pairs, abundant macroinvertebrates and seeds as food, and cover for fledglings (Prosser 1998). Both resident and migratory populations of Canada geese are common. Heronies of great blue heron, common egret, and night herons also occur, particularly in the vicinity of large rivers or large wetland complexes, but are not abundant. Inventories of colonial-nesting waterbirds, particularly those found inland from coastal beaches and islands, have not been conducted regularly in the region. Monitoring protocols for colonial-nesting waterbirds have been implemented (e.g., http: www.manomet.org), but to date, data must be gleaned from reports compiled by individual states. Portions of the MAR provide important migratory stopovers and wintering habitats for a broader range of species than those nesting or denning here.

Fisheries, as an important commercial and recreational resource, are commonly monitored in streams and rivers by resource agencies, yet seldom have appropriate levels of financial resources been available to adequately survey the correspondingly diverse wildlife community. Several states in the region have completed or updated Breeding Bird Atlases (BBA) (e.g., for Pennsylvania's 1st BBA see Brauning (1992) (available online at http://www.carnegiemnh.org/powdermill/atlas/1pbba-book.html); 2nd BBA for Pennsylvania is in final stages, found at http://bird.atlasing.org/Atlas/PA/; BBA for West Virginia see Buckelew and Hall 1994; and others). The Breeding Bird Survey (BBS) organized by the U.S. Fish and Wildlife Service (Sauer et al. 2008) provides long-term trends on bird populations across the country at a coarse scale. Resource agencies are responsible for tracking and managing endangered, threatened, and rare species of concern, with the U.S. Fish and Wildlife Service and state natural resource conservation agencies given the authority under the Endangered Species Act and parallel state statutes. Numerous wetland-dependent and aquatic species are on these lists in the MAR because human activities have destroyed or degraded habitats for centuries.

A commonly used surrogate for surveying the distribution of populations is to assess *potential* wildlife and fisheries habitat with Habitat Suitability Index (HSI) models. HSI models have been used as a means to estimate the level of wetland function as wildlife habitat based on consistent use of ten common species for all of Riparia's reference wetlands (Brooks and Prosser 1995; Brooks 2004; see use of HSIs for comparing wetland reference and mitigation sites in Chap. 12 of this book).

As emphasized throughout this chapter, aquatic landscapes are a collection of wetland, riparian, and stream habitats connected by the movement of water, carbon, and nutrients. However, species also move within and among various habitats and, consequently, the biological integrity of a given area also depends on factors that affect species movement. Within the riverine network itself, fish and aquatic macroinvertebrates use different habitats at different times of the day, year, and phases of their life cycle, a topic explored further in Chap. 14 of this book.

7.3 Wetland "Flagship Species": River Otter and Beaver

At one end of the taxonomic spectrum of wildlife species are two mammalian species found in the MAR, river otter (*Lontra canadensis*) and beaver (*Castor canadensis*), that attest to the importance of maintaining aquatic connectivity, as both are obligate, wetland–riparian species.

7.3.1 River otter

Within aquatic environments the river otter is best described as a habitat generalist, existing over a large geographic range in a wide range of ecological conditions from the tropics to the nearctic. River otters in North America, inhabit rivers, streams and associated backwaters, lakes, bogs, beaver ponds, emergent marshes, forested swamps, and coastal habitats (e.g., Toweill and Tabor 1982; Lariviere and Walton 1998). Den and resting sites usually occur in riparian areas. Beaver dens (lodges and bank dens), undercut banks with extensive tree root systems, rock formations, backwater coves, floodplain wetlands, and log jams with accumulated brush piles are known to serve as denning and foraging areas (Swimley et al. 1998; Stevens et al. 2011). Yet, this cosmopolitan use of habitats can be curtailed by environmental disturbances of human origin. Water pollution such as severe acidic mine drainage has indirectly caused the elimination of otter populations in drainages through the destruction of the aquatic food web. The effects of other environmental disturbances on otter populations, including levels of bioaccumulating pollutants in otter prey, alteration of riparian habitats, and construction of dams and levees, are poorly understood. Like many mustelids, otters are wide-ranging, capable of traveling many kilometers in a single day and occupying a variety of aquatic habitats (Spinola 2003; Spinola et al. 2008). Such behavior allows them to avoid unsuitable habitat or other disturbances to some extent, but also makes accurate sampling of populations challenging at best.

Serfass, his students, and colleagues, have explored the efficacy of numerous protocols with implications to estimate the occurrence and abundance of otters in the MAR (see Serfass et al. (2003) for a review)—although most of the research was conducted in Pennsylvania and western Maryland. For example, use of latrines (areas where otters deposit scats, urine, and other scent along the shoreline) is an effective way to monitor the presence of otters because latrines are relatively easy to detect by surveying riparian areas (e.g., Swimley et al. 1998; Swimley et al. 1999; Stevens et al. 2011). However, the efficiency of these surveys for latrines can be enhanced considerably by focusing on riparian areas with habitat characteristics shown by and Swimley et al. (1998) and Stevens et al. (2011) to be closely associated with latrines. Seasonality also plays an important role in the detection of latrines. Otters tend to mark more frequently at latrines during the spring and fall, than during summer (Carpenter 2001; Serfass et al. 2003; Stevens 2005). Selecting areas with certain riparian characteristics and conducting surveys during spring or fall (note: latrines may not be detectable during winter because of snow cover in some areas of the MAR) will therefore enhance the likelihood of detecting evidence of otter latrines.

The development of these techniques has allowed the monitoring of otter presence within Pennsylvania and adjacent states. As in other parts of the river otters' range in North America, intensive harvest and degradation of aquatic and riparian habitats were responsible for extirpation of populations in Pennsylvania, and other Mid-Atlantic states (Rhoads 1903). To help expand the otters' range from a stronghold in the glaciated regions of northeastern Pennsylvania, river otters were reintroduced to riverine habitats in northcentral Pennsylvania from 1982 to 1986 by the Pennsylvania River Otter Reintroduction Project (PRORP) (Serfass et al. 1986; Serfass et al. 2003). Over time, PRORP released 153 river otters in seven discrete drainage systems throughout the western and central portions of Pennsylvania and contributed to successful restoration of extirpated populations (Serfass et al. 1993; Serfass et al. 1999; Serfass et al. 2003; Hubbard and Serfass 2005). Additional otters were released in western New York (Spinola 2003; Spinola et al. 2008).

Dispersal of individuals from the Chesapeake Bay in Maryland appear to have resulted in repopulation of portions of the lower Juniata and Susquehanna rivers. Dispersal of river otters reintroduced in Ohio and New York are suspected to have contributed to the occurrence of individuals in northwestern Pennsylvania (Serfass et al. 1999; Serfass et al. 2003). Similarly, river otters reintroduced in portions of western Maryland and West Virginia may be contributing to the expansion of populations in southwestern Pennsylvania. Overall, river otters will benefit from any conservation program designed to protect or restore aquatic habitats with regard to both water quantity and water quality. They will benefit by encouraging the implementation or enhancement of programs that: (1) reduce emissions causing acid rain; (2) implement streambank fencing projects to protect riparian and aquatic habitats in areas where livestock are grazed; (3) enhance existing regulations designed to protect or limit the loss of wetlands; (4) further regulate mining activities that cause acid mine drainage; (5) implement strategies to mitigate the effects of existing acid mine drainage; and (6) enhance policies and enforcement activities to control all forms of point and nonpoint sources of water pollution. Because of these linkages, the river otter does, in fact, serve as a flagship species, calling attention to activities that degrade aquatic habitats and water quality.

Otters and beavers are sympatric throughout most of North America. Otters benefit from modifications of aquatic habitats resulting from dam and den building activities of beavers. Beaver ponds can provide an abundance of various prey items, den sites, stable water levels, and escape cover for otters. The association between beavers and otters has been demonstrated throughout the otters' range in a variety of habitat types. On Mount Desert Island, along the coast of Maine, otters selected watersheds with a high proportion of streams with beaver impounded (Dubuc et al. 1990). Latrine sites are used as an indication of the presence of otters and to infer habitat use and preference. Based on an evaluation of otter latrine sites in central Massachusetts, Newman and Griffin (1994) reported that otters used more beaver created sites than artificial impoundments during summer. In northcentral and northeastern Pennsylvania, otter latrines were strongly associated (95%) with active and recently inactive beaver habitats (Swimley et al. 1998; Swimley et al. 1999). The presence of beavers may be critical to many otter populations in regions that retain substantial ice cover for long periods.

7.3.2 Beaver

Beaver, the master wetland builder, serves an important role as a flagship species as it continuously creates new wetland habitats throughout the region. As furbearers, beavers and otters are managed by the wildlife agencies of individual states. Their population densities will depend, in part, on how trapping seasons and anthropogenic disturbances affect their fecundity. With the decline of both fur prices and interest in fur trapping, managing expanding populations of beaver has become challenging to state and local wildlife biologists and managers. When viewed from the perspective of habitat creation, however, a large number of benefits are accrued.

In the MAR, the dynamic nature of beaver ponds creates a mosaic of multi-successional wetland patches within a forested matrix, providing diverse habitat for many wildlife species. The mix of successional stage beaver ponds naturally found in the landscape provides diverse habitat for avian species, including waterfowl, waterbirds, songbirds, and even raptors. Stages of beaver ponds are usually recognized as those found in a classic successional sequence (Brown and Parsons 1979): (1) new active (Stage 1, 1–6 year) with areas of open water interspersed with dying or dead trees that have been recently colonized; (2) old active (Stage 2, 7–12 year) possess decaying timber, but few plant food sources requiring beaver to travel farther for food, and possibly establish new ponds; and (3) abandoned (Stage 3, >12 year), with declining water levels and abundant emergent vegetation where dams are not longer maintained. Otters would likely use abandoned beaver lodges as denning or resting sites.

Following abandonment, the beaver habitats are assumed to gradually reforest until the available woody biomass is sufficient to provide both food and building materials, again. A study by Prosser (1998) used historic analysis of aerial photographs to show that beaver can short-circuit this successional sequence, and recolonize an emergent- and/or shrub-dominated wetland before trees become reestablished. Thus, one can recognize two new stages of colonization, one dominated by forest, and the other composed of open water and non-forest patches. Each of these types offers an array of niches of a variety of taxa. As sunlight and nutrients increase in new, forested ponds, there is often a flush of aquatic macroinvertebrate populations providing a prey base for young waterfowl broods and aerial insectivores such as swallow, swifts, flycatchers, and bats. During the intermediate stages, dead and dying trees provide cavities for nesting and denning, and foraging sites for woodpeckers and other insectivores. Once the woody component is replaced by herbaceous, emergent, and submergent plants, a broader community of herptiles is likely to use these sites for breeding, foraging, and basking. During these latter stages shortly following abandonment, both the aquatic portions and the more terrestrial herbaceous vegetation support different insect assemblages, grassland birds, secretive waterbirds such as rails and bitterns, and a diverse community of both obligate and facultative species (Brooks et al. 1993). Thus, a mosaic of all stages of beaver pond succession within a landscape appears to be most desirable for supporting a diverse wildlife community.

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Chapter 8
Wetland-Riparian Birds of the Mid-Atlantic Region

Timothy J. O'Connell, Robert P. Brooks, Diann J. Prosser, Mary T. Gaudette, Joseph P. Gyekis, Kimberly C. Farrell, and Mary Jo Casalena

Abstract Wetland-riparian birds are conspicuous in the Mid-Atlantic Region (MAR), but sometimes use aquatic habitats beyond what is typical for waterfowl, waterbirds, and shorebirds. Over the past two decades, Riparia conducted studies to determine the importance of wetlands and riparian corridors as habitats for birds covering the range from obligate to facultative users. We sampled wetlands used by wood ducks and other waterfowl species common to the Appalachians. We surveyed

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lakes and their associated fringing wetlands and riparian edges in the glaciated Pocono region. Songbirds were sampled along human disturbance gradients from wetlands into adjacent uplands. In addition, our work developed and tested an innovative way to consider biodiversity through the component life history traits of species in communities—a guild-based approach. The quantification of differences in the life history composition of different communities led to the development of a class of ecological indicators called the Bird Community Index (BCI). In this chapter, we trace the development and application of the BCI and its derivatives, including a multi-metric Regional Index of Biological Integrity for forest riparian ecosystems (RIBI). The chapter progresses through studies of birds using wetland habitats, beginning with obligate species and progressing through facultative users. We conclude with a summary of projects and monitoring protocols that use the BCI for ecological assessment, and a description of emerging issues that make such assessments a conservation imperative for the MAR.

8.1 Introduction

Birds are among the most conspicuous groups of wildlife using wetlands in the Mid-Atlantic Region (MAR). Whether the clapper rail calling from an estuarine *Spartina* marsh, the great blue heron fishing from a secluded riverbank, or the pair of Canada geese paddling with their young along the margins of a small lake, the obligate wetland birds of the MAR provide iconic indications that a given wetland is providing habitat for native wildlife. Americans should take great pride in those images, as it has been our demand for and commitment to legislation protecting and restoring wetlands, and the species that inhabit them, that has made such encounters possible today (Fig. 8.1).

These examples, however, offer just a glimpse of the importance of wetlands for many birds, most of which would be considered only facultative users of wetlands. Throughout the expansive riverine upper perennial headwaters of the Chesapeake Bay, birds not usually thought of as wetland-associated rely on the resources provided by innumerable riparian wetlands beneath forested canopies. Headwater wetlands provide foraging opportunities, nesting substrate and protective cover, and thermal cover benefits that corresponding uplands often do not. As a consequence, these wetlands help support species found in no other habitats in the MAR or support the highest densities of otherwise widespread species. Regardless of the mechanism, headwater wetlands in the MAR are vital to regional biodiversity.

In Riparia, we have conducted numerous studies of the importance of wetlands as habitats for birds covering the range from obligate to facultative users. In addition to drawing connections from specific wetland features to life history needs of birds, our work has introduced a novel way to consider biodiversity through the component life history traits of species in communities. The quantification of differences in the life history composition of different communities led to the development of a class of ecological indicators that we call the Bird Community Index (BCI).

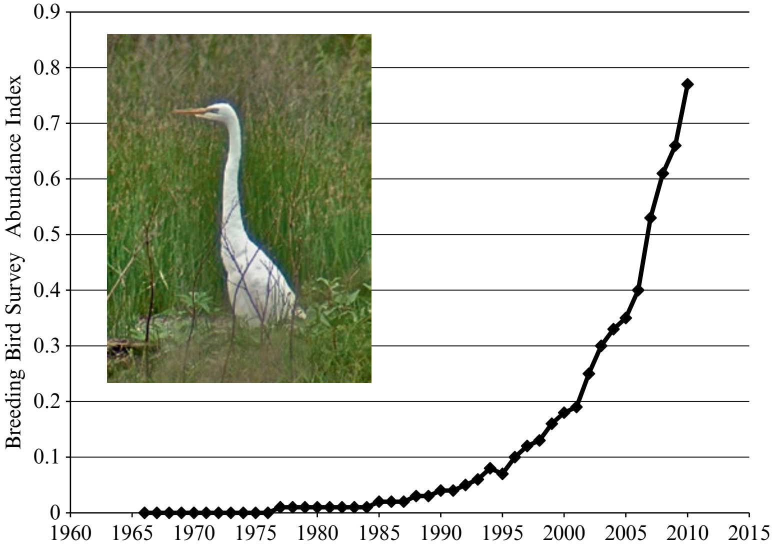

Fig. 8.1 Thanks to legislation directed toward the protection of wetlands and wildlife, obligate wetland birds such as the great egret (*Ardea alba*) have steadily increased from historical lows in the Mid-Atlantic Region. (Data from Sauer et al. 2011; photograph by T. O'Connell)

As development and application of the BCI progressed, we developed a deeper understanding of how wetlands provide the resources that support avian diversity throughout the entire region.

In this chapter, we summarize studies of wetland birds in their wetland habitats, beginning with obligate species and progressing through facultative users. We also trace the development and application of the BCI and its derivatives. The chapter concludes with a summary of projects and monitoring protocols that use the BCI for ecological assessment, and a description of emerging issues that make such assessments a conservation imperative for the MAR.

8.2 Wetland-Associated Birds in the Mid-Atlantic Region

8.2.1 Breeding Habitat Use of Wood Ducks in the Pocono Region

Waterfowl are obligate users of wetland habitats, and species have evolved to exploit resources under different conditions. For example, diving ducks inhabit coastal areas and deeper lakes where their ability to submerge and seek prey well below the surface is a distinct advantage. Dabbling ducks frequent smaller ponds and lake margins where they can obtain food on the surface of the water, just below the surface, or in adjacent terrestrial environments. The wood duck (*Aix sponsa*) is a dabbling duck that also presents a challenge for land managers: it is an obligate cavity nester and an obligate wetland bird. Wood ducks require wetland resources in proximity to trees containing cavities large enough to accommodate a mother and her brood.

In part due to the loss of wetlands and large cavity trees in the United States during the twentieth century, there have been widespread efforts to establish nest boxes for wood ducks on ponds, lakes, and in emergent wetlands. Although nest boxes are a common management practice used to enhance wood duck populations, they are frequently established without consideration for brood habitat requirements.

Wood duck broods use a variety of wetland habitats, including small beaver ponds, abandoned river oxbows, river channels and floodplains, lake shorelines, live forest, swamp shrubs, emergent vegetation, and sedge meadows (Hepp and Bellrose 1995). To learn more about how wood ducks made use of habitats for developing broods in areas near nest boxes, Claypoole (Farrell) (1997) conducted a radio-telemetry study of wood duck hens and their broods in northeastern Pennsylvania in 1992.

We studied wood duck broods on five wetlands located within Pike County. Decker Pond, Decker Outlet, and Decker Creek form a wetland complex located on State Game Land 183 in western Pike County. The 56-ha Decker Pond was dominated by wetland forest and scrub-shrub habitats to the north and west, and open water to the south and east. Prevalent plant species included: buttonbush (*Cephalanthus occidentalis*), red maple (*Acer rubrum*), and eastern hemlock (*Tsuga canadensis*). By late spring, watershield (*Brasenia schreberi*), interspersed with some pond lily (*Nuphar* spp.) and water lily (*Nymphaea odorata*), covered most open water. Decker Pond drains southeast, through a man-made berm, into Decker Outlet, a 2.5-ha wetland containing equal proportions of open water, scrub-shrub, and forested habitats, and a smaller amount of emergent habitat. Plant species included bluejoint (*Calamagrostis* spp.), sedge (*Carex* spp.), box elder (*Acer negundo*), and red maple. Decker Outlet flows north into Decker Creek, a narrow waterway extending 5 km northward before draining into the Lackawaxen River. Decker Creek flows through a densely vegetated wet meadow, dominated by sedge and bluejoint, and then enters a mix of wetland forest and emergent habitats, and upland.

Shohola, a 432-ha lake created by impounding Shohola Creek, is located on State Game Land 180 in north central Pike County. Inflows to Shohola include Mile Brook from the west and Rattlesnake Creek from the east. Shohola is impounded at its north end and drains south and west via Shohola Creek. Wetland habitats on Shohola are open water, emergent, shrub-scrub, and forest. Emergent plant species included woolgrass (*Scirpus cyperinus*), bluejoint, and smartweed (*Polygonum* spp.). Shrubs were dominated by highbush blueberry (*Vaccinium corymbosum*), and forest included red maple and swamp oak (*Quercus bicolor*).

Peck's Pond is located in the Delaware State Forest, south central Pike County. Inflows to the 332-ha pond are Maple Creek from the north and Tarkill Creek from the west. Peck's Pond drains south via Bush Kill. Wetland habitat on Peck's Pond consisted of open water, aquatic bed, emergent, scrub-shrub, and forest. Plant species included eastern hemlock, red maple, highbush blueberry, sheep laurel (*Kalmia angustifolia*), leatherleaf (*Chamaedaphne calyculata*), sedge, pickerel weed (*Pontederia cordata*), pond lily, water lily, and watershield.

We trapped wood duck hens during incubation and equipped them with a US Fish and Wildlife Service leg band, a urethane plastic nasal saddle (Greenwood 1977), and a radio-transmitter. We attached plastic nasal saddles through the nares with a nylon pin. Nasal saddles were color coded red, green, white, or black and were labeled alphanumerically, allowing for the identification of individual females during field observations. Radio-transmitters, developed by Hi-Tech Services, Camillus, NY, weighed 14 g, had an average life expectancy of 5 month, and were back-mounted with an adjustable harness (Dwyer 1972).

We began tracking hens one day after transmitter attachment, but did not include hen movements and habitat use in data analysis until 5 days after transmitter mounting. The 5-day delay allowed hens to adjust to the transmitters. Hens were located primarily by homing, supplemented by triangulation (White and Garrott 1990). Each day was split into four equal time blocks, beginning one-half hour before sunrise and ending one-half hour after sunset. Hens were located during a different time block each day, with locations rotated through all time blocks every 4 days. Time blocks were ordered so that locations were made as close to 24 h apart as possible, helping ensure location independence.

Hens were located to within 1 ha and estimated hen location to be at the center of the hectare. She recorded all locations using a Universal Transverse Mercator (UTM) grid imposed on 1:24,000 topographic maps. We identified dominant vegetation within each hectare of the hen's location to class, subclass, and plant genera (Cowardin et al. 1979). If more than one habitat type existed within the hectare where the hen was located, we determined which habitat the hen and her brood were using. We assumed hens were with their broods at the time of location. We observed broods weekly to evaluate survival and confirm that broods remained with radio-tagged hens. We tracked hens with broods for 5 weeks, at which time hen-brood bonds begin to break down (Beard 1964). We defined successful broods as those known to have at least one duckling survive to 3 weeks.

We used color infrared aerial photographs (1:24,000) to delineate the area of different habitats present on each wetland. We identified habitat to class (e.g., forest, scrub-shrub) based on criteria provided by Cowardin et al. (1979). We mapped habitat class polygons on overlays of aerial photographs and digitized these into an $\mathrm{ARCINFO}_{\mathrm{TM}}$ database to determine the area (ha) of each habitat type. We ground-checked habitat information obtained from aerial photographs during radio-telemetry locations.

We tracked eight hens with broods in the Decker wetland complex from 24 May–11 July 1992. All hens hatching young on Decker Creek moved their broods to Decker Pond within 24 h of leaving the nest box. Assuming the hens traveled by water, broods moved at least 2.0 km. Hens stayed on Decker Pond for the remainder of the brooding period, but returned to Decker Creek later in the season without their broods.

Hens hatching and raising broods on Decker Pond ($n = 4$) were located an average 0.3 ± 0.2 km from their nest boxes. Maximum brood distance from the nest box was 0.8 km. One other hen hatched a brood on Decker Pond, but in 9 days moved her brood an estimated 9.5 km to a scrub-shrub (*Spiraea latifolia* and *S. tomentosa*) and

Table 8.1 Habitat used by wood duck hens and broods at three study sites in northeastern Pennsylvania, 1992

Study site									
	Decker pond			Shohola			Pecks pond		
Habitat class	Ha	%	% Brood locations	Ha	%	% Brood locations	Ha	%	% Brood locations
Open water	2.3	4.1	2.0	174.0	40.3	0.0	96.0	29.0	0.0
Aquatic bed	12.7	22.7	34.5	n/a	n/a	n/a	61.5	18.5	17.4
Emergent	n/a	n/a	n/a	121.9	28.2	55.1	n/a	n/a	
Scrub-shrub	23.5	42.0	62.0	40.2	9.3	14.2	55.9	16.9	82.6
Forest	17.4	31.1	1.5	95.7	22.2	30.7	118.2	35.6	0.0

emergent (*Agrostis* spp., *Leersia oryzoides*, and *Carex* spp.) wetland. She remained on this wetland for the remainder of the brood-rearing period. Leg band information obtained from this hen indicated she was captured on this same wetland as a hatch-year bird in 1991. Based on 134 radio-locations, broods raised on Decker Pond (including those hatched on Decker Creek) used aquatic bed and scrub-shrub habitats most frequently, and in greater proportion than their presence on the wetland.

At Shohola, seven hens and their broods were radio-tracked from 25 May–21 July 1992. Average distance hens were located from nest boxes during brood-rearing was 0.7 ± 0.8 km. Maximum brood distance from the nest box was 3.5 km. Five of seven hens raised broods at the southwestern end of Shohola, including the four hens that nested in this area. One hen moved off of Shohola 10 days after leaving her nest box, and was located on a wetland approximately 1.0 km east of Shohola, connected to Shohola by Rattlesnake Creek. We were unable to visually locate this hen after she left Shohola and could not determine if she moved alone or with a brood. Based on 129 radio-locations, broods raised on Shohola used emergent and forested habitat most frequently, and in greater proportion than their presence on the wetland while open water was avoided.

Two radio-tagged hens hatched broods from Peck's Pond and were tracked from June 15 to July 28 1992. Hens were located an average 0.3 ± 0.1 km from their nest boxes. Maximum distance a brood was located from the nest box was 0.6 km. Based on 46 radio-locations, hens and broods on Peck's Pond used wetland scrub-shrub most frequently, and in greater proportion than its availability on the river. Emergent habitat was also available and used, but was interspersed with other habitat types, and did not dominate any hectare. A minimum of five broods were observed with unmarked hens. We assumed broods hatched from natural cavities because all hens successfully hatching nests from nest boxes on Peck's Pond were radio-tagged. Removing the area of forest and open water (214.2 ha), habitats avoided by wood duck broods, brood density on Peck's Pond was an estimated 0.06 broods/ha of suitable habitat. Habitat used by wood duck broods at all three study areas is summarized in Table 8.1.

Food and cover were critical for wood duck brood survival (Haramis 1990), and wetland habitats that provided these factors should have been used most often by wood duck broods. Flooded aquatic bed, emergent, and scrub-shrub habitats were all used when available to wood duck broods. Many invertebrates were observed clinging to surfaces of aquatic bed and emergent vegetation, providing an important forage to young ducklings (McGilvrey 1969; Smith and Flake 1985; Haramis 1990). Shrubs, such as buttonbush, also harbored insects for young ducklings and provided plant foods for older ducklings. Emergent and shrub habitats were used by broods for cover. Shrub interspersion with emergent and aquatic bed plants, such as occurred on Peck's Pond, was a valuable mixture of habitats for wood ducks (McGilvrey 1969; Hepp and Bellrose 1995).

Despite most habitat on Decker Creek being emergent, wood duck hens ($n=3$) failed to remain on the Creek to raise their broods. Hens returned to Decker Creek after bonds had broken with broods, indicating the Creek was suitable habitat for hens. Water access to emergent vegetation in Decker Creek was limited to small channels that ran intermittently through the seasonally saturated sedge hummocks. Terrestrial predators, such as raccoons, mink (*Mustela vison*), and red fox (*Vulpes vulpes*), could access most of the water channels, and little overhead cover was available in channels to protect broods from aerial predators such as hawks and owls. Potentially hens moved broods to Decker Pond to provide better protection from predators. Flooded shrubs on Decker Pond were inaccessible to land predators and provided cover to protect broods from aerial predators.

At the southwestern end of Shohola, broods made frequent use of the temporarily flooded forest. This habitat was inaccessible to land predators, and because it was interspersed with shrubs, also provided aerial cover. Broods were observed gleaning insects from the leaves of the shrub understory. Forest habitat probably was used proportionately less than its availability because it was only seasonally flooded. Forest habitats on Decker Pond and Peck's Pond were used infrequently by wood ducks. Although Gammonley (1990) determined live flooded trees were important to young wood duck broods, much of the forested habitat on Decker Pond and Peck's Pond was seasonally saturated or shallowly flooded. Broods reared in this habitat did not have protection from land predators. Although a lack of food may have been another factor deterring broods from this habitat, most studies have indicated seasonally flooded woodlands were a valuable source of invertebrates (Drobney 1990; Gammonley 1990).

Most hens nesting and raising broods on Decker Pond, Shohola, and Peck's Pond remained within 1.4 km of the nest box, a shorter distance than reported in many other wood duck brood movement studies (Hardister et al. 1962; Hepp and Hair 1977; Smith and Flake 1985; Gammonley 1990). The relatively close proximity of hens to nest sites during brood-rearing indicated nest boxes were set up in the vicinity of suitable brood habitat.

Survival of broods appeared to be negatively affected by long-distance travel (>2 km) from the nest site to rearing wetland. Managers can correct the problem of travel mortality by establishing nest boxes within or near high quality brood habitat. Broods require food, cover, and protection from predators to survive. Establishing nest boxes in habitats that provide these requirements, will allow hens to remain in the nest box vicinity to raise their broods. Flooded forest with an understory of shrubs, or shrubs interspersed with emergent and aquatic bed plants are beneficial habitats for broods. Each habitat type provides a different function or forage for broods; a combination of these habitat types will help ensure broods are able to remain in the nest box area throughout the brood-rearing season.

Temporarily flooded woods were used frequently by wood duck broods. If managers are able to control water levels on brood wetlands, spring flooding of live habitats, such as forest, shrub, or persistent emergent vegetation, will allow broods to access new, unexploited food sources. Water levels can be drawn down as non-persistent vegetation (emergents and aquatic bed) becomes available to broods later in the brood-rearing period.

Finally, homing instincts of wood ducks affected brood travel. An awareness of which wetlands are used consistently by wood ducks will help managers protect these sites, and enhance them by providing nest boxes or improving brood habitat.

8.2.2 Waterfowl Use of a Successional Gradient of Beaver Ponds

Wetland types vary in their abundance, composition, and spatial distribution across physiographic regions. In the MAR, extreme northeastern and northwestern Pennsylvania were covered by the continental ice sheet during the most recent (Wisconsin) glaciation (Oplinger and Halma 1994). As a result of glacial advance and retreat, these areas in Pennsylvania support an abundance of bogs, kettle lakes, and other wetlands providing open water areas. Wetland densities are highest in these areas, with nearly half of the state's wetlands contained in only 17% of the total land area (Tiner 1989). Elsewhere in the MAR, availability of open water is often dependent on hydrologic modification through the action of American beaver (*Castor canadensis*). Including abandoned ponds, approximately 6,500 beaver ponds existed in the state in 1996, totaling 195 km^2, or 8% of Pennsylvania's wetland area (Prosser 1998). Through field research conducted by Riparia in 1995 and 1996, we examined the role of beaver ponds in various states of natural succession in providing habitat for breeding waterfowl in Pennsylvania.

We studied 40 beaver ponds: 21 in the non-glaciated, central region of Pennsylvania in 1995 and 19 in the glaciated, northeastern, Pocono Region in 1996. Each beaver pond was initially classified into one of three relative successional stages using beaver activity as indicators (Brown and Parsons 1979). Field signatures of the three main successional stages were as follows:

- New Active (Stage 1)—possess signs of current beaver activity; well maintained dam and lodge, recent beaver cuttings, scent markings, winter food cache; timber within beaver pond boundary is either living or recently dead; approximate age ranges from 1 to 6 year.
- Old Active (Stage 2)—possess indicators of current beaver activity; flooded, deceased timber show signs of aging decay; few major branches remain; little remaining food sources exist for the beaver; approximate age ranges from 7 to 12 year.

- Abandoned (Stage 3)—no indicators of current beaver activity; lodge is often collapsed or grown over, dam often is covered with vegetation, water levels are not maintained, few snags remain (often merely stumps); approximate age is >12 year.

Additionally, we recognized that some beaver ponds could be newly established in emergent wetlands that lacked typical indications of fresh beaver activity on trees. We drew a distinction between “new-forested” and “new-open” beaver ponds (stage 1a and 1b, respectively). To determine the vegetative structure at the time of beaver colonization, we examined four historic aerial photographs for each site taken between 1944 and 1968. Aerial photographs (1:25,000) and satellite images (1:80,000) were obtained from the US Geological Survey. Each site was located on the aerial photograph and interpreted into one of five vegetative cover classes modified from the Cowardin et al. (1979) classification system for palustrine wetlands: forested, shrub, emergent, open water, or existing beaver ponds.

Nine waterfowl surveys were conducted weekly during the breeding and brood-rearing season (May–July). A combination of stationary and transect counts were used to optimize detection of waterfowl broods (Rumble and Flake 1982). Surveys were conducted during morning and evening hours (within 4 h of dawn and dusk), rotating each visit. Direct counts, including both auditory and visual observations, were conducted for 80 min from an elevated vantage point (4–6 m) using portable tree stands. Data taken for each observed individual included: species, sex (if discernible), and time recorded. The bird’s general location within the pond was mapped. Specific information for waterfowl broods included species, presence of hen, activity, number of individuals in the brood, and age class of the brood.

Vegetative structure was measured using low-altitude aerial photographs taken within the avian survey period for each study site (scale < 1:4,000; Prosser 1998). Patches were digitized using Digitize$_{\text{TM}}$ and MacGridzo$_{\text{TM}}$ Geographic Information Systems (Rockware Scientific Software 1991). Percent cover of summed habitat patches for each habitat variable was analyzed by beaver pond stage using nonparametric Kruskal-Wallis tests because variables could not be normalized using transformations (Noether 1991). Percent cover was used instead of area measures to standardize against differing pond sizes. Tree and snag density were measured separately, using the point sampling method, a plotless technique (Grosenbaugh 1952).

Six species of waterfowl were observed during this study: Canada goose (*Branta canadensis*), wood duck, green-winged teal (*Anas crecca*), American black duck (*A. rubripes*), mallard (*A. platyrhynchos*), and hooded merganser (*Lophodytes cucullatus*). Wood ducks and mallards were detected on the greatest number of sites, followed by hooded merganser, Canada goose, American black duck, and green-winged teal. All six species were observed in the northeastern region; however, American black duck and green-winged teal were not detected on central Pennsylvania sites.

Patterns in presence of waterfowl species were observed among the successional stages. In central Pennsylvania, Canada geese were observed on old-active and abandoned beaver ponds only; wood ducks and mallards were observed on all stages, and

Table 8.2 Number of waterfowl broods and successional stage locations where broods were detected for 6 waterfowl species on 40 beaver pond study sites located in central, non-glaciated Pennsylvania ($n=21$) and northeastern, glaciated Pennsylvania ($n=19$), 1995 and 1996, respectively

Species	No. Broods central region (C)	Stages with brood presence (C)	No. Broods Northeast region (NE)	Stages with brood presence (NE)
Canada goose	0	–	3	2
Wood duck	2	1a, 3	2	1a, 2
Green-winged teal	0	–	1	1a
American black duck	0	–	2	2
Mallard	4	1a, 1b, 2	4	1a, 2
Hooded merganser	1	1a, 2	3	1a, 2
Total	7	1a, 1b, 2, 3	15	1a, 2

Stage 1a = new-forested beaver ponds, stage 1b = new-open beaver ponds, stage 2 = old-active beaver ponds, stage 3 = abandoned beaver ponds

hooded mergansers were observed only on new-forested and old-active ponds. In northeastern Pennsylvania, Canada geese were detected using old-active beaver ponds; wood ducks and American black ducks using new-forested and old-active ponds; green-winged teal using new-forested ponds; and mallards and hooded mergansers using all stages.

Twenty-two waterfowl broods were observed during this study; 7 in the central region and 15 in the northeastern region. Of the seven broods detected in the central region, two were wood duck, four were mallard, and one was a hooded merganser brood. In the northeastern region, three Canada goose, two wood duck, one green-winged teal, two American black duck, four mallard, and three hooded merganser broods were detected. Waterfowl broods were observed on all successional stages in the central region, though broods were present only on new-forested and old-active beaver ponds in the northeast (Table 8.2).

Waterfowl abundance increased significantly with beaver pond area for sites in the northeastern region, but not those in the central region. This difference was most likely due to the difference in pond areas between the two regions. The larger ponds of the northeastern region were nearly three times larger than the largest pond in the central region. The limited range in area of beaver ponds in the central region may have resulted in relatively low waterfowl abundances.

Although waterfowl abundances did not differ significantly among successional stages of beaver ponds in the central region, the trend suggests highest abundances on new-forested beaver ponds. Such results concur with findings reported by Stanton (1965), Renouf (1972), and Brown and Parsons (1979). These studies attributed higher waterfowl abundances on new beaver ponds to greater amounts of wooded cover. Habitat variables of this study revealed similar trends from new-forested beaver ponds to abandoned ponds. Generally, tree density decreased with beaver pond succession, while emergent herbaceous vegetation increased.

Inspection of species composition across the successional stages offered additional explanation of the waterfowl abundance patterns. Median values for the wood duck, hooded merganser, and mallard were highest on the new-forested (stage 1a) beaver ponds. Mallards were also observed frequently on old-active ponds (stage 2), where a mix of open water and emergent vegetation provided nesting and foraging habitat.

The number of waterfowl broods per beaver pond (0–3 range) was lower than some studies conducted on beaver ponds in North America. Beard (1953) detected an average of 6.3, 7.5, 6.6, and 7.0 broods on four beaver ponds in Michigan from 1947 to 1949 (pond area approximated 5, 3, 9, and 9 ha, respectively). All beaver ponds included in Beard's study were of successional stages equivalent to old-active and abandoned beaver ponds of this study. In a Wisconsin study, 0 to 6 broods were detected on 15 beaver ponds ranging in area from 0.7 to 11.6 ha, and in pond age from 9 to >53 years of age (i.e., equivalent to old-active and abandoned beaver ponds of this study).

Multiple studies report movement of young broods from small, deep-water beaver ponds to larger, shallower, densely vegetated ponds that provide greater cover and invertebrate food resources (Kirby 1973; Baldassarre and Bolen 1994). Hepp and Hair (1977) observed that one third of the females ($n=9$) moved their broods an average of 3.2 km overland to reach densely vegetated beaver ponds. Claypoole (1997) reported differential habitat use of wood duck broods both daily and throughout the breeding season. Broods of her study fed in shrub vegetation during the day, and aquatic bed vegetation during morning and evening hours. They also shifted from temporarily flooded forested wetlands to emergent wetlands in mid-June. Wood duck broods observed during this study exhibited use of younger beaver ponds within their first month and older ponds within their second month, concurring with results reported above.

Particularly in non-glaciated areas, such as central Pennsylvania, where wetland area and abundance are less than that of the glaciated regions (Tiner 1989), availability of beaver pond wetlands provide important avian breeding habitat. Forested wetlands provide wooded cover for waterfowl, but often lack the necessary surface water. New-forested beaver ponds, however, provide much of the wooded wetlands preferred by wood ducks, American black ducks, and hooded mergansers. In general, the increase in wetland abundance and area of forested, shrub, emergent, and open water wetlands to this region because of beaver ponds benefits many avian species that have diverse habitat structure requirements.

A combination of successional stages of beaver ponds in the landscape will provide the greatest diversity of habitat for waterfowl of different species and phenologic stages. New-forested and old-active beaver ponds were most valuable for waterfowl. As management of beaver ponds becomes a greater concern, the effects should be considered for the entire avian community. A host of passerine, heron, and rail species use beaver ponds of various successional stages, not all of which are similar to those used by waterfowl (Prosser 1998; Grover and Baldassarre 1995; Reese and Hair 1976). A mosaic of all stages within a landscape appears to be most desirable for management of a diverse avian community.

8.2.3 Habitat Use of Birds in Lakes of the Pocono Region

In 1991, we conducted a pilot study to test a method for sampling avian communities on a series of lakes in northeastern Pennsylvania and to determine whether differences between undisturbed and disturbed sites could be detected from a census period consisting of two surveys taken during the breeding season. At the time, we were seeking to determine whether birds could serve as adequate indicators of changes in the condition of a variety of aquatic resources, including vegetated wetlands, riparian corridors, and lake-wetland complexes. Although few birds are completely aquatic, we have found in subsequent studies that they have the potential to be suitable indicators, as useful as entirely aquatic organisms such as fish, sediment diatoms, and zooplankton (Moors 1993; O'Connell et al. 2000; Gyekis 2007). Obviously, obligate avian species such as waterfowl, wading birds, and shorebirds rely primarily on waters for foraging, and nest proximal to shores. In addition, many terrestrial species of birds rely upon aquatic resources during all or part of their life cycle. They may feed on insects that have an aquatic larval stage that may be strongly influenced by changes in the condition of the water body. These changes may be reflected in the composition of avian communities. Data on avian communities is also relatively easy and inexpensive to obtain, often through citizen science surveys, adding another favorable aspect to their use as indicators.

Sampling occurred on 17 lakes of varying size in the glaciated Pocono region in northeastern Pennsylvania. Lakes were selected to represent three size classes (small, 2–20 ha; medium 25–100 ha; large, 150–500 ha), two disturbance classes with regard to surrounding land use and shoreline structure (undisturbed, disturbed), and access for sampling by canoe and on foot. These lakes represent both natural and man-made impoundments exhibiting a wide variety of habitat types and levels of disturbance. Prehistorically, this part of the state experienced extensive modification by glaciers, which carved a multitude of shallow depressions in the original plateaus and river valleys. Approximately 40% of Pennsylvania's wetlands occur in this region and occupy about 20% of the land area (Brooks and Tiner 1989). More recent changes have been the result of converting vegetated forest-shrub wetlands and bogs to open water lakes for vacation homes, and limited extraction of Sphagnum peat for commercial sale.

Small lakes were randomly assigned three sampling stations while medium and large lakes were assigned six and nine stations, respectively. Before sampling, the perimeter of each lake was assessed to determine the major habitat types occurring there. Land use and degree of disturbance were also noted. Those sections having suboptimal conditions for parameters known to influence bird use were classified as disturbed (e.g., hardened shorelines, higher density of dwellings, lack of fringing aquatic vegetation, less natural vegetation along shores). Sampling was performed in the early morning hours during June and July 1991 by experienced technicians. The technicians visited each lake prior to the scheduled survey to locate the transects and access points and to ensure that there would be no problems on the morning of the survey.

Table 8.3 Comparisons in avian species richness across three size classes of lakes, in either a relatively undisturbed and disturbed condition, in northeastern Pennsylvania

Lake condition	Undisturbed	Disturbed
Lake size		
Small	37	30
Medium	40	29
Large	41	28

Each station was sampled using a set transect length of 1,000 m. Five-minute point counts (based on 0.2 ha-diameter plots) at 100-m intervals were used to census the avian communities along these transects. Presence/absence and abundance data were gathered using visual and auditory cues. The focus was on the edge habitat along the shoreline, although birds observed in the open water were recorded. Once the data was collected, species lists, species richness, and bird abundance were compiled and compared for disturbed and undisturbed sections of each lake size. Jaccard's coefficient of community, and the percent similarity method were used to aid in the comparisons. Jaccard's coefficient equals 0 (or 0%) when there is no similarity, while 1 (or 100%) indicates that two areas have exactly the same community composition.

In each size category, the lakes showed distinct differences in species richness between undisturbed and disturbed communities, with the undisturbed sections always having the greater numbers of species. Undisturbed areas on small lakes had 19% greater richness than disturbed areas, while medium and large lakes showed 28% and 32% differences, respectively (Table 8.3). Thus, with regard to species richness, the level of disturbance appears to influence the number of species present vs. lake size.

The substantial variation in species richness between undisturbed and disturbed sections indicates a difference in community structure. Species richness is, however, only one measure of the number of species present in a given community, and represents only a single aspect of its structure. Indices of community similarity provide another way to examine the community. Using Jaccard's coefficient of community, we determined that undisturbed and disturbed sections of the small lakes harbor avian communities that are 52% similar, the measures were 35% for medium lakes and 38% for large lakes.

Coefficients of community are useful indicators with regard to the presence or absence; however, they overlook the relative abundance of species (Brower and Zar 1984). The percent similarity measure was used to reflect this aspect of community structure yet the degree of similarity between undisturbed and disturbed sites remained virtually unchanged for each size category (50% for small, 36% for medium, and 38% for large). For both of these measures, it is interesting to note the large difference between the small lakes and the other two categories, as well as the closeness of values for medium and large lakes.

An analysis of guilds provided more insight into the response of the avian community to disturbance on these lakes. Using rankings for habitat specificity set forth by Brooks and Croonquist (1990), we compared the percentages of species that are specialists (ranked 5) and/or landscape dependent (ranked 3) that occur on disturbed and undisturbed sited in each size category. A ranking of 1 indicated that a species is a generalist and capable of tolerating a substantial amount of habitat disturbance.

For small disturbed sites, 17% of the community had a ranking of 3 or higher, compared to 30% on the undisturbed sites. The percentages of specialist species were 11% and 3% for undisturbed and disturbed, respectively. On medium lakes, in disturbed communities, 23% ranked 3 or above with 10% rated as specialists. Undisturbed sites showed 39% ranked 3 or above with 12% being specialists. Disturbed sites on large lakes had 28% of the community ranking 3 or higher while undisturbed sites showed 44%. The percentage of specialists was 7% and 12%, respectively. These values show that across the board, undisturbed sites attracted more habitat specialists that disturbed lakes. These are the groups that will best indicate community responses to impacts. Also notable are the appearances on undisturbed sites of several species that are dependent upon a relatively undisturbed landscape, and have been shown to be sensitive to changes in land use, e.g., veery (*Catharus fuscescens*), hermit thrush (*C. guttatus*), and Louisiana waterthrush (*Parkesia motacilla*). Certain habitat-specific birds like the black-throated blue warbler (*Setophaga caerulescens*) and the chestnut-sided warbler (*S. pensylvanica*) were detected in undisturbed lakes in every size category while remaining absent in all disturbed lakes.

These results indicate that there is a strong association between disturbance levels along lakes and the composition of the avian community. They suggest that bird communities do indeed have significant potential as indicators of the condition of surface water systems. Of course, these results are only preliminary, but they suggest that there are negative impacts on bird communities from lake development. Further analysis of avian guilds using these lakes should be completed to help determine which taxa can provide the most useful information regarding changes in aquatic resources.

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8.2.4 Wetland Songbirds in the Ridge and Valley Province

In addition to wading birds, waterfowl, and rails, many other species depend on wetlands or at least depend on vegetation structure or microhabitat conditions frequently

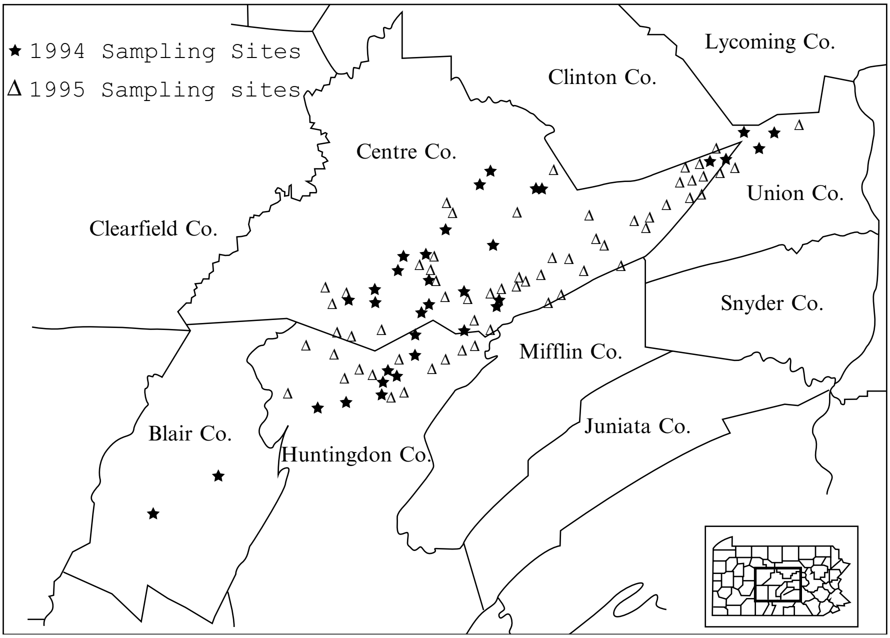

Fig. 8.2 Approximate locations of 95 wetlands in Central Pennsylvania sampled for breeding songbirds during 1994 and 1995

associated with wetlands. The species assemblage using a wetland at any given time will be comprised of both obligate and facultative wetland species. In 1994 and 1995, Riparia conducted a study of breeding bird communities in small wetlands that were less likely to provide habitat for obligate species such as waterfowl. Instead, this study focused on the life history and habitat associations of songbirds breeding in the wetlands (Gaudette 1998). We tested the general hypothesis that wetlands providing habitat for many species with specialist life history traits were qualitatively different than wetlands in which such species were underrepresented.

Fieldwork for this study involved point counts for breeding birds located along transects in wetlands and extending 1 km away from the wetlands into adjacent uplands. Sampling permitted the ability to characterize not just the wetland itself but the immediate watershed area around the wetland as well. We sampled 95 wetlands in the predominantly forested Ridge and Valley Physiographic Province of Central Pennsylvania (Fig. 8.2). The sites permitted analysis to determine species' reliance on wetlands and their response to land cover disturbance from forest fragmentation and agricultural or urban development.

Community organization in this study was determined through analysis of avian response guilds, or groups of species united by commonalities of life history (e.g., number of broods, primary nesting substrate) or response to anthropogenic stressors. For example, many forest-breeding birds in the MAR are limited in distribution to large blocks of mature forest where conditions for reproductive success are more favorable (Brittingham and Temple 1983; Wilcove 1985; Askins 1995). Results indicated that communities organized broadly according to vegetation structure and land cover. Forested landscapes supported communities with many species that are sensitive to anthropogenic disturbances. These species often displayed attributes of sensitivity that had little direct bearing on a specific relationship to "forest." For example, single-brooded species and long-distance migrants were better represented in forested landscapes than in other sites along the disturbance gradient. Disturbed sites, whether dominated by urban or agricultural land cover, supported bird communities dominated by exotic species, omnivores, and species that normally raise more than one brood per season.

Wetland dependency was limited among the primarily Passerine species detected in the study. We found a strong association with forested headwater streams for Louisiana waterthrush (*Parkesia motacilla*), northern waterthrush (*P. noveboracensis*), dark-eyed junco (*Junco hyemalis*), and acadian flycatcher (*Empidonax virescens*). Shrub wetlands provided habitat for alder (*E. alnorum*) and willow (*E. traillii*) flycatchers, as well as yellow warbler (*Dendroica petechia*) and swamp sparrow (*Melospiza georgiana*). Open water was an important determinant for belted kingfisher (*Ceryle alcyon*) and tree swallow (*Tachycineta bicolor*).

Results of this study helped focus attention in Riparia on related and subsequent studies that investigated landscape condition inferences drawn from the life history composition of the species they supported. This study also illustrated the importance of wetlands in providing vegetation structure attractive to species that otherwise would be considered upland species.

8.3 Avian Diversity in Context: Indicators of Ecological Condition

8.3.1 Conceptual Development of the Bird Community Index

In the 1990s, the USEPA's Environmental Monitoring and Assessment Program (EMAP) committed to the development of indicators of ecological condition that could be applied across broad areas of the country and comprise an important component of a "national environmental scorecard." At the federal level, it is important to prioritize the conservation of entire ecoregions, as these vary both in region-wide stressors and/or their likelihood of ecological restoration. Is it better, for example, to invest in hydrologic restoration of the Everglades or address exurban residential development in the Southern Rocky Mountains? A standardized suite of ecological assessment tools can facilitate comparisons like these among ecoregions by estimating the proportion of land area in various states of condition within those ecoregions. Through collaborative research with EMAP, Riparia set out to develop a songbird-based indicator of ecological condition suitable for the Mid-Atlantic Highlands, which included all mountainous terrain in the MAR.

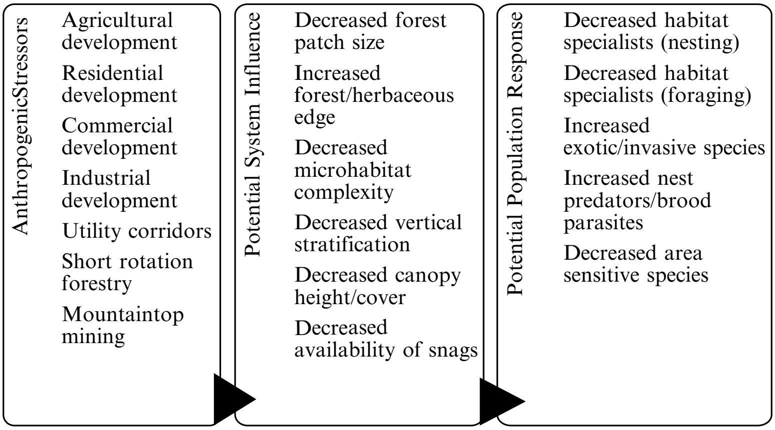

Fig. 8.3 Conceptual model of the Bird Community Index, illustrating hypothetical links from stressor to ecosystem influence to species' responses (adapted from O'Connell et al. 2007)

Indicators for ecological assessment are most robust and useful when they adhere to the qualities of the index of biotic integrity or IBI. The IBI concept was pioneered by Karr and Dudley (1981) who described integrity as "…a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region." Importantly, "diversity" comprises just one piece of our understanding of biotic integrity. Despite their widespread *de facto* use as indicators of condition, indices such as species richness and Shannon-Weiner diversity are often ill-suited to the purpose of broad scale ecological assessment. This is because simple numeric diversity indices often peak at intermediate levels of anthropogenic disturbance, providing similar results for both highly degraded and near pristine sites. Research both confirming and refuting the "intermediate disturbance hypothesis" (Connell 1978) has come from multiple taxa and with data collected at multiple scales. For the purpose of an applied model for ecological assessment, however, the limitations of simple numeric indices are clear: If two sites support 20 species we have little information with which we can order those sites on a scale of anthropogenic disturbance until we know *which* 20 species they are. A biota-based indicator should detect information on the population, distribution, or behavior of organisms that results from influences on ecosystems that stem from specific anthropogenic stressors on that ecosystem (Fig. 8.3).

The idea that some species can serve as indicators of ecological function is well-founded, and it makes intuitive sense. For example, a nesting pair of Red-cockaded Woodpeckers (*Picoides borealis*) indicates that the highly specific habitat needs of that species are being supplied over an area large enough to support at least one pair's home range. Nesting Red-cockaded Woodpeckers can be a sign that pine trees have achieved maturity, and that fire return intervals are sufficient to create an open forest structure with an abundant herbaceous understory (Fig. 8.4).

Fig. 8.4 This Red-cockaded Woodpecker is an indicator of habitat that supports multiple species dependent on pine savannas, but its absence from a site would not necessarily mean that the site is in degraded condition. (Photograph by M. Lanzone)

This, in turn, indicates that habitat is also being provided for a host of other species that rely on a fire-maintained pine savanna structure, such as Bachman's Sparrow (*Aimophila aestivalis*).

There are, however, problems with placing too much emphasis on the occurrence or abundance of single indicator species, especially at fine scales. Although the confirmed occurrence of breeding Red-cockaded Woodpeckers would be seen as a positive indicator for pine savannas, the absence of the species leads to equivocal interpretations. That absence might mean that the area is incapable of supporting Red-cockaded Woodpeckers, but it just as easily could be the result of low population density and a clumped distribution of the woodpeckers. In other words, there will often be suitable habitat available that rare species have not colonized, simply because the suitable habitat is not saturated. To combat that problem, we decided to pursue an indicator that could provide unambiguous information from any site sampled in the study area. When every species encountered contributes information for the indicator, a major source of ambiguity in indicator interpretation is eliminated.

From city centers to rugged mountaintops, there is some form of a predictable assemblage of bird species in all terrestrial landscapes of the MAR. This is the primary strength of using bird community data for an indicator intended for application to an entire ecoregion. Birds also incorporate stressors acting at multiple scales. For example, cavity-nesting birds rely on a supply of potential nest trees (generally snags) occurring within the area of an individual's home range. Such fine-scale

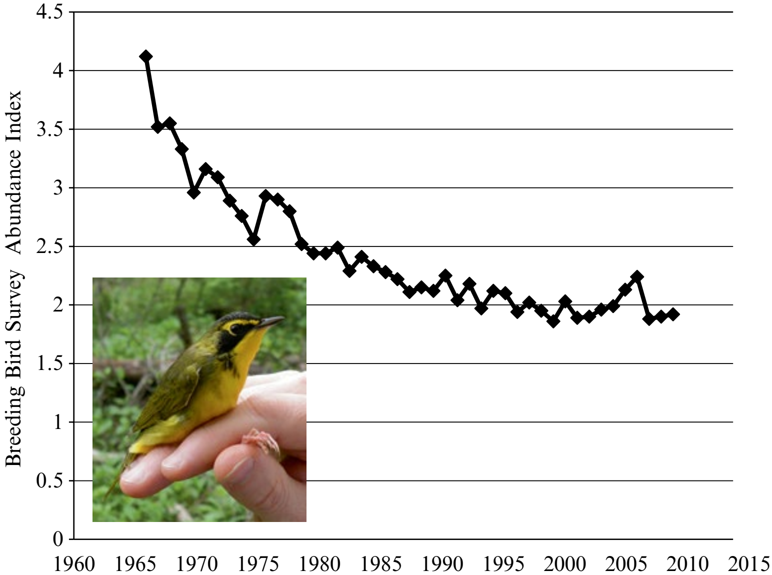

Fig. 8.5 Throughout the MAR, forest-breeding species such as the Kentucky Warbler (*Geothlypis formosus*) have experienced long-term population declines. (Data from Sauer et al. 2011; photograph by T. O'Connell)

features are balanced by species that migrate long distances between breeding and wintering home ranges. Long-distance migrants might incorporate stressors at continental scales, in addition to their fine-scale habitat needs. Thus, bird populations tend to be heavily influenced by the composition of landscapes, with distinct communities resulting in landscapes dominated by major land cover classes such as forest, grassland, and urban.

In addition, birds are more efficiently sampled than some other taxonomic groups, with a comparatively large number of professionals and amateurs who are skilled in field identification. Birds are identified at the moment of detection, rather than collected, sorted, and identified through the use of a dichotomous key in the laboratory, which can greatly increase the time required to generate community data to analyze. Most songbirds and related groups synchronize breeding activities to a few months each spring and advertise their presence on breeding territories through songs and distinctive calls. Thus, conducting general auditory surveys for birds during the spring breeding season is an efficient way to sample the majority of species that occur in a given region. For these reasons, an ecological indicator populated with bird community information is highly attractive for ecological assessments intended for broad scales across entire ecoregions like the MAR.

Prior to our development of the BCI, it was well established in the literature that several species of native birds in the MAR had experienced population declines and range contraction in landscapes where regional forest cover was lost to anthropogenic disturbances (Fig. 8.5). What we did not know was how much forest cover

Table 8.4 Operational response guilds for Pennsylvania birds and mammals, proposed to assess cumulative impacts to wetlands and riparian areas (adapted from Croonquist and Brooks 1991)

Response guilds	Scores	Response guilds	Scores
Wetland dependency		Habitat specificity	
Obligate species (>99% in wetlands)	5	Alpha–stenotypic, specialist	5
Facultative wet (usually in or near wetlands)	3	Gamma–landscape dependent	3
Facultative (wetlands not essential)	1	Beta–generalist, edge	1
Facultative dry (occasional or no use)	0		
Upland (>99% in wetlands)	0		
Trophic level		Seasonality (birds only)	
Carnivore, specialist (restricted diet)	5	Neotropical migrant	5
Carnivore, generalist	4	Short-distance migrant	4
Herbivore, specialist (e.g., nuts, nectar)	3	Year round resident	3
Herbivore, generalist	2	Nonbreeding season resident only	2
Omnivore (plants or animals)	1	Migratory transient	1
		Occasional	0
Species status			
Endangered, endemic, of concern	5		
Commercial, recreational value	3		
Other native species	1		
Exotic	0		

needed to be retained for Mid-Atlantic landscapes to support their full complement of native species. In addition, we did not know how the occurrence of those avian species most sensitive to forest loss correlated with the occurrence of other taxonomic groups. For the BCI to be useful as an ecological indicator, it needed to address both of those issues. We also recognized that an indicator too heavily weighted toward native forest fauna might overlook the biodiversity value of certain non-forested native landscapes in the region.

We decided, therefore, to develop an indicator that addressed multiple aspects of species' life history traits. This approach was developed by Croonquist and Brooks (1991) who assigned Pennsylvania mammals and birds *a priori* into operational response guilds, or groups of species with similar life history traits that are similarly sensitive to anthropogenic stressors (Table 8.4). The intent of that study was to determine if analysis of avian and mammalian community data viewed through the lens of response guilds could reveal cumulative, negative influences of anthropogenic disturbances in wetlands. The approach was to compare two watersheds in the Ridge and Valley province of central Pennsylvania, one with minimal anthropogenic disturbances and one with obvious sources of degradation such as residential and agricultural development along the primary stream reach. White Deer Creek with its 94% forested watershed served as the model of reference condition and Little Fishing Creek (70% forested with at least 25% residential and agricultural cover) illustrated moderately degraded condition (Croonquist and Brooks 1991).

Results indicated that both avian and mammalian communities were more similar throughout the less disturbed White Deer Creek than through the more disturbed Little Fishing Creek watershed. In other words, the land cover change in lower reaches of Little Fishing Creek had a strong influence on the species composition in different portions of the watershed. Simple indices of community change, however, do not illustrate which species gained or lost habitat in response to human influences. In the analysis of response guilds, mammalian guilds showed no consistent pattern with land cover. Avian response guilds, however, illustrated a statistically significant and predictable relationship to land cover disturbance, with habitat specificity and seasonality providing the most consistent information (Croonquist and Brooks 1991). This result provided the impetus for a more extensive analysis of avian response guilds for development of a broad scale ecological indicator.

Fieldwork to develop the BCI focused on sampling from a number of sites representing a gradient of conditions that might be encountered anywhere in the Mid-Atlantic Highlands. Through prior research in Riparia, Brooks et al. (1996) characterized the ecological condition of multiple wetland sites in the Ridge and Valley physiographic province of central Pennsylvania. These sites had been featured in long-term research involving plants, soils, amphibians, and landscape composition, thereby providing an independent characterization of biotic integrity to which data on avian response guilds could be compared. We sampled 34 sites from this gradient, all centered on small (<15 ha) wetlands from which we established two, 1-km sampling transects into adjacent uplands. Avian sampling consisted of breeding season point counts for songbirds and other small landbirds (e.g., doves, woodpeckers) with points located along the transects emanating from the central wetland (O'Connell et al. 1998).

To begin the conceptual work of packaging response guild information into a quantitative index, we compiled literature on various aspects of each species' life history, e.g., typical nesting substrate, number of broods, trophic position, and migratory behavior. From an initial 32 behavioral and physiological life history traits, we iteratively examined correlation matrices to eliminate redundancy, and ultimately included 16 response guilds in 8 categories in the BCI (Table 8.5). We grouped those guilds into three major categories intended to convey information about structural, functional, and compositional ecosystem elements (Noss 1990). Finally, we hypothesized each response guild as "specialist" or "generalist" with the interpretation that sites dominated by generalist guilds would indicate pervasive anthropogenic disturbance, and *vice versa*. Because the guild categories address different aspects of life history, each species is assigned to multiple response guilds. For example, Carolina Chickadee (*Poecile carolinensis*) is simultaneously a bark-probing insectivore, resident, single-brooded, forest generalist.

In an additional effort to maximize utility of the BCI, we designed it to be applicable without abundance data for individual species. This is because many potential end-users (e.g., parks, government agencies) might have the ability to generate species lists from their monitoring efforts, but robust estimates of species abundance are often unavailable due to the added sophistication in sampling design and analysis

Table 8.5 Sixteen specialist and generalist response guilds in eight categories incorporated into the Bird Community Index

Integrity element	Guild category	Response guild	Specialist	Generalist
Functional	Trophic	Omnivore		X
	Insectivore foraging	Bark prober	X	
		Ground gleaner	X	
		Upper canopy	X	
		Lower canopy	X	
Compositional	Disrupter	Nest predator/ brood parasite		X
	Origin	Exotic		X
	Migratory	Resident		X
		Temperate migrant		X
	Fecundity	Single-brooded	X	
Structural	Nesting	Canopy	X	
		Shrub		X
		Grassland-ground	X	
		Forest-ground	X	
	Primary habitat	Forest generalist		X
		Interior forest	X	

needed to produce those estimates. For application of the BCI, simple species lists suffice because the response guilds are quantified as the proportion of species in each guild as a function of the total number of species on the list. For example, 5 single-brooded species from a total list of 20 species results in a 0.25 for the single-brooded response guild.

The BCI ranks high proportions of specialist guilds as indicative of ecosystem integrity. Using exploratory data analysis, we identified statistically separable categories of occurrence for individual guilds and applied numeric ranks to those different proportions of guilds. The BCI sums the ranks of the 16 guilds so comparatively high proportions for the 9 specialists guilds and low proportions for the 7 generalist guilds would produce a high BCI score.

As a first test of the BCI, we applied it to the 34 central Pennsylvania sites that had been independently ranked in three categories of ecological condition based on soils, hydrology, wetland plants, and habitat suitability index models (Brooks et al. 1996). The BCI correctly placed those sites in their respective categories, thereby illustrating its ability to integrate stressors affecting multiple ecosystem attributes, as opposed to merely being a "bird" indicator. All 16 guilds demonstrated a statistically significant relationship with at least two categories of condition, and 5 guilds were able to discriminate among all three categories (Table 8.6). For example, single-brooded species averaged 70% of the species in high-integrity sites, 56% in medium-integrity sites, and 40% in the low-integrity sites.

Table 8.6 Mean proportion (±SE) of total species at a site in each guild in each of three ecological integrity categories: high, medium, and low

Response guild	High integrity ($n=9$, rank = 3)	Medium integrity ($n=12$, rank = 2)	Low integrity ($n=13$, rank = 1)	P
Omnivore	0.30 ± 0.02^{a}	0.47 ± 0.02^{b}	0.57 ± 0.02^{c}	<0.001
Bark prober	0.12 ± 0.02^{a}	0.08 ± 0.01^{b}	0.03 ± 0.01^{b}	0.001
Ground gleaner	0.10 ± 0.01^{a}	0.05 ± 0.01^{b}	0.03 ± 0.01^{b}	<0.001
Upper-canopy forager	0.13 ± 0.01^{a}	0.08 ± 0.01^{b}	0.06 ± 0.01^{b}	<0.001
Lower-canopy forager	0.23 ± 0.02^{a}	0.19 ± 0.01^{ab}	0.13 ± 0.02^{b}	0.002
Nest predator/brood parasite	0.09 ± 0.01^{a}	0.14 ± 0.01^{b}	0.14 ± 0.02^{b}	0.024
Exotic	0.00 ± 0.00^{a}	0.02 ± 0.01^{ab}	0.06 ± 0.02^{b}	0.020
Resident	0.29 ± 0.02^{a}	0.40 ± 0.02^{b}	0.40 ± 0.02^{b}	0.004
Temperate migrant	0.22 ± 0.02^{a}	0.24 ± 0.01^{a}	0.32 ± 0.01^{b}	<0.001
Single-brooded	0.70 ± 0.03^{a}	0.56 ± 0.02^{b}	0.41 ± 0.02^{c}	<0.001
Canopy nester	0.36 ± 0.02^{a}	0.32 ± 0.02^{ab}	0.27 ± 0.01^{b}	0.003
Shrub nester	0.20 ± 0.02^{a}	0.29 ± 0.02^{b}	0.32 ± 0.02^{b}	<0.001
Open-ground nester	0.01 ± 0.01^{a}	0.06 ± 0.01^{b}	0.10 ± 0.01^{c}	<0.001
Forest-ground nester	0.22 ± 0.01^{a}	0.08 ± 0.01^{b}	0.03 ± 0.01^{c}	<0.001
Forest generalist	0.36 ± 0.02^{a}	0.44 ± 0.02^{b}	0.31 ± 0.02^{a}	<0.001
Interior forest obligate	0.43 ± 0.02^{a}	0.15 ± 0.02^{b}	0.05 ± 0.02^{c}	<0.001

Significant differences in guild proportions are based on one-way ANOVA ($df = 2, 33$) with Tukey's test for multiple comparisons. Within rows, values with different superscripts are significantly different at $P < 0.05$ (table from O'Connell et al. 2000)

8.3.2 Application and Interpretation of the Bird Community Index in the MAR

Armed with the results of the central Pennsylvania study that the BCI successfully discriminated independently derived categories of condition, Riparia embarked on a 2-year effort to survey breeding birds from a blocked random selection of sites from the larger Mid-Atlantic Highlands Assessment (MAHA) area. The MAHA area encompasses approximately 168,420 km^2 in the mountainous physiographic provinces of USEPA Region III, and is dominated by the Blue Ridge, Ridge and Valley, Allegheny Plateau, and Ohio Hills physiographic provinces of Pennsylvania, Maryland, Virginia, and West Virginia. The MAHA served as the first test case for the BCI, both to examine landscape characteristics associated with BCI scores and to characterize the relative proportions of that assessment area in various states of condition.

Sampling for BCI fieldwork was conducted within the framework of the USEPA's EMAP probability sampling grid (White et al. 1992). Messer et al. (1991) described the reference grid density to include approximately 12,500 points in the Lower 48 US States, with each point approximately 27 km away from neighboring points in a triangular pattern. Points from the grid can be randomly selected to provide information that can be used to develop a snapshot of condition across the entire assessment region, based on the specific indicators applied to the selected points. For our study, we used two selections from the reference grid density (one for each

Fig. 8.6 Approximate locations of 126 sites surveyed in 1995 and 1996 from the EMAP probability sampling grid in the Mid-Atlantic Highlands Assessment area

year of the study), and collected information on breeding birds and the structure and composition of vegetation and landscapes around our sample sites. By sampling locations from the EMAP grid, we were able to consider proportions of sites in different states of ecological condition to be representative of proportions of the assessment region, i.e., the MAHA area (Fig. 8.6).

Sampling from randomly selected locations presented some difficulties in the field. For example, some points occurred in lakes where it would have been untenable to establish survey points for breeding songbirds. Some points occurred in roadless areas in National Forests that were difficult to access given the time constraints of field sampling. Field crews needed to make a decision in the field if it was worth the time to invest in a long hike to get to a particular point or to sample from similar local habitat with easier access so that a greater number of sites could ultimately be sampled for the study. The most isolated point we surveyed was a 4-km hike from the nearest road access in Virginia's Shenandoah National Park. The majority of sites occurred on private lands, and access necessitated prior contact and permission from those landowners. Many of these landowners were absentee or did not respond to initial written requests for access so field crews conducted many door-to-door inquiries in obtaining access. Given the scale of our sampling at each site (five points sampled along a randomly oriented 1-km transect; Fig. 8.7), we often needed access permission from multiple landowners, especially in urbanizing areas. Where access permission was denied or difficult to obtain, field crews made the decision in the field to abandon the site that had been identified or sample it as closely as possible from a nearby road.

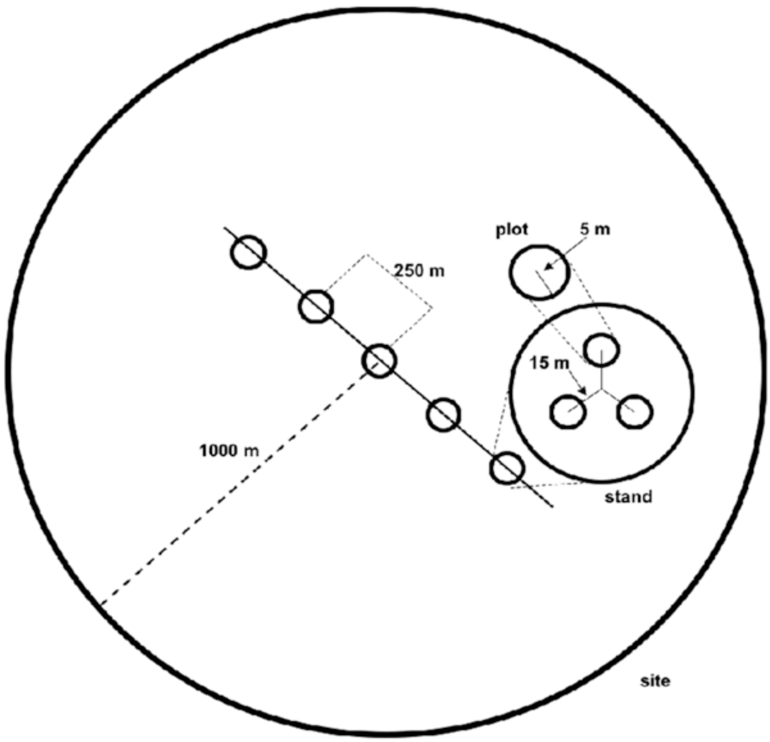

Fig. 8.7 Sampling design for the BCI study in the Mid-Atlantic Highlands Assessment area. The randomly selected coordinates were used to identify the center point of a 5-point, 1 km sampling transect. We explored multiple land cover scales around sites, including the 1-km radius view illustrated above. Ground-level vegetation structure and composition was assessed at finer stand and plot scales

At each bird sampling point, we surveyed songbirds with a 10-min, 30 m-radius point count between sunrise and 10:00 h EDT (Ralph et al. 1993). For these analyses, we used a total species list compiled from unlimited radius point counts at each of the five plots. We sampled a suite of vegetation variables to characterize the local habitat. We recorded the percentage herbaceous cover of graminoids, forbs, mosses, and ferns in three, 5-m radius, circular subplots located 15 m from plot center at 120, 240, and 360°. Also in the subplots, we recorded the percentage cover of shrubs from 0.00–0.50, 0.051–2.00, and 2.01–5.00 m, as well as the percentage canopy cover of overstory trees. From plot center, we used an angle gauge to sample trees over 10 cm dbh. All live trees were identified to species and the dbh was recorded for trees and snags. In addition, at each plot we recorded canopy height, slope, and aspect.

To characterize the local landscape configuration, we obtained aerial photographs of the circular area bisected by each transect. For the 1995 and 1996 sites, this resulted in a circular site (i.e., a “landscape circle”) with a 0.5 km radius covering an area of approximately 79 ha. The photographs were interpreted and polygons of

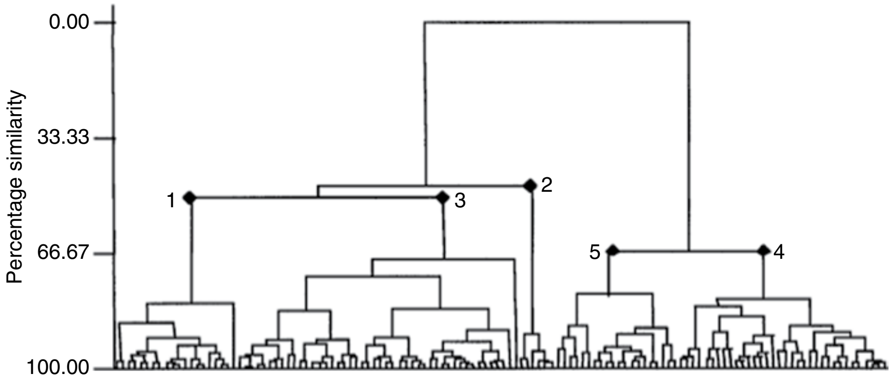

Fig. 8.8 Cluster analysis dendrogram of bird communities in the Mid-Atlantic Highlands Assessment (MAHA) area determined through the proportions of response guild representation (reproduced from O'Connell et al. 2000)

six cover types were digitized in a GIS and entered into a modified version of the spatial analysis software package SPAN (Miller et al. 1997). The SPAN output provided information on landscape diversity, dominance, and contagion, the amount of edge between cover types, and the areal coverage within the circular site of urban development, agricultural land, forest, woody shrubs, open water, and barren land.

We ultimately surveyed 126 sites, and built the BCI with data from 112 species of birds. We used exploratory data analysis to illustrate the number of distinctly different communities of birds that could be identified among those species grouped into their respective response guilds, and ranks of those response guild proportions to determine how many of those communities sorted into different categories of BCI scores. A cluster analysis of bird community profiles of 16 guilds at the 126 probability-based sample locations in the entire MAHA area indicated five distinct groupings of sites with a mean within-cluster sum of squares of 1.28 (Fig. 8.8). We ranked clusters according to the relative proportions of specialist and generalist guilds at the sites in each cluster. The ranking scheme allowed us to place the five clusters into four distinct categories of BCI scores. According to our bird community-based criteria for defining biotic integrity, approximately $16 \pm 5.5\%$ of the MAHA area supported the highest integrity communities, $27 \pm 6.8\%$ was high integrity, $36 \pm 7.3\%$ was medium integrity, and 21% of the MAHA area supported two separate categories of low-integrity bird communities (i.e., "low 1" $= 16 \pm 5.5\%$ and "low 2" $= 5 \pm 3.4\%$) (Fig. 8.9).

Because we built the BCI to hierarchically compile scores from 16 response guilds, the composite BCI score used to determine the number of different categories of biotic integrity can be readily divided into subscores for each guild. We found dramatic differences in response guild representation along the gradient of anthropogenic disturbance, and some indicated profound ecological differences among the categories (Table 8.7, Fig. 8.10). For example, insectivorous species greatly outnumbered omnivores in sites indicating high integrity. At sites indicating

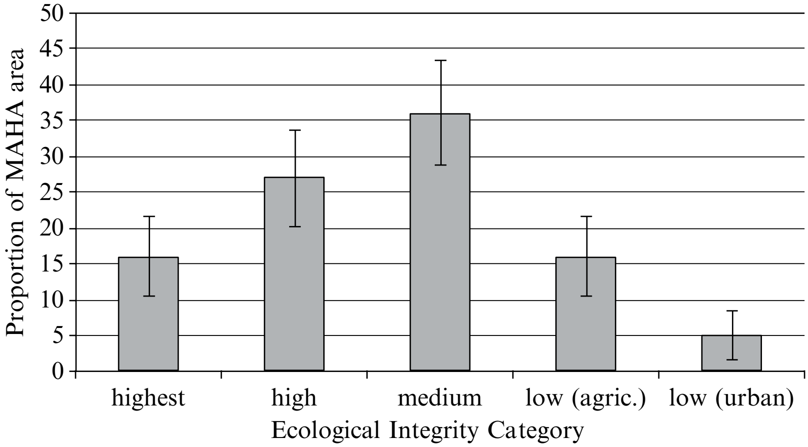

Fig. 8.9 Relative proportions of the Mid-Atlantic Highlands Assessment (MAHA) area assigned to different categories of condition using the Bird Community Index. Error bars represent the 95% CI estimate for the percentage of land area in the MAHA area supporting bird communities indicative of the five categories of biotic integrity. (Adapted from O'Connell et al. 2000)

low integrity, that relationship was reversed and omnivores became dominant. Thus, the predominant trophic pathway that evolved in the MAHA region ceased to function in sites indicating low integrity. The most dramatic losses in insectivores were represented by species that forage in the upper canopy, on the ground, and on the bark of trees. Among compositional guilds, exotic species did not occur at any sites in the "highest" and "high" integrity categories but increased to an average of 16% of the species at certain low-integrity sites. Single-brooded species accounted for 74% of the species at the highest integrity sites, but were only 38% of the species at low-integrity sites. Forest-associated ground nesting birds were 21% of the species at the highest integrity sites, but were not represented at all at low-integrity sites.

We found that the different bird communities indicated by cluster analysis did not always separate according to overall BCI score. There were two distinctly different community types that shared indistinguishable BCI scores at the "low" end of the biotic integrity gradient. One community supported a greater proportion temperate migrants and ground nesting birds of grasslands; we interpreted this community as dominated by landscapes resulting from agricultural disturbances. The other community was more dominated by resident species of birds, and supported the highest proportions of exotic species and nest predators/brood parasites in the MAHA area. We considered this latter community to be indicative of urban disturbances.

The next step in BCI application was to determine how BCI scores developed from theoretical responses of avian life history traits to habitat actually correlated with real habitat features, either remotely sensed or measured in the field. As expected, mature forested cover in the local landscape was a predominant driver of BCI scores. Forest cover was similar between highest and high-integrity sites, but

Table 8.7 Proportion (±SE) of total species at a site in each guild in each of five ecological condition categories

Response guild	Highest ($n=20$)	High ($n=34$)	Medium ($n=46$)	Low 1 ($n=20$)	Low 2 ($n=6$)	P
Omnivore	0.24 ± 0.01^{A}	0.37 ± 0.01^{B}	0.45 ± 0.01^{C}	0.61 ± 0.01^{D}	0.53 ± 0.00^{D}	<0.001
Bark prober	0.17 ± 0.01^{A}	0.11 ± 0.01^{B}	0.08 ± 0.01^{C}	0.03 ± 0.01^{D}	0.03 ± 0.01^{CD}	<0.001
Ground gleaner	0.12 ± 0.01^{A}	0.10 ± 0.01^{A}	0.06 ± 0.01^{B}	0.03 ± 0.00^{C}	0.03 ± 0.00^{BC}	<0.001
Upper-canopy forager	0.18 ± 0.01^{A}	0.15 ± 0.01^{A}	0.10 ± 0.01^{B}	0.04 ± 0.01^{C}	0.01 ± 0.01^{C}	<0.001
Lower-canopy forager	0.21 ± 0.01^{A}	0.17 ± 0.01^{A}	0.17 ± 0.01^{A}	0.12 ± 0.01^{B}	0.14 ± 0.02^{AB}	<0.001
Nest predator/ brood parasite	0.07 ± 0.01^{A}	0.11 ± 0.01^{B}	0.10 ± 0.01^{B}	0.16 ± 0.01^{C}	0.21 ± 0.02^{D}	<0.001
Exotic	0.00 ± 0.00^{A}	0.00 ± 0.00^{A}	0.02 ± 0.00^{B}	0.07 ± 0.01^{C}	0.16 ± 0.03^{D}	<0.001
Resident	0.28 ± 0.02^{A}	0.34 ± 0.01^{B}	0.35 ± 0.01^{AB}	0.42 ± 0.02^{C}	0.69 ± 0.03^{D}	<0.001
Temperate migrant	0.16 ± 0.02^{A}	0.18 ± 0.01^{A}	0.26 ± 0.01^{B}	0.36 ± 0.01^{C}	0.19 ± 0.01^{AB}	<0.001
Single-brooded	0.74 ± 0.01^{A}	0.68 ± 0.01^{B}	0.53 ± 0.01^{C}	0.35 ± 0.01^{D}	0.38 ± 0.03^{D}	<0.001
Canopy nester	0.37 ± 0.02^{A}	0.37 ± 0.01^{A}	0.30 ± 0.01^{B}	0.25 ± 0.01^{C}	0.29 ± 0.01^{BC}	<0.001
Shrub nester	0.19 ± 0.01^{A}	0.22 ± 0.01^{A}	0.27 ± 0.01^{B}	0.29 ± 0.01^{B}	0.19 ± 0.04^{A}	<0.001
Open-ground nester	0.01 ± 0.00^{A}	0.02 ± 0.01^{AB}	0.07 ± 0.01^{C}	0.13 ± 0.01^{D}	0.06 ± 0.01^{BC}	<0.001
Forest-ground nester	0.21 ± 0.01^{A}	0.18 ± 0.01^{A}	0.09 ± 0.01^{B}	0.03 ± 0.01^{C}	0.00 ± 0.00^{C}	<0.001
Forest generalist	0.35 ± 0.01^{A}	0.38 ± 0.01^{A}	0.37 ± 0.01^{A}	0.27 ± 0.02^{B}	0.30 ± 0.02^{AB}	<0.001
Interior forest obligate	0.49 ± 0.01^{A}	0.36 ± 0.01^{B}	0.17 ± 0.01^{C}	0.06 ± 0.01^{D}	0.05 ± 0.01^{D}	<0.001

Significant differences in guild proportions are based on one-way ANOVA ($df=4, 125$) with Tukey's test for multiple comparisons. Within rows, values with different superscripts are significantly different at $P<0.05$; table from O'Connell et al. (2000)

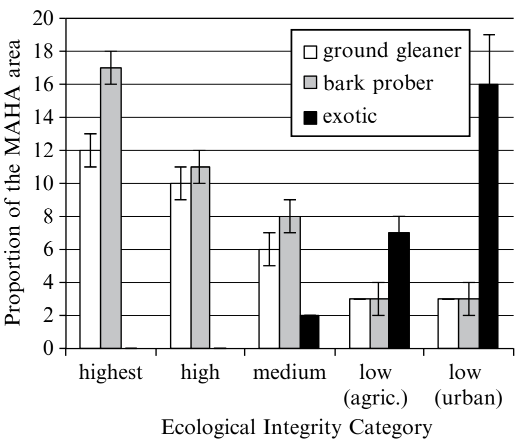

Fig. 8.10 Mean percentages (±SE) of three response guilds at sites in five categories of ecological integrity determined with the Bird Community Index

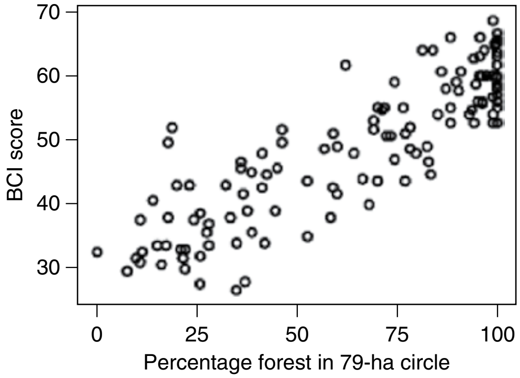

Fig. 8.11 Linear relationship between mature forested cover and Bird Community Index score in the Mid-Atlantic Highlands Assessment area; reproduced from O'Connell et al. (2000)

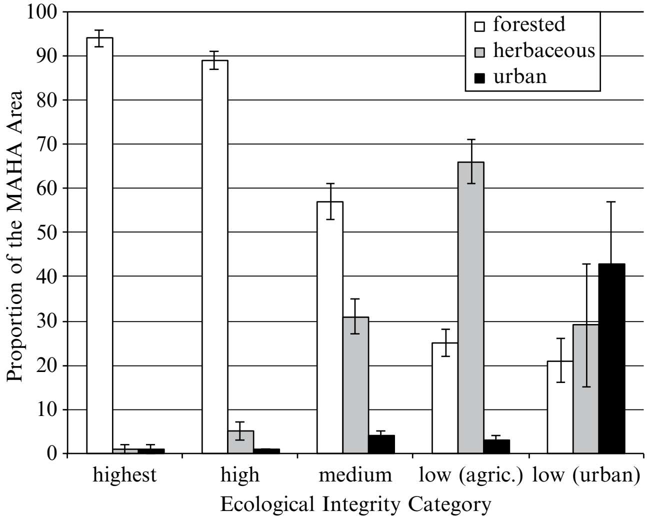

Fig. 8.12 Mean (±SE) percentage of primary land cover at sites in five categories of ecological integrity identified with the Bird Community Index

the loss of forest cover was an important predictor for the other categories. Through regression analysis, we found that the loss of forest as the local land cover matrix was the single most important feature among all tested in predicting overall BCI score (Fig. 8.11). Low-integrity sites, whether dominated by agricultural or urban cover types, were characterized by landscapes in which non-native cover types replaced forest as the matrix. Sites in the medium-integrity category illustrated the vital transition between a forested and non-forested matrix.

Significant differences occurred in many landscape and vegetation variables at the site scale among the five bird-based categories of biotic integrity (Fig. 8.12).

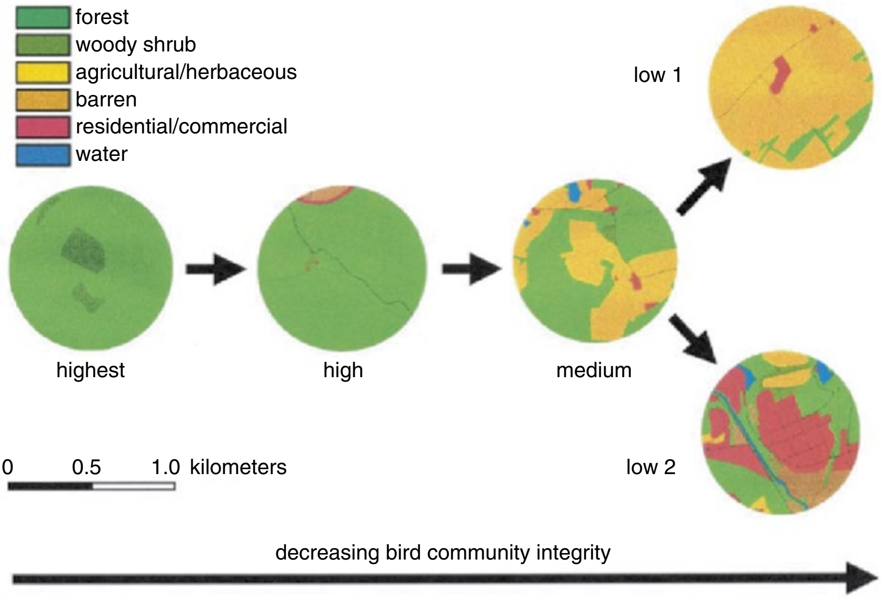

Fig. 8.13 Typical land cover composition at sites placed in five categories of ecological condition using the Bird Community Index; adapted from O'Connell et al. (2000)

For example, the urban ("low 2") sites contained significantly higher percentages of developed cover than sites in any other category ($F_{2,125}=35.91$, $P<0.001$). The agricultural sites ("low 1") contained significantly more herbaceous cover than sites in any other category ($F_{4,125}=40.16$, $P<0.001$). Medium-integrity sites exhibited roughly equivalent proportions of forested and non-forested cover, and differed in this regard from sites in all other categories ($F_{4,125}=63.38$, $P<0.001$). Sites in the high- and highest integrity categories could be separated by any of the landscape patch variables interpreted from aerial photographs (all pairwise Tukey 95% C. I.s included zero). The high- and highest integrity categories, which differed significantly according to BCI, could only be separated by plot level vegetation variables. The highest integrity sites contained significantly higher mean canopy height (Tukey 95% C. I. = −8.030, −0.254) and greater mean canopy closure (Tukey 95% C. I. = −0.265, −0.025). We interpreted this finding to be associated with older stands in the highest integrity sites. A schematic representation of BCI-determined landscapes is illustrated in Fig. 8.13.

8.3.3 A Bird Community Index for the Piedmont and Coastal Plain

As a regional index of biotic integrity for general ecological assessment, the BCI is limited in its application until similar models are developed for additional ecoregions.

In the MAR, the BCI developed for the Mid-Atlantic Highlands is not appropriate for application to the Piedmont and Coastal Plain ecoregions that comprise the eastern portions of the Mid-Atlantic States. This is because major differences in landforms and climate between the regions contribute to differences in native land cover and species composition. For example, extensive areas of the Mid-Atlantic Coastal Plain supported fire-maintained oak and pine savannas prior to settlement by Europeans. Although limited in distribution today due to habitat loss and degradation, native pine savanna occupied a significant proportion of the Mid-Atlantic Coastal Plain, and this vegetative community is not a significant component of the Mid-Atlantic Highlands. In addition to primary vegetation, bird communities differ between the two ecoregions. In the MAR, at least 22 species of small land birds, including nine species of warblers, breed in the Highlands, but not in the Piedmont/Coastal Plain area. Approximately six species breed in the Piedmont/Coastal Plain but not in the Highlands. Thus, due to differences in potential vegetation and potential breeding bird fauna, we concluded that the original "Appalachian" BCI was inappropriate in the Piedmont/Coastal Plain area, and development of a new BCI for the Piedmont/Coastal Plain was warranted. In 2001, we began research to develop this new BCI (O'Connell et al. 2003a, b).

We conducted fieldwork within upland environments of the Mid-Atlantic Piedmont and Coastal Plain physiographic provinces of Pennsylvania, Maryland, Delaware, Virginia, North Carolina, and the District of Columbia. This region contains examples of habitats, natural disturbance regimes, environmental stressors, and potential avifauna representative of the entire Piedmont/Coastal Plain area. We developed the BCI with data from 81 sites in this region intended to reflect a gradient of ecological condition from near pristine to severely degraded (Fig. 8.14). Because these sites were selected to represent the gradient of condition rather than from the EMAP probability sampling grid, this study allowed us to develop and describe the new Piedmont/Coastal Plain BCI, but not to perform an ecological assessment for the region.

Like development of the BCI for the Highlands, we conducted point counts for breeding birds at the sample sites, collected data on plot and stand level vegetation, and characterized land cover in a buffer zone around each site using remotely sensed data in a GIS. We grouped sites into categories based on the composition of 18 life history guilds and identified five distinct bird community types that, when ranked according to "specialist" vs. "generalist" guilds indicated four distinct states of ecological condition among the sites.

Analysis indicated that nine guilds conferring information on structural, functional, and compositional attributes were appropriate for the Piedmont and Coastal Plain (Table 8.8). Calculation of the BCI is a simple summary function of three main variables that represent average ranks in each guild:

$$V1 = \sum \text{Structural Guild Ranks} / 4$$

$$V2 = \sum \text{Functional Guild Ranks} / 4$$

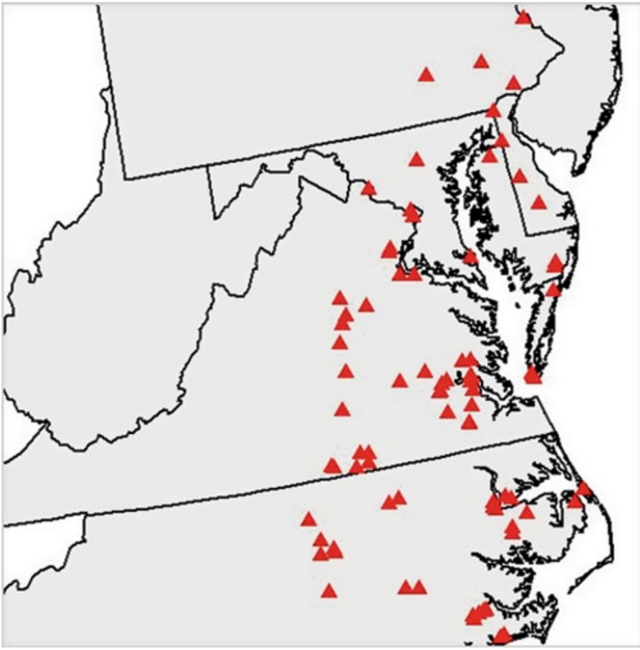

Fig. 8.14 Approximate locations for 81 sites sampled for the development of a Bird Community Index for the Mid-Atlantic Piedmont and Coastal Plain

Table 8.8 Nine response guilds included in the Bird Community Index for the Mid-Atlantic Piedmont and Coastal Plain

	Rank			
Structural guilds	1	2	3	4
Forest interior	0–10.0	10.1–20.0	20.1–28.0	28.1–100
Pine-associated	0	0.1–2.0	2.1–5.0	5.1–100
Urban/suburban	60.1–100	47.1–60.0	20.1–47.0	0–20.0
Functional guilds				
Bark prober	0–9.0	9.1–16.0	16.1–20.0	20.1–100
Upper canopy gleaner	0–4.0	4.1–12.0	12.1–18.0	18.1–100
Ground gleaner	0	0.1–3.0	3.1–7.0	7.1–100
Compositional guilds				
Single-brooded	0–16.0	16.1–34.0	34.1–46.0	46.1–100
Nest disrupter	23.1–100	16.1–23.0	0.1–16.0	0
Exotic	11.1–100	1.1–11.0	0.1–1.0	0

Within cells, values and ranges indicate the percentage of species in a community belonging to each response guild; column headings indicate the rank for those respective values used in calculation of the BCI

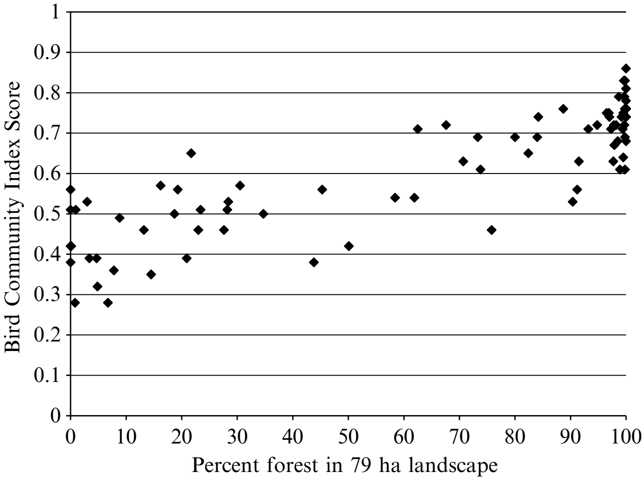

Fig. 8.15 As in the Mid-Atlantic Highlands, Bird Community Index scores increased with increased forested cover in the local landscape of bird sampling sites

$$V2 = \sum \text{Compositional Guild Ranks} / 4$$

$$\text{BCI score} = \sum (V1 - V3) / 9$$

As in the Highlands, response guilds illustrated dramatic changes along the gradient of ecological condition. For example, omnivores outnumbered all insectivores in degraded landscapes of the Piedmont and Coastal Plain. Forest cover, whether in mature broadleaf stands or in pine-dominated savannas, was again an important predictor of ecological condition for the region (Figs. 8.15 and 8.16). Representation of specialist response guilds was low in landscapes in which the local matrix had been converted from forested to either urban or agricultural land uses. Thus, although the bird species, landforms, climate, and response guilds used to build the index were all different in the Piedmont and Coastal Plain that applied in the Mid-Atlantic Highlands, the basic pattern of a forested land cover matrix associated with the distribution of ecological integrity was repeated.

8.4 Regional Index of Biotic Integrity for Forested Riparian Ecosystems

While working to expand the BCI to different ecoregions, Riparia also pursued more specific applications of the concept to finer scales within the Mid-Atlantic Highlands. Based on a conceptual Regional Index of Biotic Integrity for Forested Riparian Ecosystems (Brooks et al. 1998), and in collaborative research with the Carnegie

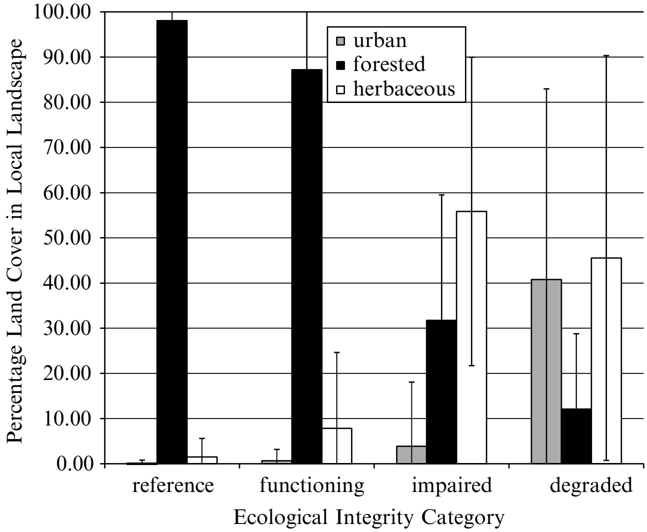

Fig. 8.16 Primary land cover (mean ± SE) percentage in four categories of ecological condition delineated with the Bird Community Index in the Mid-Atlantic Piedmont/Coastal Plain

Museum of Natural History's Powdermill Nature Reserve and East Stroudsburg University, we embarked in 1998 on a study of riparian headwater systems (detailed methods and results can be found in O'Connell et al. 2003a, b). Our objective was to develop a user-friendly and broadly applicable indicator of ecological condition specifically geared toward assessments of the ecological integrity of first and second order streams, and their confining watersheds. The research blended remotely sensed information on land cover within watersheds, an analysis of general bird communities with the BCI, physical characterizations of instream structure, water quality, an intensive characterization of benthic macroinvertebrates, and multiyear monitoring of reproductive success of the obligate riparian songbird, Louisiana waterthrush (*Parkesia motacilla*; hereafter "waterthrush" or "LOWA," Fig. 8.17).

Forested headwater streams, and their associated riparian wetlands, represent the reference condition for ecological integrity for this ecosystem type throughout this region. Headwater streams (first and second order) contribute 60–75% of the total stream length and total drainage area of watersheds in the Mid-Atlantic States. The ecological integrity of headwaters is important to the region (e.g., Sweeney 1992), but they are significantly impacted by a variety of environmental stressors (Sweeney et al. 2004; Brooks et al. 2009).

Maintaining and restoring the ecological integrity of forested riparian buffers have been identified as important strategies to protect the water quality and living resources of the Chesapeake Bay/Susquehanna Basin. The effects of forest buffer width on biotic communities had been studied (e.g., Brooks et al. 1991; Croonquist and Brooks 1991, 1993), but the implications for maintaining biological integrity are uncertain.

Fig. 8.17 The only riparian obligate songbird in the Mid-Atlantic Region, the Louisiana waterthrush (*Parkesia motacilla*) (photo by T. O'Connell)

During a previous series of studies, Brooks et al. (1996) and Miller et al. (1997) developed and evaluated tools for assessing cumulative impacts on wetlands and associated streams and riparian areas by characterizing their current structure, potential functions, and restoration potential in a watershed context. This research focused primarily on wetlands and riparian areas associated with streams equal to or lower than third order, or headwaters. Individually, headwater streams and wetlands are smaller in scope than the more expansive areas of forested floodplains found downstream, but they assume a relatively more important role in maintaining instream water quality, because proportional to size, more overland flow passes through these low order riparian wetlands than through bottomland forests (Brinson 1993). In most watersheds, there are more headwater streams, with a larger cumulative length, than mainstem rivers (Leopold 1974, see Chap. 1).

To construct the indicator, field sampling in headwater streams and riparian habitats occurred over three ecoregions in the MAIA at different scales (e.g., 4.5 ha/territory, 25 ha/reach, 250 ha/watershed, and 2,500 ha/landscape). Each bioindicator is most strongly associated with measures of habitat at a particular scale. Measuring waterthrush productivity related primarily to quality of riparian habitat, but it was also dependent on the availability of macroinvertebrates as food. Biomass and composition of macroinvertebrate communities related to instream and wetland habitat and measures of water chemistry and sedimentation. Avian communities related primarily to landscape metrics. Attributes of the waterthrush productivity spanned the widest range of scale among the indicators.

We collected data from 23 stream reaches in three distinct study areas in Pennsylvania. Six eastern study streams were contained within the Delaware River Watershed, seven central Pennsylvania streams were located within the Susquehanna

River watershed, and ten streams in western Pennsylvania contributed to the greater Ohio River watershed. The separate study areas typified vegetation and terrain of the glaciated Poconos Plateau (east), Ridge and Valley (central), and Allegheny Plateau (western) physiographic provinces in Pennsylvania. Individual study stream reaches were first or second order perennial segments that were 2–3 km in length.

For this study, we included forest fragmentation as a stressor under observation, but selected "fragmented" study reaches that were only mildly disturbed. We selected fragmented study reaches that still supported breeding waterthrush rather than deforested reaches where there would be no waterthrush at all. Thus, our objective was to more clearly define a threshold of forest cover at which waterthrush breeding success was compromised, as evidence for degradation of headwater stream ecological integrity. A decline in reproductive success in forest fragments is a common mechanism cited for the pattern of area sensitivity in songbirds that breed in Nearctic temperate forests (e.g., Hoover and Brittingham 1998; Rodewald 2002).

Tetra Tech Inc., in cooperation with the West Virginia Department of Environmental Protection, developed the Stream Condition Index (SCI) for US EPA Region 3, specifically for West Virginia wadeable streams. The SCI is a reference condition bioassessment approach that compares a stream's biological condition to that of unimpaired streams of the same region (Gerritsen et al. 2000). The index is comprised of six discriminatory metrics representing three different categories of benthic community attributes: taxonomic richness (counts of distinct taxa within selected taxonomic groups), taxonomic composition (proportions of individuals in selected taxonomic groups), and tolerance to environmental stress (Gerritsen et al. 2000).

We applied the SCI to macroinvertebrate data collected at our study sites and found associations with N concentration, pH, and the suite of instream and riparian corridor condition stressors (e.g., sedimentation) indicated by the USEPA's existing Stream Habitat Assessment (SHA). For N, we had an insufficient gradient to consider eutrophication as a stressor. The SCI exhibited a statistically significant relationship to pH. Our results suggested three categories of indicator coding in response to acidification. For pH above 6.5, we found no evidence of degradation to the benthic macroinvertebrate community. We documented some level of degradation in a range of pH between 6.5 and 5.5. According to our data, reaches supporting pH less than 5.5 will support a degraded benthic macroinvertebrate community.

Benthic macroinvertebrates also revealed statistically significant shifts in SHA scores. Our data suggested that reaches with SHA scores above approximately 165 (out of a maximum possible 200 points) exhibited no degradation of benthic macoinvertebrate communities discernible with the SCI. Some degradation was possible in a range from about 150 to 165. Recognizable degradation occurred on reaches that scored between about 125 and 150. Our data suggest that SHA scores below 125 indicated a degraded community of benthic macroinvertebrates.

The BCI applied to data on the community of breeding songbirds and near-passerines confirmed a statistical relationship between landscape level forest cover and ecological integrity. The BCI discriminates four categories of ecological integrity in the Mid-Atlantic Highlands when applied to songbird community data; three categories are identifiable with data on forest cover in the local landscape.

We demonstrated relationships between aspects of waterthrush breeding biology and stressors affecting both instream and landscape condition. As with data from the SCI, our results indicated that waterthrush populations may begin to experience detrimental effects in a range of pH from roughly 5.5 to 6.5.

We found waterthrush variables to be related as well to SHA scores. In this case, we documented declines in waterthrush territory density and reproductive success at roughly where SHA scores fell below 150. Thus, our data from waterthrush point to three categories of condition in SHA scores: no degradation (150–200), possible degradation below 150, and probably to roughly 110, and degraded condition below 110.

With respect to landscape level forest cover, if we base the threshold for waterthrush suitability on territories/km, we found that only sites with at least 70% forest supported two or more waterthrush territories/km. Our data suggested that waterthrush populations become somewhat degraded where forest cover falls below 70%, and previous work indictated that waterthrush would not occur at sites with less than 40%. Thus, less than 40% forest cover with the riparian corridor and adjacent uplands is another logical threshold break for waterthrush. We concluded that the occurrence of waterthrush integrated across taxonomic groups and stressors so well, that we had little justification in calibrating to adjust assessment thresholds established by the other macroinvertebrate and songbird indices.

Our data showed that information as simple as the richness of a suite of riparian-associated songbirds can provide a linkage between instream and landscape level condition. We found significant differences in riparian songbird richness at roughly the same threshold values for SHA as determined with the SCI. Thus, the riparian songbird community, like LOWA, integrated macroinvertebrate data and provided a calibration to the stressor of stream habitat degradation.

In addition, the riparian songbird community was significantly associated with percent forest cover at local landscape scales. Our data suggested that the greatest richness of the riparian community occurs where forest cover approaches 100%. This result indicated that the riparian songbird community, like waterthrush, also integrates with the larger songbird community and calibrates the stressor of forest fragmentation.

Thus, after testing the Regional Index of Biotic Integrity for Forested Riparian Ecosystems (RIBI), first proposed by Brooks et al. (1998), we found that the RIBI integrated stressors across spatial scales, and across physical, chemical, and biological measures important to the integrity of headwater systems in the MAR. Detailed methods and results are available in O'Connell et al. (2003a, b, found at http://www.riparia.psu.edu/MARbook).

8.5 Ecological Assessments with the BCI: Practical Applications

8.5.1 Riparian Assessments with the BCI

When applied to a large number of sites drawn from a random sample of an ecoregion, the BCI can provide an ecological assessment of the region in the form of proportions of sample sites in different categories of ecological condition. The BCI is not intended to provide a robust indication of the ecological condition at any one, fine-scaled sampling location. At fine scales, additional information is often available that would be important in determining the conservation value of the site but would be poorly reflected in the location's BCI score. For example, the unique plant and insect community that could develop on a granite flatrock could make the site a regional priority regardless of the BCI one might calculate for the site. At a fine scale, the BCI should be regarded as a hypothesis of ecological condition rather than a determination of the site's ecological condition.

Despite caveats regarding the subtle nature of BCI application, potential users of the index are often tempted to apply it to fine scales. Thus there is great interest in examining the performance of the BCI to indicate ecological condition at sites where ancillary information is available. In 2004, Riparia conducted a study that involved rapid assessment via the BCI at sites in the MAR in which additional data on assessment were available (Gyekis 2007).

Based on previous results, it was expected that BCI scores would correlate with desirable riparian habitat characteristics measured in various other studies, including the Riparian Invertebrate Community Index (RICI) scores (Laubscher 2005) and SHA scores based upon the Environmental Protection Agency's Rapid Bioassessment Protocol for Wadeable Streams and Rivers (Barbour et al. 1999). The study examined how BCI scores and bird species richness related to various measures of physical and biological habitat quality in riparian habitats of the Mid-Atlantic Highlands.

Fieldwork for the study included point counts for breeding songbirds in riparian zones where additional assessment metrics were also applied. The sample was non-random and included 197 point count locations in Pennsylvania and West Virginia, with most of the points in five watersheds in central Pennsylvania. BCI scores were calculated from 81 songbird species encountered at these sites.

Correlation between a macroinvertebrate-based index of biotic integrity (the RICI) and BCI scores at 37 sites distributed throughout central and southeastern Pennsylvania were found to be highly significant. Final scores from the RICI were divided into three sections, one for a standard stream analysis, a floodplain analysis, and a combined score. The BCI score was significantly correlated with the standard stream, the floodplain, and the combined metrics at these 37 sites (Fig. 8.18).

We also found that the BCI correlated with an independent assessment of physical attributes of the riparian zone. Both SHA and BCI data were available for five Pennsylvania watersheds. A quadratic regression best predicted BCI score from

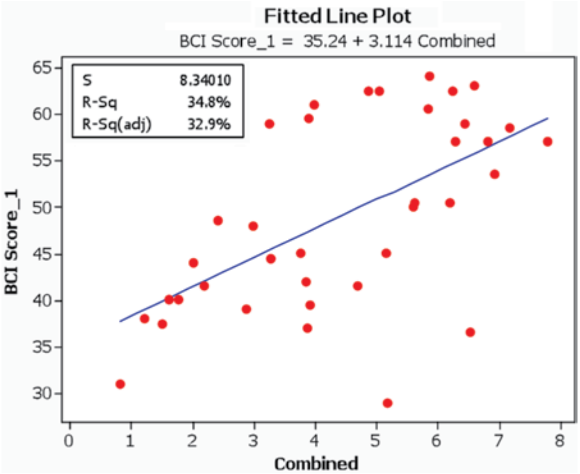

Fig. 8.18 Statistical relationship between Bird Community Index and the Riparian Invertebrate Community Index (Gyekis 2007)

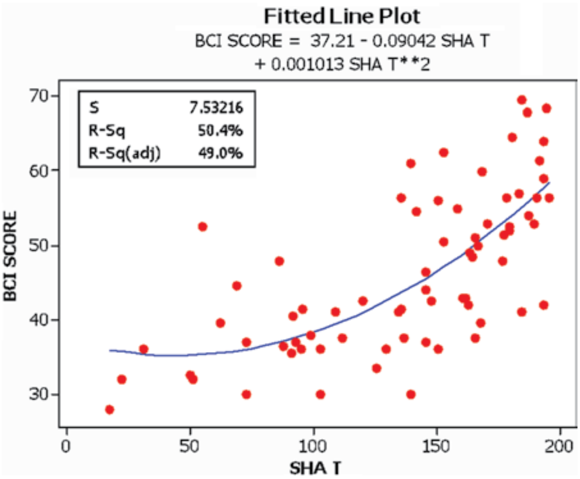

Fig. 8.19 Regression equation predicting Bird Community Index score from Stream Habitat Assessment score (Gyekis 2007)

SHA and was both highly significant and exhibited a strong positive correlation of $r = 0.68$ (Fig. 8.19). In contrast, simple songbird richness was negatively correlated with SHA ($r = -0.27$).

8.5.2 *Independent Uses of the BCI and Future Directions*

Since publication of the BCI and its use in ecological assessment of the MAR several independent efforts have expanded the concept and its application (Canaan Valley Institute 2002). Allen et al. (2004) used the BCI applied to Breeding Bird

Survey data from New Hampshire as one component of a statewide assessment of human health and ecological risk. In New York, Glennon and Porter (2005) applied the BCI to data from the Adirondack Park collected during the first breeding bird atlas. They found that the areas supporting highest integrity were distant from roads and often characterized by small wetlands. As with our work in the MAR, Glennon and Porter (2005) found that sites supporting the highest BCI scores had low species richness compared to other sites in the region. The National Park Service uses both the BCI and a modified Headwater Stream Assessment for condition monitoring in the Eastern Rivers and Mountains Network (Marshall and Piekielek 2007). In an application to a radically different landscape, Coppedge et al. (2006) developed a Grassland Disturbance Index modeled on the BCI for application to grasslands in western Oklahoma. In Riparia, we have provided advice to researchers who are interested in developing bird-based indicators in Ecuador, Thailand, India, Nepal, and other countries as well.

We continue to work on expansion and modification of the BCI concept to meet challenges of ecological assessment at broad scales (O'Connell 2009). Anthropogenic disturbances from land cover change, invasive species, energy exploration and development, and climate change point to increasingly dynamic landscapes in North America in the future. Conservationists will be increasingly called upon to delineate those areas of refuge for native species and communities, to determine landscapes capable of absorbing development without substantial loss of ecological integrity, and monitor condition over broad areas.

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Chapter 9
Assessing Wetland-Riparian Amphibian and Reptile Communities of the Mid-Atlantic Region

James T. Julian, Gianluca Rocco, Melinda M. Turner, and Robert P. Brooks

Abstract The dependency of many amphibian and reptile species on aquatic habitats is well known. Here, we summarize four studies that investigated aspects of herptile life histories and developed models and tools to assess their responses to disturbance and changing environmental conditions. An Amphibian Index of Biological Integrity (AIBI) was developed and tested for amphibian communities found in headwaters of the Ridge and Valley ecoregion of Pennsylvania. The AIBI demonstrated how amphibian species are significantly and negatively affected by changes in land use, and how conserving an intact wetland-riparian corridor is extremely important for maintaining amphibian biodiversity. A study of pond-breeding assemblages of amphibians in the Delaware Water Gap National Recreation Area demonstrated their response to a hydrologic gradient of connectivity. The degree of pond isolation, as defined by hydrologic connectivity, land use, and predator access, significantly impacted these assemblages, and thus, can be used as predictors of amphibian species occurrence. This study confirmed the importance of protecting isolated wetlands in the landscape. A third study investigated the response of the stream-dwelling plethodontid salamanders to acidified conditions caused by atmospheric deposition and acid mine drainage in western Pennsylvania. This study revealed that stream plethodontid abundance, presence, and diversity were

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severely suppressed in acidified environments. The value of stream salamanders as a bioindicator was confirmed by this and a subsequent study of similar assemblages throughout the Mid-Atlantic Highlands. The final study involved development of a Habitat Conservation Plan (HCP) for the federally threatened bog turtle, a wetland-dependent reptile with a stronghold in southeastern Pennsylvania and northwestern Delaware. Teams of investigators from multiple organizations assessed ecological, legal, socioeconomic, and land management factors to arrive at a recommended HCP. A process to locate and operate conservation banks in prime recovery areas was established. Through a system of credit generation, critically important habitats for breeding colonies of bog turtle would be protected.

9.1 Introduction

Many amphibian and reptile species of the Mid-Atlantic Region (MAR) are dependent on aquatic habitats during some part of their life cycle. In this chapter, we summarize four studies that investigated aspects of herptile life histories for the purpose of developing assessment models and tools regarding their responses to disturbance and changing environmental conditions. In the first study, we used a portion of Riparia's set of reference wetlands (http://www.riparia.psu.edu/MARbook, see Chap. 2) to develop and test an Amphibian Index of Biological Integrity (AIBI) for headwaters. Next, we review a study of pond-breeding amphibians to demonstrate their response to a hydrologic gradient of connectivity and disturbance. We also examined the utility of using stream-dwelling salamanders as bioindicators of acidified conditions. Finally, we review the development of a Habitat Conservation Plan (HCP) for the federally threatened bog turtle, for which we assessed ecological, legal, socioeconomic, and land management factors. Thus, the intent of this chapter is to summarize several investigations conducted by the faculty and students of Riparia at Penn State. We do not provide detailed descriptions of their taxonomy or natural history, because wetland-riparian-dependent amphibians and reptiles of the MAR are covered in an array of existing publications (e.g., McCoy 1989; Shaffer 1991; Hulse et al. 2001; Tiner 2005).

9.2 Disturbance Gradients and Amphibian Index of Biological Integrity

Anthropogenic disturbances surrounding a wetland impose a strong influence on the composition of its amphibian assemblage because upland habitats serve as important foraging, overwintering, and migration habitat for some amphibian species. In this regard, the most detrimental landscape disturbance in the MAR is the conversion of forested habitats to agricultural and urbanized landscapes. Studies we have

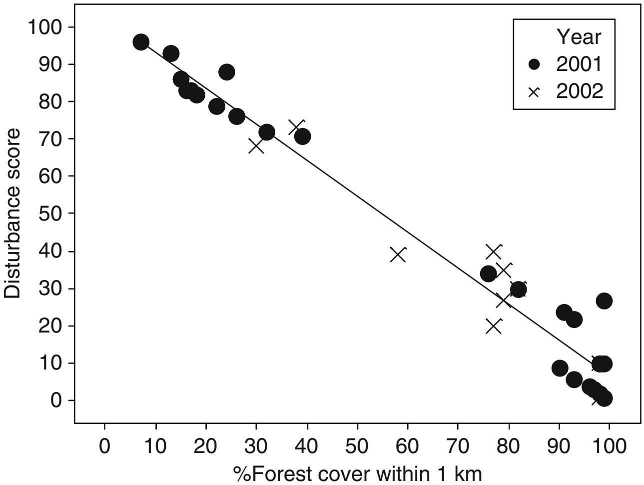

Fig. 9.1 Disturbance score vs. percent forest for 37 headwater wetlands in central Pennsylvania

conducted in the Ridge and Valley region of central Pennsylvania have shown that amphibian species richness decreases as anthropogenic disturbance increases, and several amphibian species common to relatively undisturbed wetlands are rarely found at more disturbed sites. This work has allowed us to develop an AIBI that consistently relates functional assessments of wetland disturbance to amphibian species richness and composition.

9.2.1 Study Area, GIS Analysis, and Developing an Amphibian Index of Biotic Integrity

In 2001 and 2002, we quantified wetland disturbance metrics at headwater-complex wetlands in the Ridge and Valley region of central Pennsylvania, and sampled their amphibian communities ($n=27$ in 2001; $n=10$ in 2002) (Farr 2003). Disturbance scores were calculated according to Brooks et al. (2004) by quantifying land use within 1 km of the wetland center, and by characterizing on-site and riparian buffer characteristics that could compromise the wetland's ecological functions (e.g., "stressors"). Disturbance scores were reported on a scale of 0 (no detectable disturbance or degradation) to 100 (severely disturbed/degraded). The most influential variable in quantifying disturbance scores was the percent of forest cover present within 1 km of a wetland (Fig. 9.1), which was strongly correlated with disturbance scores ($n=27$, $r=-0.979$, $p<0.001$). These wetlands were placed into two disturbance categories; "reference wetlands" with disturbance scores ≤40 and 99–58% forest cover ($n=24$), and "disturbed wetlands" with disturbance scores ≥68 and 39–7% forest cover ($n=13$).

Table 9.1 Amphibian life history and disturbance tolerance categories for species encountered in headwater complex wetlands in the Ridge and Valley Province of Pennsylvania

Life history	Species	Disturbance tolerance
Vernal pool obligate	Wood frog *Rana*[a] *sylvatica*	Intolerant
	Spotted salamander *Ambystoma maculatum* Shaw	Intolerant
	Jefferson salamander *Ambystoma jeffersonianum* Green	Intolerant
Pond-breeding (non-vernal obligate)	Pickerel frog *Rana palustris* Le Conte	Intermediate
	Red-spotted newt *Notophthalmus viridescens viridescens* Rafinesque	Tolerant
	N. Spring peeper *Hyla crucifer crucifer* Wied	Tolerant
	American toad *Bufo americanus americanus* Holbrook	Tolerant
	Green frog *Rana clamitans melanota* Latreille	Tolerant
Woodland salamanders	Redback salamander *Plethodon cinereus* Green	Intolerant
	Slimy salamander *Plethodon glutinosus glutinosus* Green	Intolerant
Streamside salamanders	N. Dusky salamander *Desmognathus fuscus fuscus* Rafinesque	Intolerant
	N. Red salamander *Pseudotriton ruber ruber* Latreille	Intolerant
	Longtail salamander *Eurycea longicauda longicauda* Green	Intolerant
	N. Spring salamander *Gyrinophilus porphyriticus porphyriticus* Green	Intolerant
	Mountain Dusky salamander *Desmognathus ochrophaeus* Cope	Intolerant
	Two-lined salamander *Eurycea bislineata bislineata* Green	Intermediate
Habitat specialist	Four-toed salamander *Hemidactylium scutatum* Temmick and Schlegel	Intolerant

Tolerance categories are based on literature reviews (Klemens 1993; Petranka 1998) and personal observations

[a]The genus *Rana* has been proposed for change to *Lithobates* (accepted by Crother (2008)), but has not been universally accepted (e.g., Pauly et al. 2009). In this chapter we use *Rana*

Amphibian species occurrences were used to calculate an Amphibian Index of Biotic Integrity (AIBI) for each wetland. Amphibian communities were sampled using diurnal visual encounter surveys, dipnet surveys, and nocturnal call surveys. During surveys all individuals encountered were characterized by their species and life stage (egg mass, larvae, adult, or call), and enumerated. Amphibian species were characterized a priori into three categories that described their tolerance to wetland disturbance (Table 9.1). Intolerant species were those most closely associated with wetland habitats that retain a high percent of intact forest surrounding them, while tolerant species can often be found in wetlands surrounded by very little upland forest. The AIBI was calculated based on five, equally weighted metrics: (1) amphibian species richness, (2) the number of intolerant species found at a site,

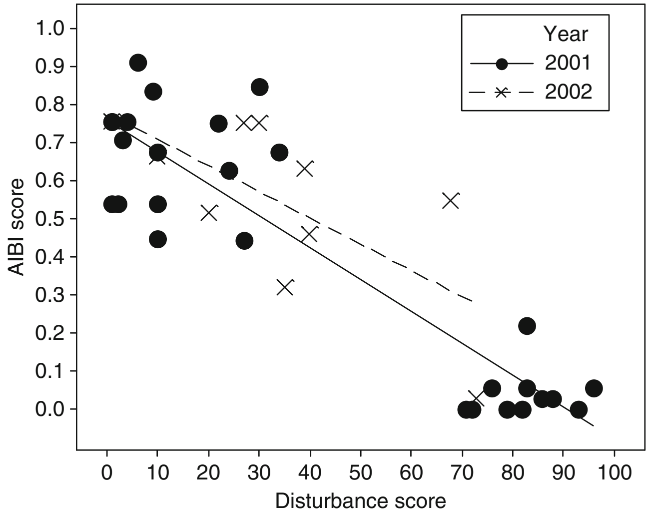

Fig. 9.2 Amphibian Index of Biological Integrity scores vs. disturbance scores

(3) the percent of intolerant amphibian species at a site, (4) the presence of intolerant pond-breeding species (either wood frog or spotted salamanders), and (5) the presence of the Northern dusky salamander (an intolerant stream-breeding species).

9.2.2 AIBI Scores Across Disturbance Gradients

Amphibian communities at disturbed wetlands were less diverse than reference wetlands, and rarely contained intolerant species. AIBI scores were negatively correlated with disturbance scores in both 2001 ($r = -0.907$, $p < 0.001$) and 2002 ($r = -0.678$, $p = 0.031$) (Fig. 9.2). All five metrics that comprised the AIBI were negatively correlated with disturbance scores in 2001 ($p < 0.001$ for all metrics), although only the presence of northern dusky salamanders was significantly correlated in 2002 wetlands ($p = 0.009$) (Fig. 9.3). More species were found in reference sites (mean = 5.29, SD = 1.85) than disturbed sites (mean = 1.15, SD = 1.21), and intolerant species were more likely to be found at reference sites than disturbed sites. Amphibian communities at nearly all disturbed wetlands consisted of only spring peepers and/or American toads (11 out of 13 sites). As a result, disturbed sites produced very low AIBI scores (mean = 0.0785, SD = 0.1532) compared to reference sites (mean = 0.6518, SD = 0.1480), and only one disturbed site had an AIBI score that was as high as AIBI scores from reference sites.

Reference wetlands were diverse in the number of species they contained, and the life history strategies of their amphibian communities (Table 9.2). Reference wetlands typically contained five or more amphibian species, and always contained

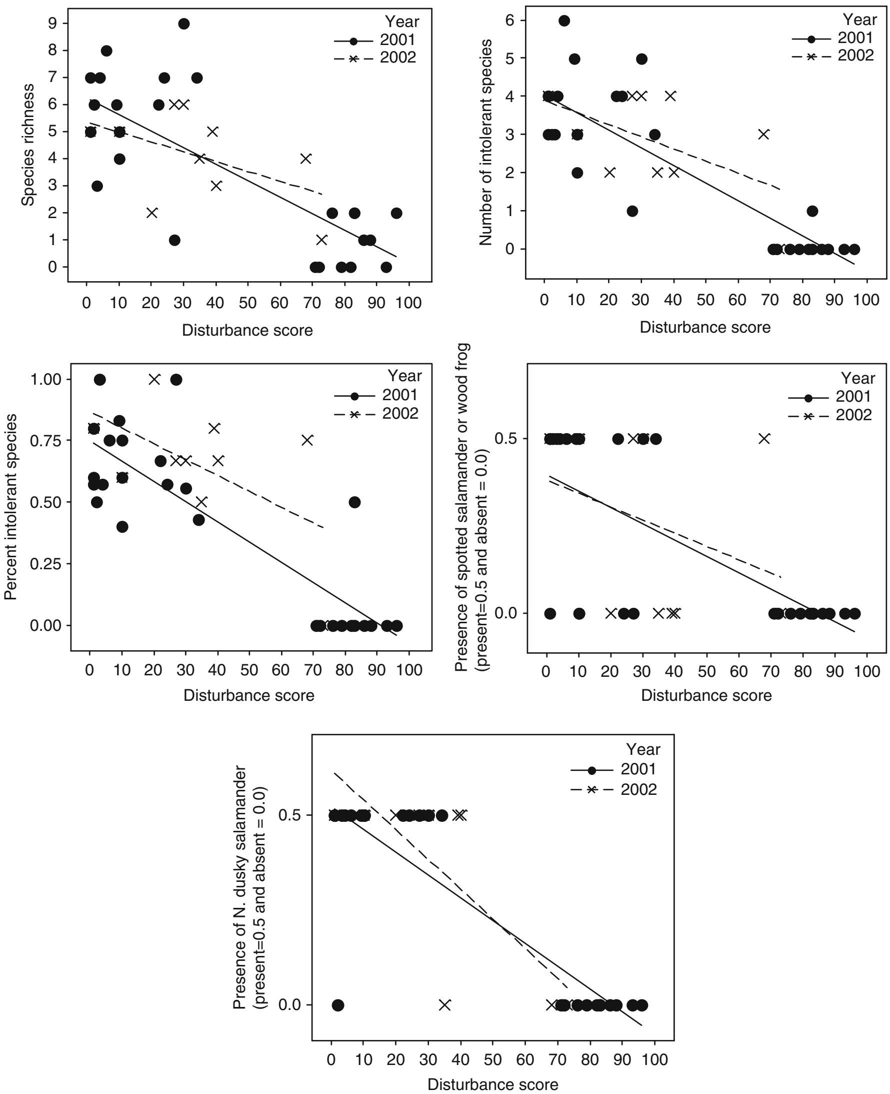

Fig. 9.3 Individual metrics used to calculate AIBI vs. disturbance scores from headwater wetlands sampled in 2001 ($n = 24$) and 2002 ($n = 13$)

at least one or more intolerant species. A larger fraction of reference wetlands contained vernal pool obligate species, streamside salamanders, and four-toed salamanders (a habitat specialist of sphagnum bogs) than disturbed wetlands. In addition, reference wetlands were the only sites where woodland salamanders were found. While 75% of reference wetlands contained species from three or more life history strategies, only one disturbed site contained species with more than one life history strategy.

Table 9.2 The percent of reference wetlands ($n=24$) and disturbed wetlands ($n=13$) that contained at least one amphibian species from given life history strategies

Life history strategy	Reference wetlands ($n=24$) (%)	Disturbed wetlands ($n=13$) (%)
Vernal pool obligate	62.5	7.7
Pond-breeding (non-vernal pool obligate)	66.7	61.5
Woodland salamander	37.5	0.0
Streamside salamander	95.8	7.7
Habitat specialist	29.2	7.7

The loss of forested area and water-retaining microhabitats are disturbances that most likely prevent human-altered wetlands from containing diverse amphibian communities. In our study area, typical wetland disturbances were due to the expansion of agriculture and involved the clearing of trees and woody debris, as well as the draining/regrading of water-retaining depressions. Among wetlands sampled in 2001, there was significantly higher canopy cover and more downed wood debris within reference wetlands than disturbed wetlands. These habitat elements are important in retaining moist microclimates for all species of amphibians, notably, the woodland salamanders that lay their eggs under cover objects on land and rely entirely on cutaneous respiration for gas exchange. The lack of forested landscape surrounding disturbed wetlands likely impacted vernal pool amphibians the greatest because these habitats are important foraging and hibernation grounds, and serve as moisture-retaining migration corridors to breeding pools located within wetlands. Reference wetlands contained significantly more microhabitats than disturbed sites, and microhabitats often retained water for longer periods of time in reference wetlands. Depressions that form ephemeral water-bodies are microhabitats that serve as breeding habitats for vernal pool amphibians. These small habitats are easily filled-in or drained by regrading, and evidence of these activities was noted in disturbed wetlands.

It is possible that diverse amphibian communities may remain unchanged in the face of some human disturbances, but quickly shift towards communities dominated by pond-breeding, tolerant species after a critical threshold of disturbance occurs. We propose this because there was no significant relationship between AIBI scores and disturbance levels among reference wetlands only ($r=-0.298$, $p=0.158$, $n=24$), nor did we find such relationships among disturbed wetlands only ($r=-0.357$, $p=0.231$, $n=13$). While a linear function adequately models the relationship between AIBI score and disturbance level among all wetlands (R^2 adjusted$=76.0\%$, $p<0.001$, $F=115.03$, DF$=1$), a threshold response modeled by a third-order polynomial appears to fit these data slightly better (R^2 adjusted$=78.7\%$, $p<0.001$, $F=45.36$, DF$=3$) (Fig. 9.4). This function suggests that amphibian communities can change rapidly at intermediate disturbance scores between 30 and 60. At these levels of disturbance, amphibian communities rapidly transform amphibian assemblages that breed on land, in ponds, and in streams, into depauperate assemblages consisting of disturbance-tolerant, pond-breeding anurans.

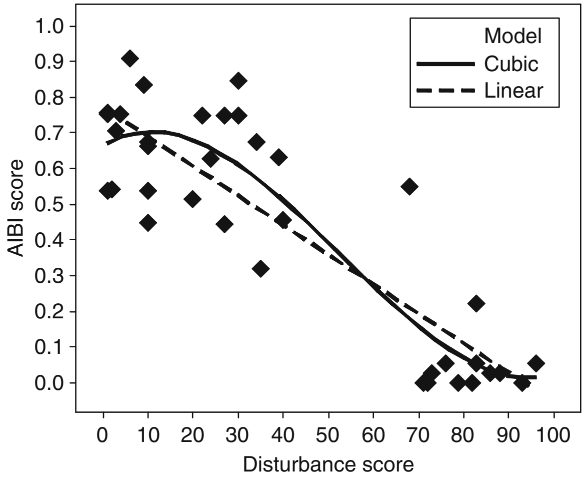

Fig. 9.4 Comparison of a modeled linear response (*linear*) vs. a threshold response (*cubic*) for AIBI across a disturbance gradient for 37 headwater wetlands

We cannot decisively rule out either linear or threshold functions to model AIBI scores across disturbance gradients because of a lack of data from intermediate disturbance levels. In fact, intermediate disturbance levels in headwater wetlands may be rare since land often is converted to agriculture in an all-or-nothing manner. Wetlands with intermediate disturbance levels may, however, be more common in urbanizing landscapes with low-density housing than in the agricultural areas used in our studies. In urbanizing landscapes, scientists have begun to examine how to conserve amphibian populations by retaining narrow, forested corridors that link vernal pools to larger patches of forested upland habitat. Incorporating wetlands from these landscapes into manipulative or retrospective studies could determine if amphibian communities display either a linear or a threshold response along disturbance gradients. Such studies could be used to direct residential planning that will best maintain amphibian communities at wetlands in urbanizing landscapes.

9.3 Distributions of Pond-Breeding Amphibians Across Stream Connectivity Gradients

Amphibians are the most abundant vertebrate taxa in many lentic wetlands, and our research has found their occurrence is strongly influenced by a wetland's hydrology, surrounding landscape, and its hydrologic connections to adjacent bodies of water (e.g., hydrologic connectivity). Because hydrogeomorphic (HGM) classification explicitly categorizes wetlands using these characteristics, we believe it is the most useful of the major wetland classification taxonomies for predicting amphibian assemblages at a wetland. While the influence of hydrology and landscape elements have long been used as predictors of amphibian occurrence, we have been successful at using hydrologic connectivity as a gradient that predicts species occurrence at

Table 9.3 Amphibian life history characteristics across wetland hydroperiod gradient

		Shorter hydroperiod species	Longer hydroperiod species
Adult life history	Breeding phenology	Late winter/early spring	Late spring through summer
	Breeding season synchronicity	Days to week	Weeks to months
	Migrations and residency time at breeding sites	Upland migrations short residency	No migrations and year-round inhabitants
Larval life history	Growth and activity rates	Rapid growth and high activity rates	Slow growth and low activity rates
	Times to, and size at, metamorphosis	Weeks to months and small size	Months to years and large size
Biotic interactions in aquatic environment	Competitive strength	Good w/o predators present	Poor w/o predators present
	Susceptibility to predators	Highly susceptible	Low susceptible

wetlands in the Delaware Water Gap National Recreation Area (DEWA). Hydrologic connections are associated with important abiotic (wetland hydroperiod) and biotic (predator presence) conditions that shape amphibian assemblages. Therefore, classification systems that do not inherently account for these conditions will not provide insight into likely amphibian assemblages at wetlands. For instance, wetlands with short hydroperiods are only suitable for amphibian species with relatively fast larval development rates (Table 9.3). These species tend to migrate from upland, forested habitats to breeding ponds during a breeding season that will last, at most, several weeks. While these species can (and do) breed in wetlands with permanent hydroperiods, their larvae are highly susceptible to fish and macroinvertebrate predation, and lose their competitive advantage over other amphibian species when in the presence of predators.

9.3.1 Study Area and GIS analysis

Our work evaluating the effect of hydrologic connectivity on amphibian assemblages was conducted in the DEWA located along the Delaware River in northeastern Pennsylvania (DEWA-PA) and northwestern New Jersey (DEWA-NJ) (Julian 2009). This 28,000 ha national park unit consists of mostly forested habitats that include hemlock-dominated headwater gorges, mixed hardwood-terraced benches, and steeply sloping ridge-valley landforms. Previous work in DEWA identified 352 lentic wetlands that could be used by amphibians for breeding (Snyder et al. 2005). We defined these “breeding ponds” as lentic wetlands that typically retained standing water through the month of April (when pond-breeding amphibian species begin to lay eggs) (i.e., depression, with either seasonal or temporary hydrology, Brooks et al. 2011, see Chap. 2). Breeding ponds were identified between 2001 and 2003 by evaluating over 1,000 wetland polygons identified from 1:12,000 aerial photography, and active searches for small, isolated wetlands.

Spatial locations for breeding ponds were organized in a GIS, and were attributed with area estimates of their flooded perimeters. Using a photo-interpreted, digital vegetation map of DEWA (Fike 1999), we quantified the proportion of the area surrounding each wetland that contained at least 50% forest canopy cover within 25 m (P25), and within 250 m (P250) of their flooded perimeter. We also quantified the number of potential breeding wetlands within 1 km of the flooded perimeter of each pond or wetland (Ponds1km).

9.3.2 Wetland Assessments and Developing Species Distribution Models

We evaluated the degree of hydrologic connectivity to nearby bodies of water for all breeding ponds located in the New Jersey portion of DEWA (NJ-DEWA) ($n=175$). Breeding ponds categorized as "strictly isolated" showed no evidence of channelized inflow or outflow of surface water, and would be classified within HGM as temporary depressions. "Seasonally connected" ponds possessed a seasonal hydrologic connection that consisted of surface water inflow or outflow channels that were observed to dry at least once during the year. Within the HGM classification, most of these ponds would be considered seasonal depressions, while the remainder would be classified as riverine headwater complexes. "Permanently connected" ponds had at least one channel that had never been observed to dry, and were mainly a mixture of HGM riverine lower perennial beaver, and human-impounded, wetlands.

Amphibians were surveyed in 44 ponds in New Jersey, and 32 ponds in Pennsylvania, in 2005 and 2006, respectively. A combination of anuran call surveys, visual encounter survey, and larval dipnet surveys were employed to identify eggs, larvae, and adults to the level of species. Each pond was visited biweekly from March through July, with a minimum of five visits per pond. The presence of predatory fish species at ponds was also recorded during amphibian surveys.

For nine species of amphibians, we estimated the probability of a species occurring at a site as a function of pond size, hydrologic connectivity, isolation from other ponds, forest canopy cover, and fish presence. These relationships were estimated using occupancy models (MacKenzie et al. 2002) that account for the imperfect detection of a species by adjusting the probability of occurrence (ψ) by the probability the species was present, but went undetected. We used data from ponds sampled in New Jersey ponds to develop occupancy models (a model training data set). Occupancy models were then used to predict the presence of amphibian species at all 175 breeding ponds in New Jersey. These models were also fit to the 32 Pennsylvania ponds sampled in 2006, and we used presence/absence data to assess the accuracy of the occupancy models (a model validation data set).

Table 9.4 Summary of regression coefficients (β ± 1 SE) for probability of occupancy (ψ) from final occupancy models for 9 amphibian species sampled in the DEWA in 2005

Species	Area[a]	Connect[b]	Area × connect[c]	P250[d]	P25[e]	Ponds 1 km[f]
A. maculatum	1.88 ± 0.92			3.71 ± 1.79		
N. viridescens				7.80 ± 3.45	−7.95 ± 2.86	
Bufo species					−7.87 ± 4.12	
H. versicolor					−11.09 ± 4.99	1.04 ± 0.52
P. crucifer	9.36 ± 1.11	11.94 ± 1.60	−3.15 ± 0.39			
R. catesbeiana		2.05 ± 0.95				
R. clamitans	1.96 ± 0.92	2.34 ± 0.78				
R. palustris	10.85 ± 0.67	13.80 ± 0.94	−3.34 ± 0.20	−8.29 ± 2.86		
R. sylvatica		−2.15 ± 0.79				
Fish presence		1.40 ± 0.56				

[a]Area: wetland area
[b]Connect: strictly isolated = 1, seasonal connection = 2, permanent connection = 3
[c]Area × Connect: interaction term between Size and Connect
[d]P250: proportion of forested area within 250 m of flooded perimeter with >50% canopy cover
[e]P25: proportion of forested area within 25 m of flooded perimeter with >50% canopy cover
[f]Ponds 1 km: number of amphibian breeding wetlands within 1 km of flooded perimeter

9.3.3 Hydrologic Connectivity Gradients to Predict Amphibian Species Occurrence

Hydrologic connectivity was the most common predictor variable among the best-fit occurrence models we generated for species (Table 9.4). We found three different relationships between amphibian occurrence and connectivity among five species of amphibian. Species whose larvae require at least 1 year to complete metamorphosis were most likely to be found in ponds with permanent hydrologic connections. These species included *Rana clamitans* (green frog) and *Rana catesbeiana* (bullfrog) that were ten and eight times, respectively, more likely to breed in ponds with permanent hydrologic connections than ponds with seasonal connections. The vernal pool obligate species *Rana sylvatica* (wood frog) was 8.6 times less likely to be found in ponds with permanent hydrologic connections than any other wetland type. And lastly, in *Pseudacris crucifer* (spring peeper) and *Rana palustris* (pickerel frog), we found that occupancy was positively influenced by wetland size among isolated ponds, but size was less influential in determining occupancy among ponds with seasonal and permanent connections.

Hydrologic connectivity appears to be a useful predictor of amphibian occurrence because it is associated with two major factors that regulate larval survival: pond drying and predation. Among the 171 ponds in DEWA-NJ, we found that isolated ponds were five times less likely to maintain water levels as similarly-sized ponds with sea-

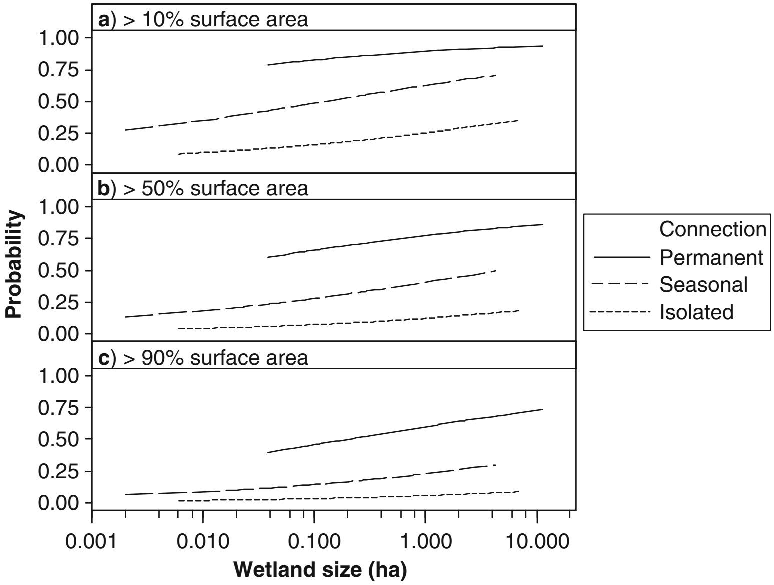

Fig. 9.5 Logistic regression model predictions for the probability a wetland will retain at least (**a**) 10%, (**b**) 50%, or (**c**) 90% of its estimated maximum surface area in mid-June as a function of wetland size for wetlands with permanent stream connections (*permanent*), seasonal connections (*seasonal*), and strictly isolated wetlands (*isolated*) sampled in DEWA-NJ in June of 2005

sonal connections, and 25 times less likely to maintain water levels as those with permanent connections (Fig. 9.5). Longer hydroperiods benefit all species of pond-breeding amphibian because it allows more time for their larvae to complete metamorphosis. However, longer hydroperiods are also associated with higher abundances of aquatic predators, and surface water channels/connections allow dispersal corridors to allow ponds to be colonized by fish. We did, in fact, find that fish were four times more likely to be found in ponds with permanent connections than seasonal connections, and fish presence was not significantly related to wetland size (Table 9.4).

In one set of occupancy models, we replaced the predictor of hydrologic connectivity with the observed presence of fish at ponds. In these models, fish presence negatively influenced the occurrence of *Ambystoma maculatum* (spotted salamander), *Hyla versicolor* (grey treefrog), and *R. sylvatica*, suggesting the benefit of longer hydroperiods are offset by the costs of fish predation for these species (Table 9.4). However, in the remaining six species, fish failed to show a negative impact on occurrence and several of these species did appear to benefit from the more stable hydroperiods of wetlands with permanent and seasonal hydrologic connections.

9.3.4 Importance of Small, Isolated Wetland on Early-Breeding Amphibians

We used amphibian occupancy models to predict species occurrence across the entire landscape of breeding ponds in DEWA-NJ. Our model validation results were

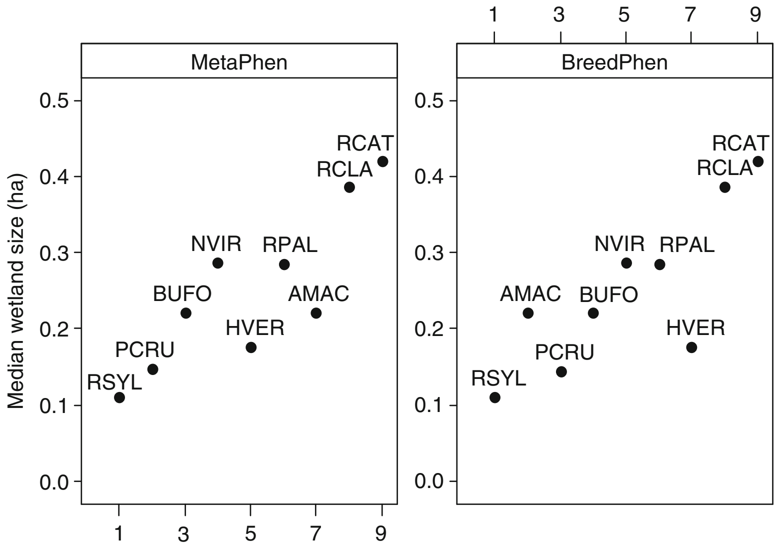

Fig. 9.6 Species rankings for metamorphosis phenology (MetaPhen) and breeding phenology (BreedPhen) plotted against the median size of predicted breeding wetlands (in hectares) for amphibian species in NJ-DEWA (AMAC *A. maculatum*, BUFO *Bufo* species, HVER *H. versicolor*, NVIR *N. viridescens*, PCRU *P. crucifer*, RCAT *R. catesbeiana*, RCLA *R. clamitans*, RPAL *R. palustris*, RSYL *R. sylvatica*)

comparable to, or better than, predictive models found in the literature (Kolozsvary and Swihart 1999; Dettmers et al. 2002; Hepinstall et al. 2002; Klute et al. 2002; Shiner et al. 2002; Suzuki et al. 2008), with a mean correct classification rate of 72.9% across all nine species (min = 62.5%, max = 90.6%). We found that amphibian species that breed early in the year were predicted to breed more frequently in smaller, hydrolgically isolated ponds than species with later breeding phenologies. We quantified breeding phenology by ranking all species by the time of the year they initiated egg-laying, and similarly quantified metamorphosis phenology based on when a species larvae complete metamorphosis.

We found a species breeding phenology (early to late), was positively correlated with the median size among all ponds where the species was predicted to occur ($r = 0.811$, $p = 0.008$) (Fig. 9.6). This same trend held up when correlating metamorphosis phenology with median pond size ($r = 0.854$, $p = 0.003$). Most notably, early-breeding species such as *R. sylvatica* and *P. crucifer* were predicted to contain at least half of their populations in wetlands smaller than 0.15 ha.

In the last decade, trends in small wetlands loss have likely had a disproportionate impact on pond-breeding amphibians with early-breeding phenologies. Recent decadal assessments of wetlands in the United States (Dahl 2006, 2011) suggests that half of the wetland area destroyed was among wetlands <0.5 ha in size. Given the size distribution of breeding ponds in DEWA, these trends would translate into the destruction of 22 wetlands smaller than 0.5 ha, for the destruction of every one wetland larger

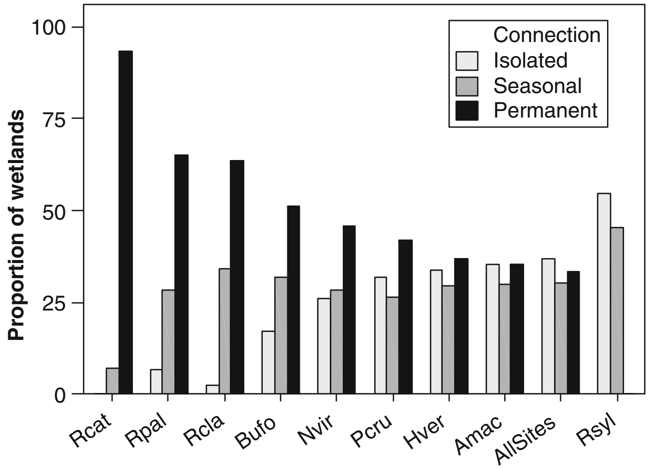

Fig. 9.7 The proportion of predicted breeding wetlands distributed among hydrologic connectivity classes in DEWA-NJ (Amac *A. maculatum*, Bufo *Bufo* species, Hver *H. versicolor*, Nvir *N. viridescens*, Pcru *P. crucifer*, Rcat *R. catesbeiana*, Rcla *R. clamitans*, Rpal *R. palustris*, Rsyl *R. sylvatica*, AllSites *All potential breeding wetlands*)

than 0.5 ha. In the conservation of pond-breeding amphibians it is important to assess wetland loss in terms of the number of wetlands lost because these species tend to establish metapopulations across the landscape, where individuals show a high degree of fidelity to a single breeding pond. Wetlands <0.5 ha contain the majority of breeding sites for all amphibian species, and would include nearly all ponds containing early-breeding species like *R. sylvatica* and *P. crucifer*.

Hydrologically, isolated ponds were also associated with the predicted occurrence of early-breeding species. The proportion of a species breeding sites classified as hydrologically isolated was negatively correlated with both breeding phenology (early to late) ($r = -0.834$, $p = 0.005$), and metamorphosis phenology ($r = -0.738$, $p = 0.023$) (Fig. 9.7). Only 1/3 of the species were predicted to have more than half of their breeding sites in ponds with permanent stream connections, despite these ponds being nearly three times the size or other wetlands.

Federal protection of isolated wetlands has diminished since the Supreme Court decisions in *Rapanos vs. U.S. Army Corps of Engineers* (126S. Ct. 2208, 2006), and *Solid Waste Authority of Northern Cook County v. United States Army Corp of Engineers* (531 U.S. 159, 2001) (i.e., *SWANCC*). In *SWANCC*, isolated wetlands that provided habitat for migratory birds lost their jurisdictional status under the Clean Water Act, and the plurality opinion in *Rapanos* suggested a permanent hydrologic connection to a navigable body of water was necessary to protect a wetland under the Clean Water Act. If federal protection was the only means of preserving wetlands, it appears that most pond-breeding species we studied would lose at least half of their breeding populations (Fig. 9.7).

Table 9.5 Summary of regression coefficients ($\beta \pm 1$ SE) for probability of occupancy (ψ) from occupancy models that incorporate fish presence, but not hydrologic connectivity for wetlands in the DEWA in 2005

Species	ΔAIC from final model	ψ Covariate with regression coefficient				
		Area[a]	Fish[b]	P250[c]	P25[d]	Ponds 1 km[e]
A. maculatum	+1.58	1.93 ± 6.85	−0.66 ± 1.52	3.24 ± 2.46		
N. viridescens	+2.00		−0.080 ± 1.44	7.76 ± 3.51	−7.98 ± 2.92	
H. versicolor	+1.19		−1.88 ± 2.12		−12.6 ± 5.7	1.03 ± 0.53
P. crucifer	+5.91	1.17 ± 0.63	0.16 ± 0.88			
R. catesbeiana	+4.21		1.49 ± 1.08			
R. clamitans	+9.58	2.00 ± 0.72	1.99 ± 0.95			
R. palustris	+6.14	1.74 ± 0.71	1.52 ± 0.99	−4.70 ± 2.49		
R. sylvatica	−4.83		−2.67 ± 0.78			

[a]Area: wetland area
[b]Fish: fish present = 1, fish not present = 0
[c]P250: proportion of forested area within 250 m of flooded perimeter with >50% canopy cover
[d]P25: proportion of forested area within 25 m of flooded perimeter with >50% canopy cover
[e]Ponds 1 km: number of amphibian breeding wetlands within 1 km of flooded perimeter

9.3.5 Importance of Wetland Assessments that Characterize Connectivity

Our work with pond-breeding amphibians illustrates the usefulness of classifying wetlands using a taxonomy that implicitly considers hydrologic connectivity. Currently, national trends in wetland loss are framed within a system (Cowardin et al. 1979) that classifies freshwater, lentic wetlands in the context of vegetation communities. This system does a poor job of linking wetland functions to wetland types (Dahl 2006), and has been unsuccessful at predicting amphibian species composition (Knutson et al. 1999).

The HGM classification systems do classify wetlands according to their hydrologic connectivity (e.g., Brooks et al. 2011), and we believe HGM does a significantly better job at linking wetland functions to wetland types. To support this, we offer evidence that hydrologic connectivity affects amphibian species composition, predatory fish presence, and hydrologic stability. While not easily executed, HGM wetland classification can perfectly match wetland function(s) to wetland class. Using a HGM approach has been shown to predict important biotic and abiotic factors that shape ecological communities and influence wetland functions.

9.4 Sampling Streamside Salamanders Across an Acidic Gradient

A properly selected and evaluated ecological indicator is responsive to the condition of the environment it purports to indicate. The biphasic life cycle and permeable skin of amphibians, among other attributes, make them potentially ideal ecological indicators. Considerable efforts were expended in the 1980s to early 1990s to evaluate the impact of acid deposition on vernal pool (isolated depression wetlands) amphibians and small stream-dwelling fishes. These studies, many of which were completed at Riparia at Penn State, confirmed that low pH, concomitant increases in Al, and other dissolved heavy metals, as well as other impacts on water chemistry from acid rain, posed significant challenges to amphibian survival. Osmoregulatory failure, resulting from disruption of the Na pump was shown to be the primary mechanism for physiologic stress and mortality. Acidic and metal bearing waters were also implicated in the disruption of food webs, predator–prey interactions, and diminished growth and fitness.

In the late 1990s, Riparia, funded by the U.S. Environmental Protection Agency's Environmental Assessment and Monitoring Program (EMAP), evaluated amphibians as bioindicators. The acid deposition research cited earlier suggested that sampling amphibian assemblages across an acid gradient would provide unprecedented opportunities to evaluate their usefulness as bioindicators. An in-depth literature review revealed that stream plethodontids, lungless stream-dwelling salamanders, might be ideally suited for this purpose. Several attributes suggested their potential suitability. Population size and age structure of stream plethodontids tend to be more stable than vernal pool-breeding amphibians. Less variability would facilitate stressor response detection and measurement, i.e., signals from stressors well above "background noise." Stream plethodontids are relatively diverse, abundant as well as widespread in most low order Appalachian streams and up to seven species may be found in Pennsylvania. Life histories are highly variable and stream dependence ranges from minimal to high (Table 9.6).

For example, Northern Spring Salamanders (*Gyrinophilus p. porphyritics*) oviposit on the underside of stream-submerged rocks and hatchlings require up to 4.5 years as aquatic larvae before transforming into their final terrestrial life stage. Their survival and continued presence is intimately tied to the stream. The Mountain Dusky salamander (*Desmognathus ochrophaeus*), on the other hand, is on the opposite end of the life history and stream dependence spectrum. Grape-like clusters of eggs are oviposited and guarded by the mother in moist terrestrial cavities. Hatchlings move to streamside shallows after a brief nest sojourn to continue their development as aquatic larvae for several months. In some populations, hatchlings transform into miniature adults within the nest and effectively bypass the aquatic larval stage.

Thus, even within such a small assemblage the odds were in favor of seeing marked responses to varying stream conditions, i.e., aquatic taxa and life stages would probably emerge as the least tolerant (most sensitive) of stream degradation whereas the opposite (highly tolerant) would occur in the more terrestrial forms.

Table 9.6 Summary of life histories for Pennsylvania Central Appalachian lungless salamanders, family Plethodontidae

Stream dependence	Taxa	Environment of		Months to transformation
		Nest	Larvae	
None	Woodland salamanders (*Plethodon* spp.)	Terrestrial	No larval period; hatchlings are miniature adults	
	Mountain Dusky Salamander (*D. ochrophaeus*)	Moist terrestrial	Brief nest residency, followed by move to aquatic streamside habitat to continue development	2–8
	Northern Dusky Salamander (*D. f. fuscus*)			7–10
	Appalachian Seal Salamander (*D. monticola*)			12–13
	Long-tailed Salamander (*E. longicauda*)	Aquatic	Aquatic	3–3.5
	N. Two-lined Salamander (*E. b. bislineata*)			24–36
	N. Red (*P. r. ruber*)			32–36
High	N. Spring Salamander (*G. p. porphyriticus*)			36–48

Stream dependence is none to minimal in woodland salamanders and becomes increasingly more important to critical in stream-dwelling forms with some taxa requiring up to 3–4 years to transformation as aquatic larvae

Table 9.7 Stream water chemistry by stream condition category for the 14 Pennsylvania watersheds sampled in 1997–1998

Attribute	AMD, high acid (watersheds = 3)	Episodic acidification, low acid (watersheds = 3)	Reference, nonacidic (watersheds = 4)	Fragmented, high alkalinity (watersheds = 4)	*p*-value <
Temperature (C)	13.4 ± 0.72, **25** AB	12.4 ± 0.79, **15** AB	11.75 ± 0.60, **27** A	14.68 ± 0.69, **39** B	0.015
pH	4.32 ± 0.16, **27** C	4.83 ± 0.11, **24** B	6.91 ± 0.07, **29** A	7.63 ± 0.03, **40** D	0.000
$CaCO_3$ mg/L	0.27 ± 0.12, **27** B	0.06 ± 0.01, **16** A	11.29 ± 1.22, **29** A	55.51 ± 4.04, **40** C	0.000
NO_3-N mg/L	0.21 ± 0.02, **25** B	0.12 ± 0.02, **22** B	0.57 ± 0.05, **25** A	0.62 ± 0.04, **34** A	0.000
ANC mg/L[1998]	−326 ± 116.7, **16** B	−6.6 ± 2.19, **14** A	180.7 ± 39.4, **13** A	1,052 ± 95.5, **16** C	0.000
DOC mg/L[1998]	2.5 ± 0.22, **16**	1.91 ± 0.23, **14**	2.17 ± 0.36, **13**	2.83 ± 0.24, **16**	NS
SO_4 mg/L[1998]	249 ± 80.5, **16** B	8.5 ± 0.27, **14** A	7.1 ± 0.64, **13** A	47.6 ± 9.05, **16** A	0.000
Fe mg/L[1997]	0.18 ± 0.05, **9** A	0.32 ± 0.13, **8** B	0.05 ± 0.01, **12** A	0.18 ± 0.04, **18** A	0.04
Fe mg/L[1998]	1.17 ± 0.47, **16** B	0.10 ± 0.03, **6** AB	0.06 ± 0.01, **13** A	0.06 ± 0.01, **16** A	0.013
Mn mg/L[1998]	7.06 ± 2.6, **16** B	0.05 ± 0.01, **6** AB	0.00 ± 0.00, **13** A	0.02 ± 0.02, **16** A	0.003
Al mg/L[1997]	0.45 ± 0.23, **9** B	0.51 ± 0.06, **8** B	0.04 ± 0.01, **12** A	0.05 ± 0.01, **18** A	0.001
Al mg/L[1998]	15.1 ± 5.61, **16** B	0.22 ± 0.03, **14** A	0.03 ± 0.00, **13** A	0.04 ± 0.00, **16** A	0.001

Cell contents include mean and standard error + or – (SE), and number of samples (**n**). Significant differences across categories of stream condition are based on one-way ANOVAs with Tukey's test for multiple comparisons. Within rows, cells with different letters are significantly different at *p*-value >0.05. AMD = acid mine drainage-contaminated

Variable tolerance among assemblage members to environmental conditions is the basis for an effective bioindicator group.

In 1997 and 1998, painstaking hand-sampling of 340, 4-square meter quadrats (equal to 1,360 m^2 of wet and dry streambank) from 14, first to third order streams resulted in the capture of slightly more than 4,000 stream plethodontids. Laboratory analysis of stream water grab samples from 58 sampled transects (there were 5 quadrats per transect) confirmed that sampling had indeed occurred in a strong acid to alkaline gradient and across highly variable water chemistries (Table 9.7). For example, pH ranged from 3.73 to 8.31, whereas alkalinity ($CaCO_3$ mg/L) and dissolved Al (mg/L) ranged from 0.01 to 84.8, and 0.01 to 4.64, respectively. Previous studies had revealed amphibian intolerance or toxicity to conditions well within the range of the aforementioned water chemistries.

The study revealed that stream plethodontid abundance, presence, and diversity were severely suppressed in acidified environments (https://etda.libraries.psu.edu/paper/7854/). Quadrats from acid mine drainage (AMD) impacted headwaters were largely devoid of aquatic and terrestrial life stages (median plot density = 0.5 salamander/m^2). Alkaline watersheds in fragmented landscapes, on the other hand, were quite the opposite and accounted for the capture of 85% of all salamanders despite more than 40% of plots being located in acidic watersheds. Median density in nonacidic plots was 4.5 salamanders/m^2 with some quadrats producing as many as 20–30 animals per square meter, over 100 salamanders in an area twice the size of an average office desk!

Acidic streambanks not only had fewer salamanders and fewer species but the species present also differed from those commonly found in nonacidic sites, e.g., the Mountain Dusky commonly replaced the Northern Dusky in more acidic streambanks. Thus, the reduction in assemblage complexity resulted from decreased abundance, reduced taxonomic and life stage diversity, and species replacement.

There were also unexpected findings. Early amphibian life stages (aquatic gilled forms) were observed across a greater variety of environments, trended towards less optimal habitats than their adult counterparts. While counterintuitive from a purely physiological and toxicological standpoint, early life stages disperse widely and may have yet to succumb to the stressors that limit persistence in older or adult forms. The other unexpected finding was that life history and nesting habits were poor predictors of survival or tolerance in acidic streams. The Northern Spring Salamander, the taxon most dependent on the aquatic environment (aquatic nester, longest aquatic larval stage, etc.), should have been the most adversely affected in acidic streams. In effect, it was surprisingly persistent and was almost as hardy as the Mountain Dusky, a terrestrial nester that may even bypass the aquatic stage. Subsequent in situ testing suggested that physiologic and perhaps behavioral adaptations, similar to other acid-tolerant species may explain the persistence of the Northern Spring Salamander in acidic streams.

The research described herein led to a second 3-year study to investigate the responses of stream plethodontids across a wider range of stream conditions throughout the Mid-Atlantic Highlands. That report may be accessed online at the following url: http://cfpub.epa.gov/ncer_abstracts/index.cfm/fuseaction/display.abstractDetail/abstract/281/report/F.

9.5 Habitat Conservation Plan for the Bog Turtle

In 2003, a team of wildlife biologists, land stewards, legal counselors, and regulators from Riparia at Penn State, Brandywine Conservancy, Natural Lands Trust, Delaware Department of Natural Resources & Environmental Control, the Pennsylvania Fish & Boat Commission, and Environmental Defense, among other partners, sought to develop a conservation plan for the northern population of the bog turtle (*Glyptemys muhlenbergii*) for portions of Chester County, Pennsylvania and New Castle County, Delaware, to address some of the shortcomings and conservation measures in place. Funding was provided, primarily by the U.S. Fish and Wildlife Service (USFWS).

9.5.1 Background and Needs

The northern population of the bog turtle is listed as federally threatened and is endangered in many northeastern states. Pennsylvania is considered to be its regional stronghold, yet of the 100 or so sites known to be occupied, only a fraction of these are suspected to support expanding colonies. The bog turtle is a classic K-selected species: it exhibits slow growth, low reproduction, and is long-lived. It occurs in small colonies of several individuals to several dozen. Its preferred habitat, consisting of open, boggy, spring-fed wetlands, is threatened primarily by neglect and isolation.

Natural succession, accelerated by the absence of grazers and other natural disturbances, and by increased nutrients, is replacing once sunny, herbaceous wetlands with shady, wooded environments. Urbanizing landscapes contribute to habitat degradation and interfere with dispersal patterns. Loss of habitat connectivity threatens genetic fitness, the persistence of smaller, non-self-propagating colonies, and the establishment of new colonies in non-occupied habitats. Location is vital in determining colony viability over the long term in human-dominated landscapes. The loss of some occupied sites is inevitable as a result of these factors.

9.5.2 The Habitat Conservation Planning Project: A Solution?

The proposed HCP project sought the recovery of ecologically important sites to the bog turtle. Its goal was to accelerate the targeted protection and management of sites that remained connected and in dispersal-friendly landscapes. It sought to provide a mechanism to avoid wasting resources at nonviable sites and to generate funding from their incidental taking, unintended harm while completing otherwise legal development projects and activities. Although the endangered species act (ESA) prohibits the "taking" of federally listed species, incidental take authorization is possible through an "incidental take permit" (ITP) issued by USFWS. A regional HCP offered the opportunity to offset the loss of inferior sites with perpetual protection and management of superior sites. The HCP was intended to focus recovery efforts in areas most suited for long-term protection and sustainability. The process ultimately aimed at achieving a net positive benefit from the loss of "island" habitats likely to be lost anyway.

9.5.3 The Program in Brief

Important sites located in less urbanized, development-threatened landscapes were identified and established as potential conservation banks (Fig. 9.8). The conservation banks would generate credits that had to be purchased to offset incidental take

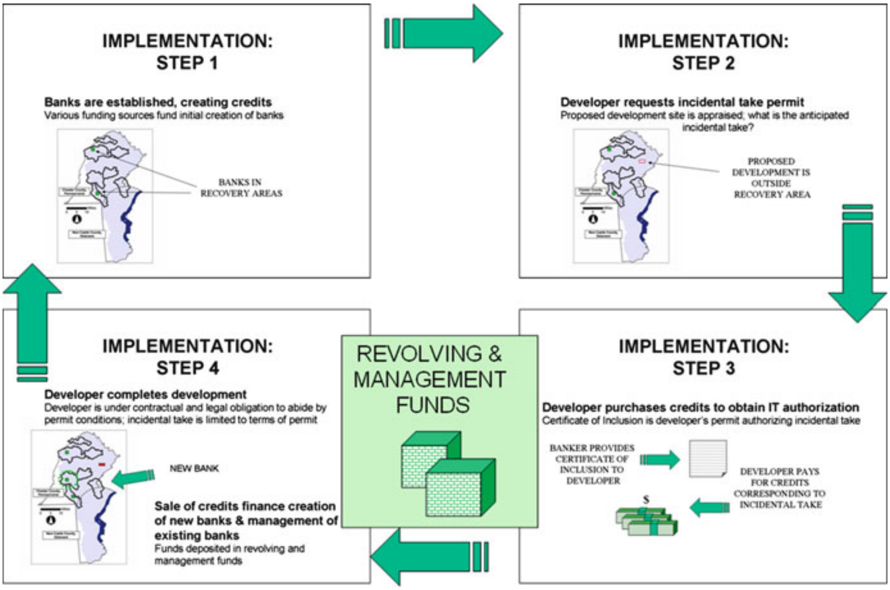

Fig. 9.8 Diagrammatic summary of proposed Bog Turtle conservation banking process

elsewhere when voluntary HCP-program participation was sought. The sale of credits would help to establish and replenish funds used to finance the creation of new banks and perpetual management of existing banks. Revolving and management funds would have been established specifically for this purpose by a certified banker(s). Thus, the ITP, issued in the form of a certificate of inclusion, would authorize the incidental take. The recipient would be under contractual and legal obligation to abide by the permit conditions and the amount of take authorized would be limited to the conditions of the permit.

9.5.4 Conservation Banks and Recovery Areas

In this HCP, conservation banks were the ecologically superior sites that had to be perpetually protected and managed to favor the recovery of the bog turtle. Banks were envisioned to comprise one or more properties each containing core habitat (wetlands) and the surrounding uplands to serve as buffer and recharge. Most banks would be occupied by bog turtles; others may not have been occupied when initially established, but support high quality habitat to provide new sites for expanding neighboring colonies or facilitate their connectivity. Banks would have been owned or leased by qualified bankers, local conservancies, or alternatively owned and managed by a newly created conservancy to serve this specific role. Multiple banks would have been established in each recovery area. Some banks would have been fee-simple acquired parcels, others would have been located in existing easements or on public lands. Because of initial real estate costs, fee-simple acquired properties would have been the most expensive type to establish. Most banks would have generated credits, with the sale of credits intended to offset bank maintenance costs and to help fund the creation of new banks.

Most conservation banks were envisioned to be located in recovery areas, landscapes within the "service area" most likely to favor bank connectivity over the long term. At the time of project completion, seven recovery areas were identified. The service area in conservation planning jargon is the geographic area where incidental take authorization may be mitigated by the use of credits generated from banks. As indicated earlier, the service area covers portions of the Delaware west bog turtle recovery area within Chester County, Pennsylvania and New Castle County, Delaware.

9.5.5 Appraising Banks and Incidental Take Sites

The appraisal of banks and proposed development sites was a crucial component of the proposed program as it provided the means to track progress as well as incidental take. Are banks meeting the goals and objectives of the HCP? Is the amount of incidental take occurring within the authorized limit? Creation of a market for credits, based on the appraisal of natural resources valuable to bog turtle recovery, was designed as the backbone to financing bog turtle recovery efforts in banks.

Rules were developed for the appraisal process for banks and incidental take sites. The basis for appraisal was the amount and quality of acres in a property corresponding to each of three bog turtle conservation zones, namely, Zone 1, the core habitat and surrounding wetlands, Zone 2, the upland buffer, and Zone 3, uplands 100 m beyond the delineated wetland edge (Fig. 9.9). The quality and status of Zone 1 was designed to yield the greatest amount of credits and in turn, determined the appraisal of acres within Zones 2 and 3. Bog turtle colony attributes, whether confirmed or suspected, were the key criteria driving the appraisal. Properties inside recovery areas were appraised higher than parcels outside them. The appraisal ultimately sought to discourage incidental taking of sites more valuable conserved as banks.

9.5.6 Focus on Common Concerns

The ITP issued to developers in any HCP is not a license to kill or to cause unrestrained harm. ITPs are issued only when the anticipated taking will be incidental and while conducting otherwise legal activities. An ITP does not cover intentional or deliberate harm, i.e., the purposeful killing or removal of turtles or for that matter,

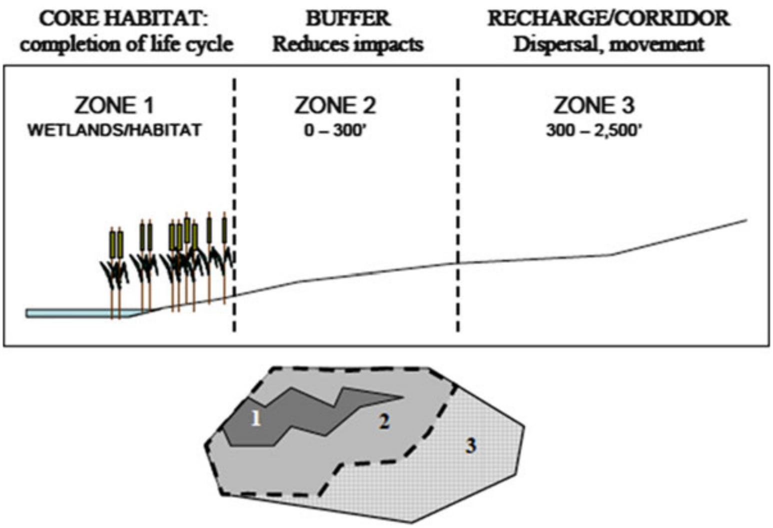

Fig. 9.9 Bog turtle conservation zones: Zone 1 Core habitat for breeding (most valuable), Zone 2 Buffer (0–100 m), Zone 3 Recharge area, and/or dispersal corridor (100–800 m; least valuable)

other protected species and habitats. Most importantly, all other environmental regulations governing the planned activity must be met. Furthermore, it requires incidental take to be minimized to the extent possible, in addition to its mitigation. In this project, the purchase of credits represented the latter.

No incidental take would have been authorized prior to the purchase of credits. Since credits can only be generated from approved, existing banks, creation of banks had to precede incidental taking. In other words, activities benefiting the species had to occur before incidental harm (incidental take) was authorized or could begin. This condition served as a very effective stop point. It was also designed as an incentive to establish banks.

The amount of incidental take established at the Master Permit level was set low. Using GIS-modeled occupied habitat, incidental take authorized was set to a fraction of the total present. Furthermore, once the amount of incidental take authorized under the Master Permit was reached, no additional take could occur under this same permit without reauthorization.

Some easement owners were uncomfortable to have credits generated from their properties because the process was viewed as "facilitating" or "speeding-up" development. Others argued that the sale of credits from eased banks did not protect new lands. Program participation was strictly voluntary, so landowners could decide if their easements generated credits or not. Protected lands occupied or conducive to bog turtle occupancy were important parts in the conservation network even if they did not generate credits.

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Chapter 10
Freshwater Macroinvertebrates of the Mid-Atlantic Region

Susan E. Yetter

Abstract Freshwater macroinvertebrates are an extremely diverse and adaptive group of organisms that have successfully invaded virtually every type of aquatic habitat. This chapter provides a habitat-based description of Mid-Atlantic region (MAR) macroinvertebrates, beginning with a review of adaptations to differing aquatic environments, followed by a synopsis of Riparia's research regarding macroinvertebrate bioassessments in various hydrogeomorphic (HGM) wetland types. Early research revealed that the HGM classification was insufficient in controlling natural variation, making it difficult to assess community responses to anthropogenic disturbance. This prompted an effort to develop a more ecologically relevant habitat classification for MAR wetland macroinvertebrates. The remainder of the chapter presents a case study of this habitat approach applied to headwater and floodplain complexes. First, we defined the riverine hierarchy by building on existing classification schemes and adding a level, the *aquatic ecological set*, to differentiate between habitats structured by flow, flood, and groundwater pulses. Next, we compiled the macroinvertebrate data collected from all aquatic habitats within reference-standard floodplain reaches and did an exploratory analysis, which revealed six major habitat types: riffle, other baseflow, and flow pulse habitats in the active zone, flood pulse habitats in the floodplain, seasonal groundwater, and temporary habitats. Further comparison with impacted riverine complexes indicated these systems respond to anthropogenic disturbance primarily through changes in hydrological connectivity and hydroperiod. The end result is loss of flow pulse habitats, floodplain terrestrialization, and loss of heterogeneity in wetland habitats, the latter primarily through a shift from seasonal to either permanent or ephemeral hydroperiods.

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10.1 Introduction

Freshwater macroinvertebrates are animals lacking an internal skeleton that generally can be seen by the naked eye (>2 mm in size) and inhabit inland waters. They represent a crucial part of aquatic food webs and are important for breaking down organic matter and nutrient cycling (Voshell 2002). These highly opportunistic organisms occur in nearly every type of aquatic habitat possible; even a mug of water left outside will eventually support macroinvertebrate organisms. The Mid-Atlantic region (MAR) contains a wide variety of freshwater aquatic habitats ranging from small, isolated wetland pools to vast lakes to extensive river systems. As a result, this area is home to a wide range of macroinvertebrate species. Because these assemblages are largely dependent on the physicochemical nature of the hydrological environment in which they reside, the approach for describing macroinvertebrates within the MAR is primarily habitat-based.

The chapter begins by discussing macroinvertebrate adaptations to different wetland environments, ranging from fast-flowing headwater streams to stagnant, isolated pools. This provides a context for viewing MAR freshwater wetlands from an organismal perspective. From there, the discussion moves on to describe earlier Riparia research in macroinvertebrate ecology, which focused primarily on wetland bioassessment and used the hydrogeomorphic (HGM) wetland classification to control for natural variation in both aquatic habitats and their associated communities. This is followed up with a "lessons learned" discussion of the disadvantages of relying exclusively on an a priori HGM classification to partition natural variation, especially when the underlying goal is to understand patterns of community responses that stem primarily from the success or failure of species' adaptations. The second half of the chapter presents a case study using a habitat approach (defined post priori through evaluation of macroinvertebrate community data) to explain pattern and process in riverine systems and involves the following: (1) defining a riverine hierarchical classification; (2) characterizing the reference standard for riverine macroinvertebrate community types and their habitats; (3) recognizing the importance of connectivity between habitats and its link to biodiversity; and (4) incorporating this information into wetland bioassessments.

10.2 Macroinvertebrate Adaptations to Differing Aquatic Environments

To utilize aquatic habitats, macroinvertebrates must be able to: (1) withstand the physical conditions of the environment (e.g., fast current, dry periods, etc.); (2) adjust to various physiological constraints (e.g., dissolved oxygen availability, osmoregulation, etc.); (3) acquire sufficient food for survival, growth, and reproduction; and (4) successfully compete for resources and avoid predation. The importance of each of these four factors changes within different wetland types. Temporary environments with short and/or flashy hydroperiods create harsh physical conditions that limit the types of organisms able to survive there. Biotic interactions play less of a role in determining community structure in these "physically controlled" communities, whereas the opposite is true of wetlands with permanent hydroperiods that provide a stable physical environment where "biologically accommodated" communities are structured primarily through predation (Sanders 1968; Schneider 1999). As a result, inhabitants of temporary environments typically demonstrate physical, physiological, or behavioral adaptations for withstanding dry periods, whereas those in permanent wetlands have evolved adaptations for capitalizing on predator/prey dynamics (Williams 2006).

Insects comprise the bulk of aquatic macroinvertebrate species and display the widest variety of adaptations, having invaded nearly every possible type of aquatic habitat in the MAR (Hynes 1970; Huryn et al. 2008). Insect species undergo either a complete or incomplete metamorphosis, with many species only requiring aquatic habitat during the larval and/or pupal stage. This requires, however, that the existence of aquatic habitat coincide with the dependent life history stage for the necessary duration. Most temperate species of aquatic insects have univoltine (1 year) life histories that exhibit strong seasonal synchrony (i.e., timing of emergence and oviposition) that are largely brought on by temperature and photoperiod cues (Hynes 1970; Newbold et al. 1994) and require at least part of the year in a state of diapause.

10.2.1 Adaptations to Lotic Environments

High gradient streams in the MAR are typically of pool/riffle regime and consist largely of inorganic substrates (e.g., boulder and cobble) with a variety of velocity/depth regimes (e.g., fast shallow, slow deep). Oxygen is rarely a limiting factor in these habitats, so cutaneous respiration and tracheal gills are common adaptations (Perlidae, Heptageniidae, Ephemerellidae, etc.) (Ward 1992). The main physical constraint is the current velocity, which can be well over 1 m^2/s (Hynes 1970; Huryn et al. 2008). Consequently, morphological adaptations for reducing drag and remaining stationary are a must in these environments and include streamlined and dorsoventrally flattened bodies, clinging appendages (e.g., hooks, claws, hydraulic suckers), and ballast (heavy stones incorporated into cases to add extra weight) (Hynes 1970; Ward 1992; Huryn et al. 2008). Swimming mayflies, such as *Isonychia* and *Cloeon*, provide good examples of streamlined bodies designed to either rest while exposed to rapid flow or to swim in slow flow. Flathead mayflies (Family Heptageniidae) are well known for their dorsoventrally flattened bodies, which both increases surface area for attachment and lowers drag by keeping the body within the lower velocities of the boundary layer. Of course, this also increases friction drag, thus increasing the probability of the organism "lifting" and being tossed into the current (Ward 1992; Huryn et al. 2008). These mayflies counteract this by anchoring themselves to the substrata with strong tarsal claws and legs. Stationary collector-filterers are quite common in these environments, such as black flies (e.g., Simulium) and net-spinning caddisflies (Hydropsychidae). Black flies anchor to substrate via tiny hooks in their posterior proleg, which are anchored to a pad they spin from silk glands (Adler and Currie 2008). Blephariceridae have highly specialized hydraulic suction cups with which they adhere to smooth stones; in fact, their entire morphology is designed to maximize the efficiency of these hydraulic suckers (Hynes 1970; Ward 1992; Huryn et al. 2008).

In lower gradient streams, oxygen is less available and slower currents result in accumulations of smaller inorganic substrates (i.e., silt and sand) and a mix of organic material (i.e., woody debris, leaves, and other detritus). Gill structures of these inhabitants are typically lamellate and operculate (first gill pair is overly large and covers subsequent pairs). This protects the gills from fine sediment accumulation. Those with filamentous gills will undulate to create ventilator currents and increase oxygen uptake (e.g., Neophylax) (Hynes 1970; Wiggins 1996). Other adaptations include dense coverings of long, fine hairs (to intercept fine sediment), hooks and tusks for burrowing in soft substrates, flattened bodies and lateral projections for resting atop soft substrate, and cryptic coloration for disguise. The mayfly *Eurylophella* and the caddisfly *Pycnopsyche* are common inhabitants of both low gradient headwater streams that lack riffle habitat and the marginal areas of higher gradient streams where silt, sand, and organic matter collect (Smock 1994; Ward 1992; Huryn and Gibbs 1999).

10.2.2 Adaptations to Permanent Wetland Environments

Permanently inundated depressional wetlands typically contain a mix of inorganic and organic substrates (e.g., muck, aquatic vegetation, sphagnum). Oxygen can be very limited in these environments, which are often anoxic, requiring sophisticated adaptations from the resident biota (Hynes 1970; Ward 1992). Examples include respiratory siphons (e.g., Culicidae, Syrphidae, Stratiomyiidae) and the accompaniment of a physical gill (e.g., Dytiscidae) for obtaining atmospheric oxygen (Ward 1992). For wetlands with open water, the surface film serves as an important habitat type. The surface film provides habitat for a community known as the *pleuston*, which includes organisms that live on the surface and those which live in the water below but stay suspended to the surface (Huryn et al. 2008). Special adaptations for the water surface include a hydrofuge (piles of hair that resist water), which allows water striders (Gerridae) to skate across the water surface. Gyrinids (whirligig beetles) are well adapted for life amidst the water surface (Hynes 1970). They have two sets of eyes, one ventral and one dorsal, which enable them to see both underwater and above the surface simultaneously (Huryn et al. 2008). Larvae living suspended from the water surface typically rely on atmospheric oxygen (e.g., Culicidae, Chaoboridae, and Dixidae) (Ward 1992; Courtney and Merritt 2008).

Biotic interactions are quite common and predator diversity is high, especially within the Odonata, Hemiptera, and Coleoptera. Consequently, diverse arrays of feeding and foraging strategies are utilized in these habitats. Odonates have specialized feeding appendages in the form of extendable labrums which quickly dart out to capture prey (Tennessen 2008). Modified hind legs for movement in water are also common: Dytiscid beetles have swimming hairs and appendages adapted for diving; while Corixidae (waterboatmen) have paddle-shaped tarsal claws for swimming (Hynes 1970; Ward 1992). Snails are common on plant stems and leaves, where they scrape diatoms and other food particles from the surface.

10.2.3 Adaptations to Temporary Environments

Perhaps the most unique adaptive strategies can be found in temporary wetlands, which represent quite harsh environments (deathtraps or sinks) for unprepared stragglers. Organisms in these habitats must be able to adapt to alternating wet and dry conditions, oxygen limitations, and wide temperature fluctuations. The tradeoff, however, is a much more conducive environment for rapid larval growth through a greater abundance and variety of food resources, warmer temperatures, and lower risk of predation (Mackay 1992; Matthaei et al. 1996; Kosnicki and Burian 2003).

Macroinvertebrate adaptations in these environments are diverse. Hynes (1970) lists six main strategies for surviving in temporary aquatic environments: (1) physiological tolerance to high temperatures and low oxygen concentrations; (2) burrowing into fine substrates as the water recedes; (3) diapause or aestivation; (4) recolonization from nearby permanent habitats; (5) highly specialized morphological, behavioral, or life history adaptations; and (6) utilization of habitats only during the drawdown or dry periods. Williams (1996) and Wissinger et al. (2003) consider rapid larval development and timing of emergence (5), adult ovarian or egg diapause (3), and egg or larval desiccation resistance (1) to be key life history traits that have enabled aquatic insects to invade these environments and make the transition from permanent to temporary hydroperiods. Most taxa employ a variety of these strategies.

Fingernail clams (*Pisidium*) and planorbid snails are particularly well adapted for temporary environments. Both burrow into the substrate to escape desiccation (Hynes 1970; Pennak 1989). Other strategies include sealing shell openings with dry mucus during dry periods, hitching a ride with an unsuspecting traveler (e.g., water birds) to a more suitable habitat, or closing valves and entering a state of dormancy (Hynes 1970; Gladden and Smock 1990). Moreover *Pisidium* populations in unstable habitats exhibit more r-selective strategies with shorter generation times and larger litter sizes than populations in more stable, permanent habitats (Bailey and Mackie 1986).

Populations of small organisms with short life cycles that reproduce and disperse rapidly are known as “r strategists” (Batzer and Resh 1992; Anderson 1999). Common in temporary habitats, they are usually the first to appear and represent early colonizers (Hynes 1970; Mackay 1992; Jacobsen and Encalada 1998). Members of the biting midge genus *Culicoides* proliferate in temporary waters, especially tree hole habitats that can be very short-lived. Their short life cycle (2–3 weeks from oviposition to adult emergence) allows them to persist in these highly ephemeral environments (Mullens and Rodriguez 1989; Kruger et al. 1990; Barrera 1996; Paradise 1998).

Many mayfly and caddisfly genera have highly specialized life histories, which allow them to utilize temporary environments, either for a particular life stage or for their entire aquatic life cycle. For example, female mayflies of the genus *Siphlonurus* oviposit in the stream channel, remain in the channel during the dry period, and migrate as larvae to the floodplain in late winter or early spring where they develop rapidly from March to May, emerging just prior to the dry period (Voshell 1982; Huryn and Gibbs 1999; Kosnicki and Burian 2003). *Leptophlebia* displays a similar trend of seasonal migration to the floodplain prior to emergence, but unlike the swimming *Siphlonurus*, possesses the ability to crawl to less connected habitats. This may help explain why this mayfly can be quite abundant in wetland habitats located in the riparian zone that are not connected to the stream channel (Lauzon and Harper 1988; Smock 1994; Huryn and Gibbs 1999; Laubscher 2005).

Cased caddisflies, especially the family Limnephilidae, are often segregated along a habitat gradient of water permanence (Liebold 1995; Wissinger et al. 2003). Phryganeidae genera, for example, are restricted to more permanent water bodies, while *Limnephilus* larvae thrive in temporary wetlands by emerging in the spring as adults before habitats dry, entering an ovarian diapause in terrestrial vegetation during the dry period and reemerging in the fall to deposit desiccation-resistant egg masses under rocks and logs in areas that will become inundated again in the early spring (Liebold 1995; Wiggins 1996; Wissinger et al. 2003; Williams 2006). First instar larvae are protected in these gelatinous matrices until the water returns (Hoopes 1976). *Ironoquia parvula* is particularly well-adapted for temporary stream and floodplain environments—mature larvae burrow into the fallen leaves along the banks or pool edges just prior to the dry period and aestivate for several month before pupation. The dry period serves as a necessary cue for the life cycle change from larvae to pupae (Flint 1958).

Many taxa colonize temporary aquatic habitats via the air (Tronstad et al. 2007). This may be through: (1) adult oviposition and subsequent hatching of eggs (e.g., Chironomidae), (2) active flight by aquatic adults (e.g., Dytiscidae), or (3) passive dispersal (e.g., crustaceans) (Tronstad et al. 2007). Beetles (Coleoptera) and bugs (Hemiptera) are common taxa that actively disperse between aquatic habitats, typically as adults in aerial flight; however this requires exceptional stamina. For example, the dytiscid *Agabus*, which are known to fly long distances in search of newly inundated areas to lay their eggs, utilizes such a strategy. *Hydroporus* beetles, which have limited flight capabilities, rely primarily on burrowing to avoid dry periods (Williams 1996).

Even common stream taxa may require temporary aquatic environments. A good example is the water penny (Psephenidae). Although both larvae and adult *Psephenus herricki* live in riffle habitats, pupae cannot complete development if submerged (Murvosh 1971). In order to provide habitat for all life cycle stages, adults seek riffle substrates that are partially exposed as oviposition sites. Peckarsky et al. (2000)

observed adult *Baetis bicaudatus* females dispersing from the main channel to smaller tributaries where they oviposited exclusively on rocks protruding above the water's surface. They concluded that local recruitment depended on the availability and accessibility of the preferred ovipositing habitat. Such habitats are more likely to occur in perennial streams with side and secondary channels where more riffle habitat is exposed during dry periods.

Using habitat to understand patterns of community structure is not new (Hynes 1970; Southwood 1977). A long-standing premise in ecology is that "habitat provides the templet on which evolution forges characteristic life history strategies" (Southwood 1977; Townsend and Hildrew 1994; Bunn and Arthington 2002). Studies based strictly on taxonomy or community richness can be difficult to interpret, since many species or genera may fulfill a similar environmental or functional niche. As a result, many adopted a more functional approach beginning with Cummins' (1973) analysis of functional feeding groups. More recently, functional approaches have become more holistic by evaluating whole suites of traits that largely relate to overcoming environmental constraints (Southwood 1988; Poff 1997; Townsend et al. 1997; Poff et al. 2006, 2010; Verberk et al. 2008). Species traits represent key links between ecosystem pattern and process, but sifting through the tremendous variation in trait combinations for particular taxa, as well as overcoming obstacles of differing taxonomic resolutions, makes it difficult to determine which combinations of traits work best for particular habitat types (Verberk 2008). Identifying the appropriate scales at which organisms respond to their environment adds to this dilemma.

Failure to consider the link between biological traits and habitat, however, can lead to difficulties in discerning and diagnosing community responses to anthropogenic disturbance, especially in lesser-studied wetland habitats. A good example of this can be found in Riparia's earlier macroinvertebrate research (Bennett 1999; Conklin 2003; Laubscher and Conklin 2004; Laubscher 2005), which controlled for macroinvertebrate community variation through an a priori habitat classification based on HGM subclass.

10.3 Macroinvertebrate Communities and the Hydrogeomorphic Approach

10.3.1 Reference-Standard HGM Wetland Types

HGM classification of freshwater wetlands can be useful for evaluating macroinvertebrate community structure, because it is largely based on hydrologic characteristics and landscape position (i.e., where is the water coming from and where is it going?). Riparia used this approach to study macroinvertebrate communities in MAR wetland habitats. The following descriptions are for reference-standard riverine, slope, and depressional wetlands.

As mentioned in Chap. 2, riverine wetlands are associated with streams and rivers and may occur both within the stream (e.g., islands) and in the adjacent floodplain. Because the macroinvertebrate community spans both main channel and floodplain, the following descriptions also include instream habitat. Headwater streams are typically of two major types: high gradient and low gradient. The former are associated with the *riverine upper perennial* subclass and are characterized by narrow stream valleys with small, sporadic wetland habitats that can be highly ephemeral. Occasionally, they may form a *riverine headwater complex* in cases where the alluvial and/or hillslope aquifers support wetland habitat. The stream habitat consists primarily of inorganic substrates (e.g., boulder, cobble, gravel, and sand) and a variety of velocity/depth regimes. This combination of physical and chemical factors creates prime habitat for many EPT (Ephemeroptera, Plecoptera, and Trichoptera) taxa, particularly various stonefly genera (Leuctridae: *Leuctra*; Nemouridae: *Amphinemura*; Perlidae: *Acroneuria*; Pteronarcyidae: *Allonarcys*; Chloroperlidae: *Swelsta*; Peltoperlidae: *Tallaperla*; and Perlodidae: *Isoperla*, *Diploperla*). The riffle beetle *Oulimnius* (Elmidae) is also highly associated with headwaters and is rarely found in larger streams. Aquatic floodplain habitats associated with these streams are highly sporadic and usually consist of leaves, sticks, sand, silt, and overall "muck." They tend to be dominated by nonbiting midge larvae (Chironomidae) and aquatic annelids (Oligochaeta). Fingernail clams (Sphaeriidae: *Pisidium*) can be found in great numbers in all types of aquatic floodplain habitats associated with headwater streams (Laubscher and Conklin 2004).

Low gradient headwater streams contain little or no riffle habitats and are dominated by sand and silty substrates mixed with woody debris and aquatic vegetation. Those located near the source of a stream are often part of a *riverine headwater complex* and occur in conjunction with depressions and slopes. These streams within the wetland usually possess a deep, narrow, and sinuous channel, where water depth is much greater and dissolved oxygen concentration, while still adequate, is usually lower than that found in higher gradient streams. Streams with large amounts of organic (allochthonous) input may create darker waters due to dissolved organic material (DOM). Although these streams may be low in diversity, they often contain taxa uncommon elsewhere: Molannidae: *Molanna*; Dipseudopsidae: *Phylocentropus*; Psychomyiidae: *Lype*; and Ephemeridae: *Litobrancha recurvata*. The floodplain is usually part of a larger wetland, receiving a mixture of groundwater and surface water flow. Substrates are quite variable but usually contain vegetation. These habitats are well connected to the stream channel and tend to share many of the same taxa listed above. Several Diptera taxa are also common here: Tipulidae: *Phalacrocera*, *Molophilus*, *Ormosia*; Ceratopogonidae: *Atrichopogon*, *Forcipomyia*, *Bezzia*, and *Ceratopogon*. The mayfly genera *Leptophlebia* (Leptophlebiidae) and *Eurylophella* (Ephemerellidae) also frequent these habitats (Laubscher and Conklin 2004).

Riverine lower perennial streams may contain pocket floodplains in unconfined reaches where the valley widens or may be part of a more expansive *floodplain complex*. In areas where the gradient is sufficient to create a pool/riffle regime, the instream habitat is dominated by mayflies, especially the families Ephemerellidae (*Ephemerella*, *Drunella*, *Serratella*) and Heptageniidae (*Stenonema*, *Epeorus*).

Stoneflies and caddisflies are also prevalent: Perlidae: *Acroneuria*, *Agnetina*, *Paragnetina*; Hydropsychidae: *Ceratopsyche*, *Cheumatopsyche*, *Hydropsyche*; Philopotamidae: *Dolophilodes*, *Chimarra*; and Rhyacophilidae: *Rhyacophila*. Because of the greater magnitude and frequency of flooding, aquatic habitats can be extensive, both within the margins of the main channel and farther out in the floodplain. Habitats receiving periodic influxes of water and nutrients consist mainly of inorganic substrate, and may contain higher dissolved oxygen concentrations and specific conductivities than headwater floodplains. During periods of flooding, these habitats can become lotic in nature and mimic the main channel. Floating aquatic vegetation may be common in backwater floodplain habitats and may support amphipods (Gammaridae: *Gammarus*; Crangonyctidae: *Crangonyx*; Talitridae: *Hyalella azteca*) and the isopod genus *Caecidotea* (Asellidae). The mayflies *Siphlonurus* (Siphlonuridae) and *Eurylophella* (Ephemerellidae) along with many limnephilid caddisflies (*Pycnopsyche*, *Limnephilus*, *Ironoquia*) can be quite abundant in the floodplain (Laubscher and Conklin 2004; Laubscher 2005).

Although *slope* wetlands often receive water from a combination of surface water and groundwater sources, they maintain the greatest potential for groundwater input (Cole et al. 1997). They often occur alongside headwater streams and may be characterized by a mix of shrubs and emergent vegetation. Since water moves vertically and laterally across a slope's surface, these aquatic areas are often in the form of channels (somewhat reminiscent of intermittent stream channels), but also contain ephemeral pool habitats as well. Water depth varies, but is normally shallow. Substrate is often a mucky mix of organic material, gravel, sand, and silt. Habitats for aquatic macroinvertebrates tend to be intermediate in characterization between riparian depression and floodplain habitats. Several taxa, however, appear to prefer slope wetlands. The caddisflies *Frenesia* (Limnephilidae) and *Oligostomis* (Phryganeidae) as well as the stonefly *Soyedina* (Nemouridae) were often seen in abundance at slope wetlands, yet occurred rarely in other wetland types. Various cranefly genera (Tipulidae) may also be extremely abundant in these habitats: *Pseudolimnophila*, *Limnophila*, *Molophilus*, and *Pilaria* (Laubscher and Conklin 2004).

Macroinvertebrate community types have been described primarily for *temporary depressions* (isolated or vernal pools) and *permanent depressions* (riparian depressions). Generally, depressions exhibit more lentic conditions, lower dissolved oxygen levels, higher water temperatures, and substrate predominantly comprised of accumulated organic matter. Consequently, macroinvertebrate taxa in these habitats are well adapted to tolerate low oxygen conditions and are typically among the following class/orders: Diptera, Coleoptera, Odonata, Trichoptera, Hemiptera, Oligochaeta, and Gastropoda. Conklin (2003) investigated differences in both environmental and macroinvertebrate communities among three types of reference-standard depressional wetlands: unglaciated isolated depressions, unglaciated riparian depressions, and glaciated riparian depressions. Canonical correspondence analysis of macroinvertebrate abundance data collected from each of these wetland types suggested that reference-standard riparian depressions generally had shallower water depths, intermediate to higher levels of dissolved oxygen levels, and higher levels of specific conductivity than reference-standard isolated depressions (unglaciated). Overall, water source and water permanence were considered to be important factors in influencing both environmental conditions and community composition. Isolated depressions, being predominantly supported by surface water, were typically more ephemeral than riparian depressions fed by groundwater. This was reflected by the community. Isolated depressions contained more taxa exhibiting life history strategies and physiological and behavioral adaptations to survive or avoid drought conditions: Libellulidae: *Sympetrum*; Lestidae: *Lestes*; Dytiscidae: *Acilius*, *Dytiscus*, *Rhantus*; Hydrophilidae: *Hydrochus*, *Tropisternus*; Chaoboridae: *Chaoborus*; Culicidae: *Aedes*; Notonectidae: *Notonecta*; Limnephilidae: *Limnephilus*. Macroinvertebrate inhabitants of more permanent depressions typically lacked drought resistance and dispersal capabilities and/or required at least 1 year for larval development (i.e., univoltine): Gammaridae, Asellidae, Corydalidae, Sialidae, and Tabanidae.

10.3.2 Macroinvertebrate Bioassessments in HGM Wetlands

10.3.2.1 Indices of Community Integrity

Nearly two decades ago, the Penn State Cooperative Wetlands Center (now Riparia) began developing wetland biological assessments using aquatic macroinvertebrates. Bennett (1999) developed a Wetland Invertebrate Community Index (ICI) for the Ridge and Valley Province that could be applied to headwater streams, riparian depressions, and slope wetlands. This work was expanded to include other HGM wetland types and a larger geographic region and resulted in ICIs for isolated depressions, riparian depressions, slopes, and riverine wetlands (Conklin 2003; Laubscher and Conklin 2004). Table 10.1 displays the proposed metrics chosen for each ICI. It is important to note that, although I've tried to adhere to the latest HGM classification throughout this chapter, it is more appropriate in this section to refer to the previous terminology that was used when the ICIs were developed. Thus, "isolated" and "riparian" depressions correspond to "temporary" and "permanent" depressions, respectively, "slopes" refer to both "stratigraphic" and "topographic" slope wetlands (although the latter were more prevalent), "headwater floodplains" and "mainstem floodplains" refer to "upper perennial" and "lower perennial" riverine wetlands, respectively. Headwater and floodplain complexes were not defined as a separate classification at this time; however, the vast majority of the low gradient headwater floodplain and mainstem floodplain sites where the data was collected would correspond to these complexes. Following are the general conclusions regarding macroinvertebrate community change across a human disturbance gradient for these HGM wetland types.

Isolated Depressions (Unglaciated)

In general specific conductivity and total phosphorus levels, and percent herbaceous vegetation surrounding the site increase with anthropogenic disturbance in isolated

Table 10.1 Metrics selected for inclusion in Riparia's Invertebrate Community Indices (ICIs) for different HGM wetland types

					Riverine HW				Riverine mainstem	
		Isolated	Riparian		High gradient		Low gradient			
Metric category	Metric	depression	depression	Slope	Stream	FP	Stream	FP	Stream	FP
Richness measures	Class/order richness	X		X			X	X		
	Taxa richness		O	X/O	X		X/O	X	X	
	EPT taxa								X	
	EPTO taxa			X	X					
	EPTMO taxa		O	O			O	X		X
	Odonata taxa			X						X
	Plecoptera taxa			X	X	X			X	
	Trichoptera taxa			X		X			X	
	Coleoptera taxa								X	
	Dytiscidae + Hydrophilidae								X	
	Noninsect taxa				X					X
	Reference taxa					X	X	X		X
	Crustacea + Mollusca					X				
Composition measures	% OMT taxa	X								
	% TMP taxa		X							
	RA of EPTM				X					
	RA of Chaoboridae	X								
	% Hydrophilidae taxa		X							
	RA of Tipulidae		X							
	% Dominance top taxon		O	O			O			
	RA Trichoptera					X	X			

(continued)

Table 10.1 (continued)

Metric category	Metric	Isolated depression	Riparian depression	Slope	Riverine HW High gradient Stream	Riverine HW High gradient FP	Riverine HW Low gradient Stream	Riverine HW Low gradient FP	Riverine mainstem Stream	Riverine mainstem FP
Trophic measures	% Predator taxa	X								
	% Shredder taxa	X								
	Collector taxa richness		X							
	RA of predator + shredder taxa	X	X							
	Predator taxa			X	X				X	
	Shredder taxa			X		X				X
	Collector-filterer taxa								X	
	% Predators		O	O	X		X/O	X	X	
	% Shredders				X					X
	% Collector-gatherers								X	
	% Omnivores						X	X		X
Tolerance measures	Intolerant taxa			X	X	X		X	X	X
	% Intolerant individuals				X				X	
	Tolerant taxa				X					
	% Tolerant individuals						X			X
	Modified Hilsenhoff Biotic Index								X	X
	Elmidae Biotic Index				X					
	Hydropsychidae Biotic Index				X					
Habitat measures	Lotic taxa					X				X
Indicator taxa	Presence of *Sialis*		O	X/O			X/O			
	Presence of *Leuctra*		O	X/O			O			
	Presence of Corixidae						O			
	Presence of Nemouridae		O	O			O			
	Presence of Hydrophilidae		O	O						

X = HGM class ICI (Laubscher and Conklin 2004); O = Wetland ICI headwater stream or ephemeral pool metric (Bennett 1999). The following abbreviations apply: *E* Ephemeroptera; *P* Plecoptera; *T* Trichoptera; *O* Odonata; *M* Megaloptera; *RA* relative abundance depressions, while percent forest decreases (Conklin 2003). Differences in macroinvertebrate community measures (e.g., richness and diversity measures) selected as ICI metrics were only apparent between reference standard and severely impacted sites (Conklin 2003). Severely disturbed sites contained higher relative abundances of Oligochaeta (aquatic worms) and higher percentages of Coleoptera and Diptera taxa, while Hydrachnida, Odonata, Trichoptera, and shredder taxa were absent but common in reference-standard sites. The relative abundance and diversity of predator taxa also declined. These community differences were not apparent between moderately disturbed sites and the reference standard, suggesting the threshold of community change exists farther along the disturbance gradient (i.e., scores >65 on a scale of 0–100, with 0 being least disturbed). This is probably due to the fact that taxa in these ephemeral habitats are already adapted to harsh conditions and, thus, will not be as sensitive to change from anthropogenic disturbance (Conklin 2003). Moderately disturbed sites did differ from reference-standard sites with respect to the mollusk community: both diversity and relative abundance of Mollusca were higher in moderately disturbed sites. However, moderately disturbed sites typically had more calcareous bedrock than reference-standard sites, which could also explain the higher abundance of mollusks (i.e., these taxa require higher concentrations of $CaCO_3$ for shell formation). Another caveat to keep in mind when interpreting these results is that they are based on only one severely disturbed site, making it nearly impossible to distinguish between site-specific differences and those attributed to human disturbance. Therefore, the author recommended that these results be considered preliminary and more information should be collected from isolated depressions across an anthropogenic disturbance gradient (Conklin 2003).

Riparian Depressions (Unglaciated)

Riparian depressions subjected to increasing levels of anthropogenic disturbance typically have higher total phosphorus levels, warmer water temperatures, and less forest cover than reference-standard depressions (Conklin 2003). These environmental changes were reflected in the macroinvertebrate community. Relative abundances of Trichoptera, Megaloptera, and Plecoptera taxa declined in moderately and severely disturbed sites, as did relative abundance of crane flies (Tipulidae). In contrast, Dytiscidae and Hydrophilidae taxa as well as Mollusca taxa displayed positive trends with increasing human disturbance, possibly due to increases in primary productivity resulting from a combination of both nutrient enrichment and reduced canopy cover (Moore et al. 1993; Conklin 2003). Like the isolated depressions, riparian depressions showed negative trends in percentages of both predator and shredder taxa with human disturbance (Conklin 2003). Unlike the isolated depressions, however, macroinvertebrate communities in riparian depressions appeared to respond earlier to anthropogenic disturbance (i.e., differentiated between moderately disturbed and reference-standard sites).

Slopes

Responses to anthropogenic disturbance in slope wetlands were similar to those displayed by communities in riparian depressions. In both wetland types, Trichoptera, Plecoptera, predators and shredders were important components of the reference-standard community. Unlike the depressions, which consisted primarily of composition and trophic measures, the slope ICI contained several richness measures that displayed significant dose–response relationships, as did a tolerance measure and the indicator taxa *Sialis* and *Leuctra* (Laubscher and Conklin 2004). Both the riparian depression and slope ICIs appeared to respond at intermediate levels of disturbance, although slope wetland communities may be more sensitive to disturbance. The majority of slope metric scores plummeted at disturbance scores between 40 and 50 and remained low thereafter (on a scale of 0–100 with 0 being least disturbed).

Headwater and Mainstem Floodplains (Riverine)

Hydrologic modification and sedimentation are two of the most prevalent types of impacts to riverine systems and typically result in complete loss of aquatic habitat outside the main channel. Thus, it is worth noting that the range of disturbance levels in riverine wetlands is expected (and was found) to be less than that pertaining to streams in general, since severely impacted streams contain no floodplain habitat. Consequently, a severely impacted floodplain in the ICI would correspond to a moderately impacted stream. This is one of the advantages of incorporating floodplain metrics into stream bioassessments (i.e., the addition of floodplain information helps differentiate between moderately impacted stream reaches).

The Riverine Invertebrate Community Indices (RICIs) each consisted of a suite of stream metrics and a separate suite of floodplain metrics. Stream metrics applied to riffle habitats (except in the low gradient headwater RICI) and were quite similar to standard metrics used in many biotic indices (e.g., EPT taxa richness). The exceptions were two tolerance metrics developed for headwater streams, which capitalized on the differences in relative abundances between intolerant and more tolerant genera of the Elmidae (the Elmidae Biotic Index) and Hydropsychidae (the Hydropsychidae Biotic Index) families. Both of these were computed in a similar manner to the well-known Hilsenhoff Biotic Index; both demonstrated a strong dose–response relationship to disturbance.

Floodplain metrics were also similar to other wetland ICI metrics: Trichoptera, Odonata, Plecoptera taxa, and proportions of predators and shredders all decreased with increasing disturbance; crustaceans and Mollusca taxa increased with disturbance (Conklin 2003; Laubscher and Conklin 2004). Like the slope ICI, tolerance and richness measures also demonstrated strong dose–response relationships with human disturbance. Unlike riparian depressions, however, Dytiscidae and Hydrophilidae taxa displayed negative trends with anthropogenic disturbance.

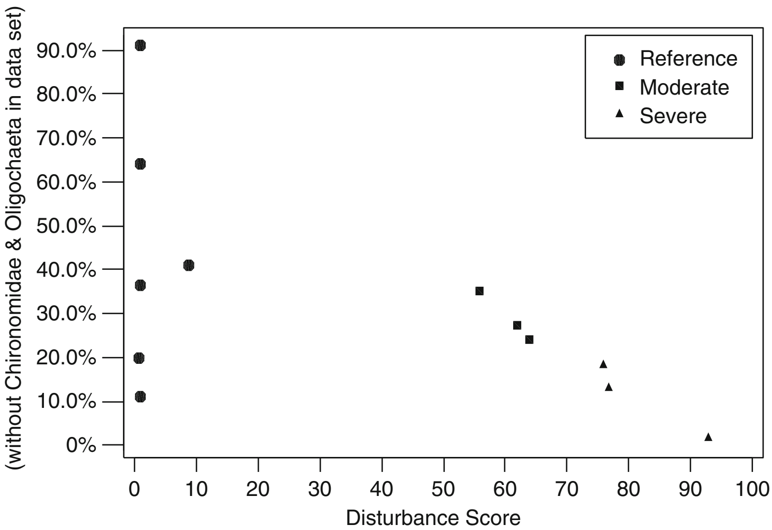

Fig. 10.1 Stressor-response plot of the attribute "relative abundance of predator + shredder taxa" selected as a metric in the unglaciated riparian depression ICI (Conklin 2003)

10.3.2.2 Lessons Learned: Disadvantages of Sampling by HGM Types

While the wetland ICIs definitely revealed a gradient of environmental change, it was apparent that an a priori HGM classification was not sufficient for partitioning natural variation in macroinvertebrate communities. Far too much variation remained unexplained, making it difficult to assess responses to anthropogenic disturbance. One common denominator across all HGM wetland types was the large range of variability within the metric scores for reference-standard sites, which made it difficult to detect impacts from human disturbance. Figure 10.1 provides a typical example with the metric "Relative Abundance of Predator + Shredder Taxa" used in the Unglaciated Riparian Depression ICI (Conklin 2003). Although the average relative abundance of predator and shredder taxa declined as disturbance scores increased, some reference-standard sites contained relative abundances that were equal to or less than those found in moderately and severely disturbed sites. To reemphasize, this variation in the reference standard was common to multiple metrics across multiple HGM wetland types. What could be the reason for all of this variation within the reference standard?

There are several possible explanations for this. First, the HGM classification considers one spatial scale, while macroinvertebrate assemblages respond to multiple scales (Parsons et al. 2004). While HGM may account for responses to different water sources (e.g., surface water vs. ground water habitats), it does not consider responses at smaller spatial scales. Macroinvertebrates respond primarily to microhabitat changes in substrate, water velocity and depth, temperature, water chemistry, and hydroperiod. Not only do these microhabitat patterns differ within an HGM type, they often repeat in different HGM wetland types. This in turn may create similar community associations in different wetlands. Bennett (1999) reached a similar conclusion when constructing her Wetland ICI. Through an ANOVA of ordination scores for stream, pool, and soil communities collected from sites of different wetland types and disturbance levels, she found that the wetland HGM classification did not distinguish between communities collected in aquatic habitats from slopes or depressions and concluded that "stratification of these wetland types is unnecessary for invertebrate studies" (Bennett 1999). Thus, in her wetland ICI she referred to these habitats collectively as "ephemeral pools."

Second, both HGM wetland types and macroinvertebrate community types rarely conform to a specific class. Rather, they often exist along a gradient of change or even as clusters of particular types, making it difficult to conduct assessments within one HGM subclass (hence the addition of the headwater and floodplain complexes). This makes classification of macroinvertebrate communities extremely difficult, especially since these organisms are, by their very nature, opportunistic and often occur in multiple hydrologic settings.

Third, this close proximity and interconnectedness of HGM wetland types is especially prevalent in natural landscapes that have been subjected to minimal anthropogenic disturbances over long time periods. This along with predictable seasonal or annual cycles of hydrologic connectivity and hydroperiod has enabled many species to evolve and capitalize on these additional habitats for optimizing food resources, emergence and oviposition sites, larval growth, and protection from both predators as well as harsh environmental conditions. Thus, the same organism may exploit more than one HGM type during its life cycle. Moreover, similar species traits or combinations of traits are advantageous in multiple HGM wetland types, which also may cause substantial overlap and obscure community responses. These factors culminate in the conclusion that the potential for biological diversity and habitat heterogeneity explodes in natural landscapes. In fact, this heterogeneity and natural variation may in and of itself represent the most important attribute defining the reference standard. In this context, classifying by wetland habitat type, rather than HGM subclass (or in conjunction with HGM), may provide more ecologically relevant assessments of both biological and habitat condition.

These patterns of community structure are illustrated in the following ordination plot (Fig. 10.2), which represents a compilation of the macroinvertebrate relative abundance data collected from reference-standard wetlands of various HGM subclasses. Riverine data from headwater and mainstem sites were separated into stream (riffle or nonriffle) and floodplain habitat. I've retained the earlier version of the HGM subclassification in this figure for two reasons: (1) to illustrate the extreme overlap in communities from slopes, riparian depressions, and headwater floodplains, which supports the decision to add complexes as an additional wetland class; and (2) to emphasize that data were collected using this a priori wetland classification, thus efforts were made to stay within a particular HGM subclass (i.e., floodplain or slope or riparian depression) leading us to conclude that there is probably more community overlap between subclasses than shown in Fig. 10.2.

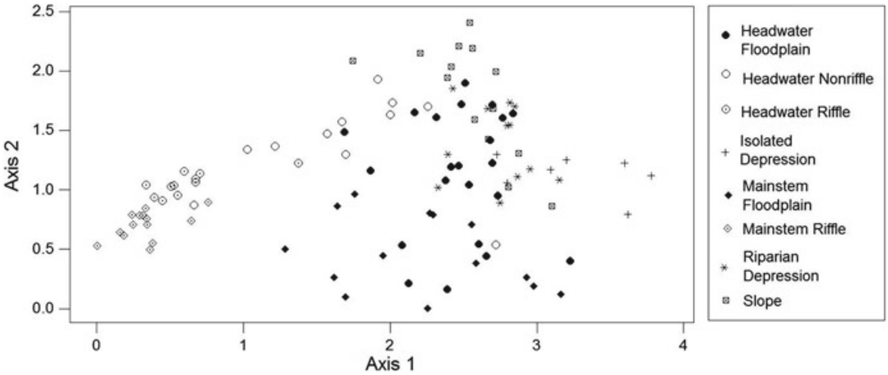

Fig. 10.2 Plot of axis 1 and axis 2 scores obtained from detrended correspondence analysis on the macroinvertebrate relative abundance data collected from various wetland types (all reference standard). The wetland classifications listed in the legend correspond to the following subclasses: isolated depression = temporary depression; riparian depression = permanent depression; headwater floodplain = riverine upper perennial or headwater complex; mainstem floodplain = riverine floodplain complex or riverine upper/lower perennial; headwater stream nonriffle = riverine headwater complex; headwater riffle = riverine upper perennial; mainstem riffle = riverine upper perennial (>3rd order)

The first ordination axis represents the classic stream/wetland gradient from permanent lotic conditions (riffles) to temporary lentic conditions (isolated depressions). Both headwater and mainstem riffle habitats contain distinct macroinvertebrate communities, while inhabitants of smaller, low gradient headwater stream habitats are more similar to other wetland habitats found within a headwater complex (Fig. 10.2). Depressions also provide habitat for different macroinvertebrate types, but communities in riparian depressions are more similar to slope and floodplain habitats, while those in temporary depressions consist exclusively of taxa adapted to ephemeral conditions. These latter results were also noted by Conklin (2003) who concluded that water permanence was a major factor in differentiating between macroinvertebrate assemblages of isolated and riparian depressions. The second ordination axis appears to suggest a continuous gradient of community change from mainstem floodplains to a mix of headwater wetland types to slope wetlands. Although there are obvious differences in macroinvertebrate community structure along these gradients, clarification by wetland subclass is probably related more to rare taxa with strong associations (a.k.a. high validities) or to taxa with strong significance (a.k.a. frequent occurrences) and abundances for a particular wetland type. Figure 10.3 displays the approximate location of these wetland subclasses (using current HGM terminology) along each axis and identifies specific taxa most associated with each subclass. It is important to understand that, while rare taxa demonstrate a strong validity with a particular wetland type (i.e., they occur primarily or exclusively within one subclass), they typically do not have a strong significance of occurrence in these types (i.e., they occur at a limited number of sites). Conversely, taxa that show a high significance in a particular wetland type may also

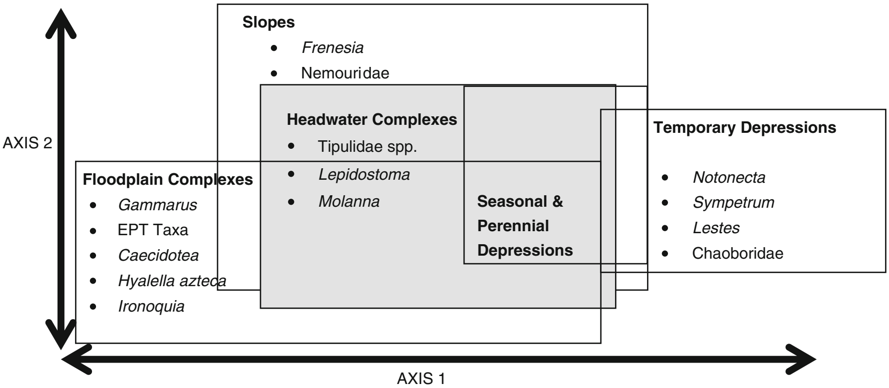

Fig. 10.3 Depiction of wetland HGM subclasses in ordination space as determined by macroinvertebrate relative abundance data. Note the significant overlap between community types, especially between seasonal/perennial depressions, slopes, and complexes

be present in other wetland types (i.e., low validity). Consequently, these taxa may not represent good indicators of particular wetland types.

In summary, patterns in macroinvertebrate community structure relate to the HGM wetland subclassification, but much of the variation appears to occur within these classes at smaller spatial scales and needs to be evaluated to understand and describe patterns of aquatic macroinvertebrate community structure. The next section presents a case study, in which I developed a more detailed habitat classification and explored patterns of community composition within the riverine headwater and floodplain complexes. The main goal was to develop a more ecologically relevant classification scheme focusing on the most prevalent and diverse aquatic macroinvertebrate habitats and then apply this classification across a human disturbance gradient to ascertain the advantages of incorporating this information into wetland bioassessments.

10.4 Understanding Pattern and Process in Riverine Systems Through Macroinvertebrate Ecology: A Case Study in Headwater and Floodplain Complexes

10.4.1 Riverine Hierarchical Habitat Classification

Riverine systems result from the dynamic interactions of hydrological, geomorphological, and biological processes acting within multiple dimensions of space (longitudinal, lateral, and vertical) and time (Stanford and Ward 1993; Petts and Amoros 1996; Poff et al. 1997; Ward 1989; Richards et al. 2002; Robinson et al. 2002; Ward et al. 2002; Naiman et al. 2005; Thorp et al. 2008). Petts and Amoros (1996) refer to these systems as "fluvial hydrosystems" where hydrological and geomorphological processes determine the types of habitat present, as well as the strength, duration, and frequency of their connectivity. Included among the key features they list as crucial to understanding these systems are (1) attention must be focused on the entire corridor, which comprises the main channel (or channels), adjacent floodplains with wetland and terrestrial habitats, and the underlying alluvial aquifer; (2) spatial patterns of abiotic and biotic components need to be described as longitudinal, lateral, vertical, and temporal (successional) gradients linked by energy and material fluxes; and (3) biological assemblages are determined primarily through autoecological processes (i.e., by environmental gradients) that are modified by biological processes. Regarding macroinvertebrates, population distributions depend on environmental factors collected at multiple scales (Poff et al. 1997; Malmqvist 2002; Richards et al. 2002). Consideration of these key factors requires a hierarchical classification scheme that incorporates all the elements of a riverine system including fluxes and exchange pathways. This also enables us to collect information at multiple levels within the hierarchy to identify the most ecologically relevant scale(s) for evaluating community (and system) responses. In riverine systems, this is often considered to be the reach scale, since flooding and channel pattern dynamics represent important controls on both habitat and biology (Richards et al. 2002).

Although multiple classifications have been proposed (e.g., Frissel et al. 1986; Petts and Amoros 1996; Thorp et al. 2008), we've tried to follow the general consensus. Figure 10.4 shows the proposed classification of Riparia's riverine assessments. Generally, I followed existing classification schemes (cited above), but added one level, the *aquatic ecological set*, to differentiate between habitats structured by flow, flood, and groundwater pulses that occur between the reach and functional set scales, but have been shown to contain different macroinvertebrate assemblages (Wardrop et al. 2012). Riparia collects biological and habitat data at multiple scales, but typically focuses on the habitat unit or microhabitat levels and scales up to look for patterns or responses at larger scales. Sites primarily correspond to the reach scale.

As mentioned earlier, spatial patterns of abiotic and biotic components need to be described as longitudinal, lateral, vertical, and temporal (successional) gradients linked by energy and material fluxes (Petts and Amoros 1996). I have just described the natural flow regime, which is the primary factor influencing aquatic macroinvertebrate community structure (Ward 1989; Poff et al. 1997; Bunn and Arthington 2002; Ward et al. 2002; Whiles and Goldowitz 2005; Williams 2006; Poff and Zimmerman 2009; Chinnayakanahalli et al. 2011). The extent and influence of the flow regime on structuring aquatic habitats is not constant, however, and can best be identified through a characterization of the reach type. In general, a reach can be defined by its degree of confinement with unconfined floodplain reaches representing areas along the riverine corridor where hydrologic exchange along these four dimensions is maximized (Ward 1989; Ward et al. 2002) (Fig. 10.5). As a result, the potential for ecological diversity is much greater in unconfined reaches than in more confined reaches with narrow riparian zones. Thus, headwater and floodplain complexes can be defined through a combination of reach type (unconstrained) and the

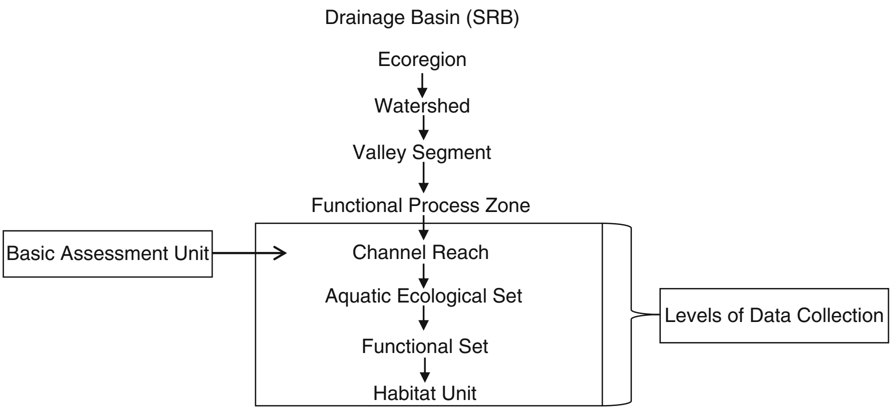

Fig. 10.4 Hierarchical framework used in Riparia's riverine assessments. Data are collected at multiple levels. Headwater complex and floodplain complex HGM subclasses (the basic assessment unit) correspond primarily to the channel reach scales. Adapted from Frissel et al. (1986), Petts and Amoros (1996), and Thorp et al. (2008)

dominant HGM factors associated with the reach or functional process zone. Headwater complexes are typically located in the production zone and consist mostly of habitat supported by groundwater. They often resemble a small low gradient stream flowing through a wetland matrix. Floodplain complexes are primarily found farther down the river continuum in the transfer zone and storage zones and contain a mix of surface and groundwater habitats structured primarily by the stream's flow regime (Fig. 10.5).

10.4.2 *Defining the Reference Standard for Riverine Macroinvertebrate Habitats*

According to Junk et al. (1989), hydrologists consider a river and its floodplain as one unit since they are inseparable with respect to their water, sediment, and organic budgets. In their extension of the flood pulse concept to include temperate regions, Tockner et al. (2000) identified "stochastic events," such as periodic flooding below bankfull (i.e., the flow pulse), as an important causal factor of temperate floodplain biodiversity. These authors introduced two very important terms to riverine ecology, *flood pulse* and *flow pulse*. The former refers to above bankfull flood events that inundate the floodplain; the latter refers to below bankfull events that inundate an area which we have termed the *active zone*. This area is represented by the shaded region in Fig. 10.6 and corresponds to locations within the banks of the main channel or in well-connected side channels or marginal areas that are connected during these flow pulses. It includes well-known instream habitats, such as riffles and runs, and lesser known habitats adjacent to or along the margins of the main channel.

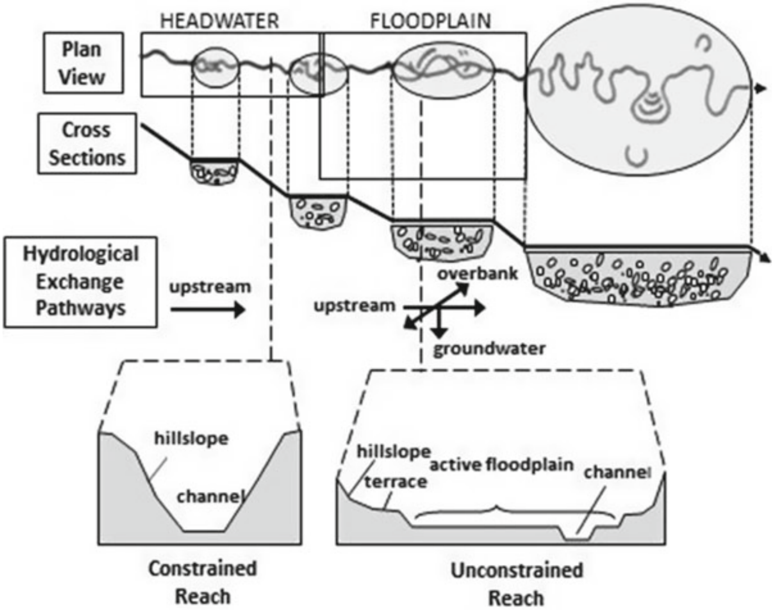

Fig. 10.5 Conceptualization of a riverine corridor from headwater to mouth showing alternating sequences of constrained and unconstrained floodplain reaches with predominant hydrological exchange pathways indicated for longitudinal (*horizontal arrows*), lateral (*oblique arrow*), and vertical (*vertical arrow*) dimensions (modified from Ward et al. 2002). The *circles* or *beads* represent unconstrained reaches where headwater or floodplain complexes occur. The black boxes indicate the range along the river continuum within which Riparia's macroinvertebrate data was collected from headwater and floodplain complexes; these regions correspond to the production and transfer zones of a drainage basin (Schumm 1977)

Since the term *floodplain* can have many connotations, some clarification is in order. Following Armantrout's aquatic habitat inventory (1998), *floodplain* refers to an "area adjoining a water body that becomes inundated during periods of overbank flooding…" when discussed in the context of aquatic habitat or aquatic ecological set. All other references to the term refer to any area that floods and is composed (at least to a certain extent) of deposited alluvium. The take-home message here is that the active zone floods far more frequently, but with less intensity than the floodplain, resulting in two major types of "floodplain" habitat. In addition to these fluvial expansion-and-contraction cycles, local groundwater regimes (e.g., recharge areas, hyporheic zone) also create aquatic habitat. This combination creates a patchwork of biodiversity within the floodplain and riparian zone in terms of habitats, species richness, and floodplain successional stages (Junk et al. 1989; Ward et al. 1999, 2001; Tockner et al. 2000; Amoros and Bornette 2002; Arscott et al. 2002; Richards et al. 2002).

Riparia's basic approach to understanding riverine ecosystems is one of observation and applied science. In this case, I observed this diversity in macroinvertebrate

Fig. 10.6 Plan view of a riverine reach depicting the main channel, side and secondary channels, aquatic floodplain, and wetland habitats throughout the riparian zone. The active zone or flow pulse area represents the shaded region; areas outside this region represent the floodplain aquatic ecological set, which receive flood water only during above bankfull events (flood pulses)

communities and related it to aquatic habitat types by collecting biotic and abiotic information at multiple reference-standard riverine sites throughout the MAR and looking for significant patterns or responses in the macroinvertebrate community data that could be explained either by the environmental information collected at the site and/or by known adaptations and life history traits associated with the taxa. Following the advice of Petts and Amoros (1996), I focused attention on the entire corridor, sampling any aquatic habitat within the reach, including riffles, pools, or runs within the main channel or channels, pools, seeps, and other aquatic areas within the riparian zone. Wardrop et al. (2012) and Yetter (unpublished data) conducted multiple ordination procedures and hierarchical cluster analysis of plot-level macroinvertebrate relative abundance data from reference-standard headwater and floodplain complexes to determine riverine macroinvertebrate habitat types and the spatial scale at which they are best defined. The unconstrained multivariate analysis was then followed up with correlations with environmental plot data and information obtained from plot photographs and site maps to help describe the structure of the different habitat types. Through a combination of macroinvertebrate life history traits, habitat, and hydrological information, links between aquatic macroinvertebrate groups, habitats, and underlying hydrologic processes were predicted.

Results revealed six major habitat types. The first two are instream baseflow habitats. The third and fourth types correspond to varying levels of surface water flooding within the active zone or floodplain and occurred mostly in floodplain complexes characterized by pool/riffle regimes (i.e., slightly higher gradient than defined by the subclass). The final two types were primarily related to seasonal groundwater recharge or both surface- and groundwater ephemeral habitats and were mostly associated with headwater complexes. Based on these results, further analysis was conducted separately for floodplain and headwater complexes. The following descriptions apply to headwater and floodplain complexes representing primarily the ends of the gradient from flood-dominated to groundwater-dominated systems and also reflect the river continuum from headwaters to lower reaches. Keep in mind, therefore, that many complexes in both headwaters and floodplains may be characterized by a more transitional mix with surface and groundwater habitats equally represented. I've found this to be especially true in second through fourth order stream systems (Laubscher and Conklin 2004; Laubscher 2005; Wardrop et al. 2012).

10.4.2.1 Floodplain Complexes

Minimally impacted (a.k.a. forested) floodplain complexes generally consist of four main habitat types for macroinvertebrates: (1) riffle habitats and (2) nonriffle baseflow habitats (both in the main channel of the active zone); (3) flow pulse habitats in the active zone; and (4) aquatic habitats in the floodplain (or flood pulse habitats). Some may also contain depression or slope habitat, which is discussed subsequently in headwater complexes. Figure 10.7 displays these major habitat types determined from hierarchical cluster analysis of the macroinvertebrate relative abundance data collected from multiple floodplain complexes. The main purpose of this figure is to place the results of the habitat classification within the context of the riverine hierarchy to demonstrate that macroinvertebrates respond to habitat changes at multiple scales. The majority of macroinvertebrate habitat types are found within the active zone and are defined at both the habitat unit (riffles and other baseflow habitats) and functional unit (flow pulse habitats) scales (Fig. 10.7). Macroinvertebrate relative abundance data did not differentiate between habitat types within the floodplain (aquatic ecological set scale). However, taxa associated with floodplain habitats were quite diverse and varied between sites, suggesting these habitats may have been too heterogeneous and lacked a consistent pattern. This is most likely due to these habitats being subjected to site-level differences in the riparian zone (e.g., soil type, vegetation, etc.).

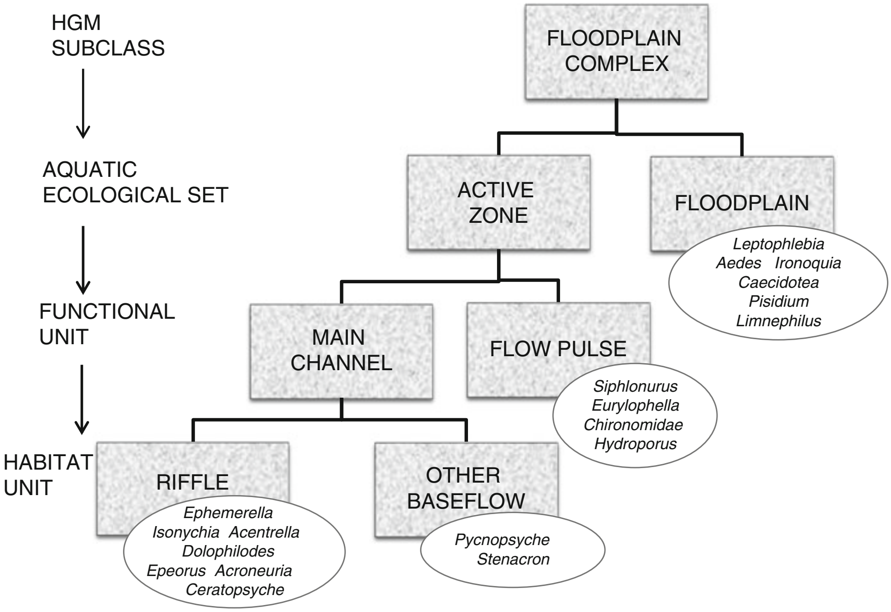

Fig. 10.7 Organizational hierarchy of forested floodplain complex habitat types and examples of macroinvertebrate taxa associated with each

This complex mix of surface water habitats appears to be structured primarily by varying hydroperiods and connectivity with the main channel. Riffles are the most well-known macroinvertebrate habitat type and contained the most distinctive macroinvertebrate community, while other baseflow habitats (primarily runs and marginal pools) were mostly frequented by detritivores preferring slower velocities (e.g., *Pycnopsyche*, *Stenacron*). Both of these areas are located entirely within the main banks of the channel and are inundated during periods of baseflow (i.e., permanent hydroperiods). Not surprisingly the community consists mostly of stream insects that require permanent lotic environments and lack adaptions to drought and high temperatures.

Flow pulse habitats are located in areas of the active zone that are typically connected to the main channel during below bankfull flood events (flow pulses). These are often in the form of side or secondary channels and are characterized by intermittent hydroperiods, seasonal connectivity to the main channel, high specific conductivity, and consist primarily of bare mineral substrate with pockets of silt or muck and deposits of woody debris. Macroinvertebrates most abundant in this habitat type include the mayflies *Siphlonurus* and *Eurylophella*, midges, and the beetle *Hydroporus*. The mayflies are usually found in the main channel during baseflow periods but migrate outward during the winter and spring when these additional habitats become available (Smock 1994). Such an adaptation most likely arose from increased population success due to more food availability, higher temperatures, and less predation than main channel habitats (Fig. 10.7). Many of the insect larvae found within this habitat type complete their larval growth by late spring and emerge as terrestrial adults before the habitats dry (Ward 1992; Laubscher 2005).

Habitats in the floodplain are often found in cut-off (para-) and isolated (paleo-) channels with semipermanent to permanent hydroperiods. Some may be maintained by a groundwater connection due largely to scouring from large flood events that either connects the bottom of the floodplain channel with groundwater seepage or results in deep pools that do not completely draw down and become anoxic. These habitats may support emergent, submergent, and floating vegetation. Substrate usually consists of fine mineral sediments (e.g., silt) and large amounts of particulate organic matter. As mentioned above, floodplains may contain a diverse array of taxa. Groundwater recharge areas often support the isopod *Caecidotea* and amphipods (*H. azteca*, *Crangonyx*, and *Gammarus*). Surface water (flood pulse) habitats, which are often characterized by intermittent hydroperiods, consist mostly of taxa with either life cycle adaptations (e.g., aestivation—*Ironoquia*, *Limnephilus*) or physiological and morphological adaptations (e.g., *Pisidium*) to drought (Ward 1992; Laubscher 2005; Williams 2006) (Fig. 10.7).

10.4.2.2 Headwater Complexes

Seasonal and temporary pool habitats are the most prevalent macroinvertebrate habitats in headwater complexes. The physical nature of the habitats is by appearance quite similar, but they seem to range from seasonal to intermittent and ephemeral hydroperiods and consist primarily of deposits of autogenic organic matter with mucky substrates often interspersed with sphagnum moss or allochthonous leaf litter. These habitats are not typically associated with channels but, rather, seem to rely on a vertical hydrological connection with groundwater sources or are merely ephemeral pools of surface water. Regardless of water source, hydroperiod, substrate, and riparian vegetation are the most likely determinants of community structure in these habitats.

Figure 10.8 illustrates the structure of macroinvertebrate habitats from headwater complexes. It is unclear to which levels of the riverine hierarchy these habitat types belong, since the macroinvertebrate relative abundance data only revealed significant differences between riffle communities and all other stream/wetland habitats. However, I would not expect a close association between headwater complexes that are mostly dominated by groundwater-fed depression and slope wetlands and a riverine classification based largely on surface water flow regimes. Although not significant, I was able to differentiate four main types of microhabitats in the seasonal and temporary wetland portions of headwater complexes (note that nonriffle headwater stream habitats contained taxa similar to those collected from groundwater seeps).

A wide variety of taxa frequent these habitats. The combination of physical habitat characteristics with taxonomic information, such as habitat requirements and life cycle strategies, enabled us to breakdown these seasonal and temporary pools into

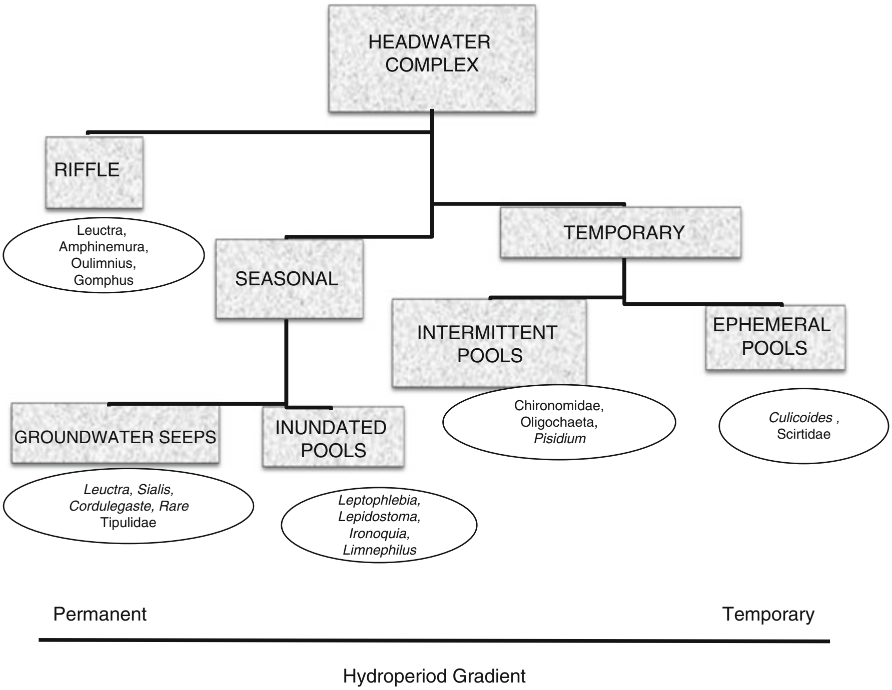

Fig. 10.8 Structure of macroinvertebrate habitat types in headwater complexes. Generally these habitat types follow a hydroperiod gradient from permanent to seasonal to temporary environments

four basic microhabitats: (1) seasonal habitats with sphagnum moss that typically occurred near groundwater sources, especially near streams that flowed through wetlands (riparian depressions) or had a lot of nearby or surrounding wetlands, and support a variety of headwater stream taxa (*Leuctra*, *Sialis*, *Cordulegaster*, etc.) and many uncommon dipterans (*Phalacrocera*, *Molophilus*, *Gonomyia*, *Ormosia*, *Forcipomyia*, *Pilaria*, *Ceratopogon*, etc.); (2) temporary habitats with lots of decaying organic matter (both woody and leaf litter), similar to that found in tree holes and containing an abundance of tree hole taxa (Scirtidae and *Culicoides*); (3) inundated habitats containing a lot of coarse organic matter and leaf litter that provide habitat for some mayflies (e.g., *Leptophlebia*, *Siphlonurus*) and cased caddisflies (*Lepidostoma*, *Ironoquia*, *Limnephilus*); and (4) small temporary pools characterized mostly by fine particulate organic matter and containing taxa with short life cycles or tolerant to desiccation and high temperature fluctuations—such as mosquitoes, chironomids, oligochaetes, fingernail clams, and snails (Fig. 10.8). Basically two types of taxa are present here: (1) those with r-selective strategies (short life cycle, rapid growth, etc.) that can complete their life cycle before habitats dry and (2) those with behavioral, physiological, or life cycle adaptations to surviving in temporary habitats (e.g., the Limnephilidae). Taxa displaying significantly higher numbers of individuals in temporary habitats compared to other floodplain types include fingernail clams (*Pisidium*), and various flies (*Chrysops*, Ephydridae, *Dicranota*, *Bittacomorpha*, and *Aedes*) (Fig. 10.8).

In summary, I was able to partition much of the variation within reference-standard riverine floodplain complexes through an exploratory analysis of the macroinvertebrate assemblages. Despite the fact that I combined data collected from multiple sites distributed across different regions of the MAR, I was still able to discern a definite and predictable spatial pattern within the habitat structure. This variation was not as easily explained in headwater complexes. Although I was able to describe different habitat types along a gradient of water permanence, the existence and spatial arrangements of these habitats were not consistent between sites.

One likely explanation is stream size. Smaller wetland-dominated headwater complexes (~0–1st order) exhibit a strong vertical connection and demonstrate the least structural pattern, with habitats often created through isolated stochastic events (e.g., tree fall) or small-scale groundwater controls on riparian vegetation. Larger stream reaches increasingly contain more surface-water structured habitats (lateral connection) that show a more consistent pattern in both macroinvertebrate and environmental structure. Thus, floodplain complexes may be more indicative of a top-down *trans-scale* process that influences patch structure and function and is influenced by catchment-wide processes (i.e., flooding), while headwater complexes are influenced more by site-specific factors (e.g., riparian vegetation communities) and demonstrate a more bottom-up *trans-scale* process to the habitat structure (Poole 2002). Both wetland types, however, are dependent on an optimal level of hydrologic connectivity.

10.4.3 The Importance of Riverine Hydrologic Connectivity to Biodiversity and Bioassessment

Macroinvertebrates are a highly diverse and abundant group; therefore, reducing environmental noise in macroinvertebrate studies is crucial for detecting biological/community responses. However, the coexistence of stream, floodplain, and wetland habitats within the stream continuum has enabled this remarkably adaptive group of organisms to evolve a wide variety of strategies for utilizing several different habitats during the course of their aquatic life cycles. Thus, the spatial distribution of these taxa depends not only on habitat condition, but also on their ability to move between and within habitat types (Stanford and Ward 1993; Malmqvist 2002; Ward et al. 2002; Wiens 2002).

Hydrological connectivity is necessary for riverine functions that involve exchange pathways or fluxes pertaining to the flow of energy, organic matter, or biota (Ward 1989; Bunn and Arthington 2002; Ward et al. 2002). It is created and maintained through flood and flow pulses, which control both the degree of connectivity for surface water habitats and the level of hydrarch succession within those habitats (Tockner et al. 2000; Ward and Tockner 2001; Arscott et al. 2002; Malmqvist

2002; Robinson et al. 2002; Wiens 2002). These pulses introduce primary productivity into the main channel from the riparian zone (Amoros and Bornette 2002; Church 2002). Connectivity between streams, floodplains, wetlands, and the subsurface is crucial for dispersal of organisms into more suitable habitats for feeding, mating, and refuge (Stanford and Ward 1993; Ward et al. 2002; Wiens 2002; Miyazono et al. 2010). For example, many taxa that utilize side or secondary channels in the active zone are primarily swimmers that lack the ability to crawl and require a surface water connection. The mayfly *Siphlonurus* occurs in main channel margins but prefers to migrate to flow pulse habitats to complete its larval development. These areas provide more suitable habitat for larval growth (e.g., warmer temperatures, more food, less predation) (Voshell 1982). For these taxa the main channel may serve as refuge during dry periods, rather than primary habitat (Laubscher 2005). A wide variety of taxa prefer less connected floodplain environments, including the caddisflies *Ironoquia*, *Limnephilus*, and *Oligostomis*, the mayfly *Leptophlebia*, and certain hydrophilid beetles. The limnephilid caddisfly *Ironoquia*, for example, aestivates in its fourth larval stage until dry conditions trigger the start of the pupal phase (Williams 1996). Others may utilize vertical (subsurface) connectivity between the surface of the floodplain and the hyporheic zone, which can be quite extensive. Stanford and Ward (1988) recorded the presence of hundreds of stoneflies in shallow groundwater wells located as far as 2 km from the main channel of the Flathead River, Montana. Species from the families Chloroperlidae, Capniidae, and Leuctridae have specific life history adaptations for occupying floodplain groundwater. They spend their entire larval period within the hyporheic zone and emerge from the main channel as terrestrial adults (Stanford and Ward 1993).

Through a qualitative analysis, Wardrop et al. (2012) ascertained preferences of particular macroinvertebrate taxa to differing levels of hydrologic connectivity and hydroperiod associated with habitats in floodplain complexes. Tables 10.2 and 10.3 display the connectivity and hydroperiod categories used to evaluate each habitat sampled. Each represents a gradient with 0 representing least connectivity and shortest hydroperiod and 7 and 4 representing the highest connectivity level and longest hydroperiod, respectively. Several taxa, including *Siphlonurus*, *Ironoquia*, *Limnephilus*, and other floodplain taxa, had higher mean relative abundances in habitats exhibiting intermediate levels of connectivity, as well as intermittent to semipermanent hydroperiods. These results support conclusions that intermediate levels of connectivity and temporary habitats are important for maintenance of biodiversity (Ward et al. 1999; Tockner et al. 2000; Williams 2006).

Riverine headwater complexes and riverine floodplain complexes represent areas along the river continuum that span the full range of both connectivity and hydroperiod gradients (Tables 10.2 and 10.3). They contain habitats ranging from highly ephemeral surface water pools to intermittently flooded cut-off channels to seasonally saturated seeps to permanently inundated habitats within the main channel. This habitat diversity or patchiness often occurs as a consequence of shifting water sources and flow pathways during the course of the riverine expansion and contraction cycle (Malard et al. 2000; Tockner et al. 2000). As a result, these complexes

Table 10.2 Codes used to define hydrological connectivity of macroinvertebrate sampling plots

Code	Hydrologic connectivity
0	Wetland with no apparent lateral (surface) or vertical (groundwater) connection to the stream
1	Wetland connected to stream through an outlet
2	Abandoned (paleo-) or high flood channel (or pooled depression) disconnected at both ends
3	Cut-off (para-) channel connected at one (usually downstream) end
4	Wetland abutting the stream with both lateral and vertical connections
5	Side or secondary channel connected to stream at both ends or marginal area disconnected during baseflow
6	Main channel high flow connection (a.k.a. usually dry during baseflow)
7	Main channel baseflow connection (primarily riffles, runs, and pools)

Table 10.3 Codes used to classify the hydroperiod of macroinvertebrate sampling plot

Code	Hydroperiod
0	Ephemeral—rarely flooded; receives water only occasionally and unpredictably
1	Episodic—fill occasionally; may hold water for months or years
2	Intermittent—receive water quite frequently and usually predictably
3	Semipermanent—inundated at least half of the year
4	Permanent—always inundated except in extremely rare long-term droughts

provide the most expansive array of aquatic macroinvertebrate habitats and may represent hotspots of biodiversity (Tockner et al. 2000; Tockner and Stanford 2002; Robinson et al. 2002; Ward et al. 2002; Naiman et al. 2005). This biodiversity is expressed at multiple scales (Whittaker 1972; Ward et al. 1999, 2002; Amoros and Bornette 2002; Richards et al. 2002). At the smallest scale (genetic diversity notwithstanding), alpha (α) diversity is expressed as the number of species (or unique taxa) in each unit. This unit may be a particular microhabitat or habitat unit (e.g., riffle) or may represent higher levels in the hierarchy (e.g., reach) depending on the circumstance. Beta diversity, or species turnover, is a measure of the proportion of habitats in which each species is present. A reach with high beta diversity, for example, would contain multiple habitat types that each support entirely different assemblages (a.k.a. no species/taxa in common). Gamma diversity is the cumulative number of species (taxa) in a region (e.g., reach or watershed) (Whittaker 1972; Amoros and Bornette 2002). Small headwater streams in the MAR often contain low numbers of taxa, yet many of these taxa are rare or even endemic to that particular stream. As a result, while individual headwater streams may have low alpha diversity (within stream) and share a similar suite of a few common taxa, the site fidelity of a large array of rare taxa contributes to high beta diversity (between streams), creating high amounts of gamma diversity at the watershed scale.

Two consistent factors vitally important for maintenance of biodiversity at all scales are “resistance” and “resilience.” Resistance refers to the persistence of an assemblage unchanged through a perturbation; resilience refers to an assemblage response to a perturbation that quickly returns to its previous state (Townsend and Hildrew 1994). While species traits, such as physiological tolerance, contribute to community resistance, perhaps the most important requirement for community resilience is habitat refugia (Sedell et al. 1990; Townsend and Hildrew 1994). The existence of habitat refugia is crucial to the success of many macroinvertebrate adaptive strategies, including recolonization and behavioral and morphological life history strategies. The availability of these refugia, however, depends on the spatial and temporal heterogeneity of the system and the degree of connectivity between habitats (Sedell et al. 1990; Townsend and Hildrew 1994; Schneider 1999; Tockner et al. 1999; Malard et al. 2000). In floodplains, habitat refugia is maintained by the natural flow regime, and macroinvertebrate diversity is largely a function of spatially complex yet seasonally predictable regimes consisting of high frequencies of low magnitude flood events (flow pulses) combined with stable frequencies of high magnitude flood events (flood pulses), all of varying magnitudes, creating the highest biodiversity potential (Poff et al. 1997; Ward et al. 1999, 2002; Tockner et al. 2000).

Collectively these results suggest that ecological diversity is expressed through spatial and temporal heterogeneity. Placed in this context, two things become apparent: (1) hydrologic connectivity is critical for maintenance of riverine biodiversity; and (2) the ecologically relevant scale of assessment should be the reach scale, where hydrologic connectivity results in close linkages between biological diversity and habitat diversity (Richards et al. 2002). The following series of figures illustrate how overall increases or decreases in hydrologic connectivity between habitats can reduce the range of connectivity levels and hydroperiods expressed by different habitat types throughout the reach, resulting in reductions in both habitat refugia and diversity. The purpose is simply to illustrate the conceptual link between hydrologic connectivity level (resulting from rising and falling stream stages) and habitat/biological diversity (Wardrop et al. 2012). In Fig. 10.9a, much of the wetland habitat is characterized by saturated conditions intermixed with inundated channels and pools. Flow pulse habitats are highly diverse and consist of shallow areas with cobble/gravel substrates, collections of woody debris and organic material, and pools filled with a mix of sand, silt, and detritus. Floodplain channels are typically a mix of inundated scour pools, floating vegetation and mucky, saturated areas along the margins. Figure 10.9b illustrates a reach with less connectivity between habitats resulting in fewer habitat types. Wetland habitats consist mostly of inundated channels or pools. Flow pulse habitats have dried up. Floodplain channels either complete hydrarch succession and become habitat for terrestrial arthropods or persist, but become anoxic, supporting tolerant taxa with physiological adaptations to anoxia. Riffles may contain fewer taxa due to lower baseflows. Other baseflow habitats remain relatively unchanged. Figure 10.9c shows a reach with higher levels of connectivity between habitats (commonly encountered during high flood stages). Wetland habitats are mostly inundated. Flow pulse habitats have been replaced by baseflow habitat. Floodplain channels contain entirely permanent hydroperiods

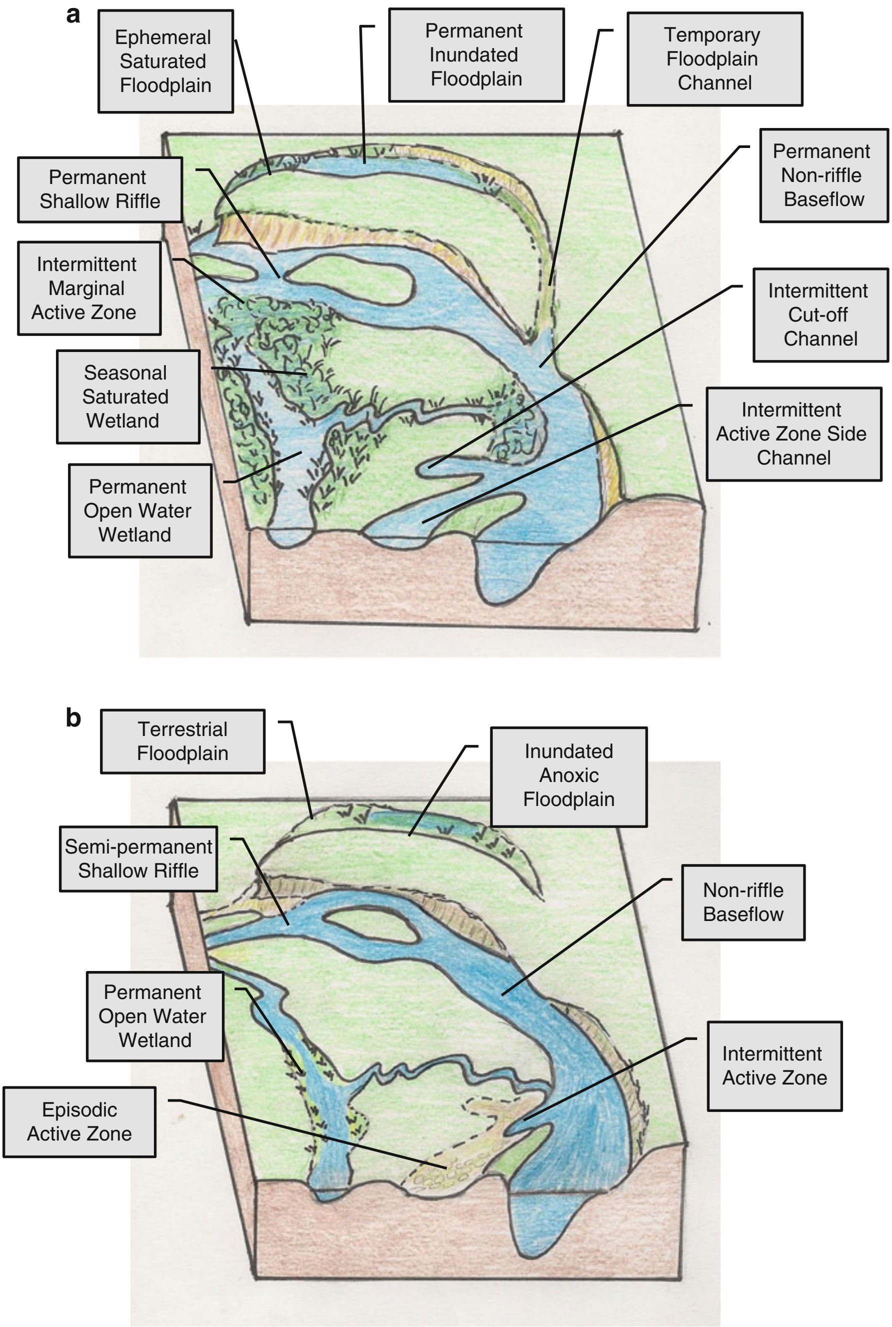

Fig. 10.9 Illustration of a forested headwater or floodplain complex with (**a**) intermediate levels of hydrologic connectivity; (**b**) decreased levels of hydrologic connectivity; and (**c**) increased levels of hydrologic connectivity. Connectivity levels reflect stream stage (e.g., high stream levels result in increased connectivity)

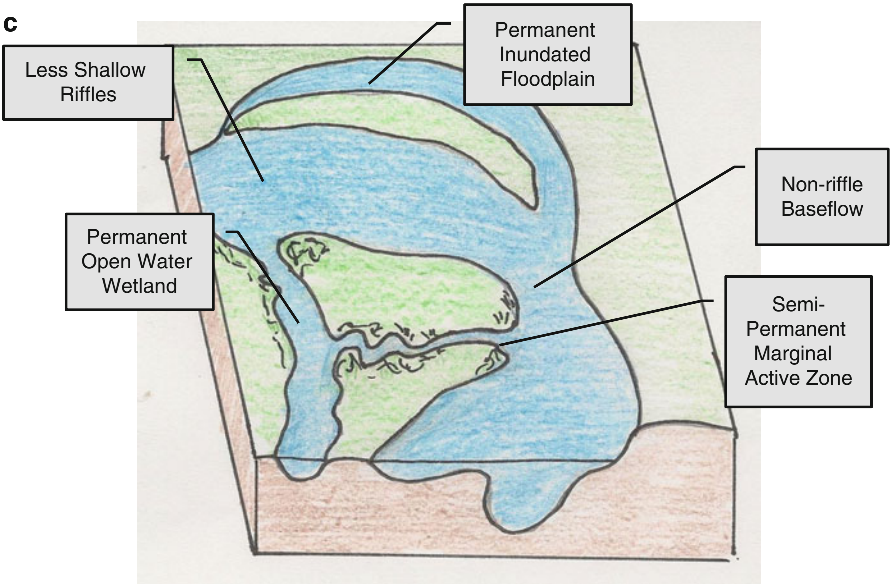

Fig. 10.9 (continued)

where floodplain taxa are outcompeted by pond taxa adapted for biotic interactions in stable lentic conditions. Marginal pools in the active zone remain relatively unchanged. Riffles may become less frequent due to higher flows.

These increasing or decreasing levels of hydrological connectivity can be expressed along both spatial and temporal dimensions. For example, the spatial dimension refers to the surface (or subsurface) water connections between habitats at any given time. The temporal dimension can be represented by differing levels of hydrological connectivity throughout the year and described to a certain extent by the hydroperiod or range of hydroperiods across habitat types. Thus, hydrological connectivity can vary quite a bit in natural systems. From the perspective of invertebrate adaptations, this increased spatiotemporal heterogeneity results in both expansions of available resources and resource partitioning and favors both biodiversity and water quality (Ward et al. 2002; Cardinale 2011; Wardrop et al. 2012). However, anthropogenic disturbances to these systems may cause increases or decreases in connectivity levels that are beyond the community's adaptive capacity, especially when impacts extend beyond the threshold of system resilience. This can result in major losses of biodiversity (Fig. 10.10).

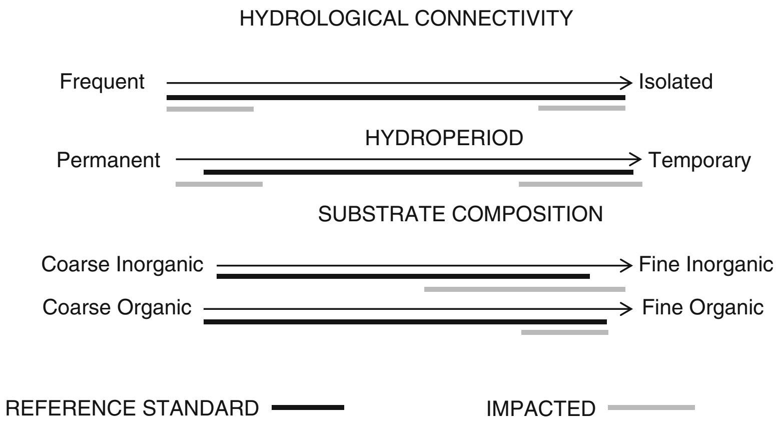

Fig. 10.10 Conceptual figure showing the range of hydrologic connectivity levels, hydroperiods, and substrate composition in reference standard vs. impacted floodplain complexes. Reference-standard systems typically span the range of environmental variability along the continuum, whereas impacted systems typically represent/support habitats represented primarily at the extreme ends of environmental gradients (e.g., either permanent or intermittent hydroperiods)

10.4.4 Advantages of Incorporating a Habitat Approach into Riverine Bioassessments

A large part of the reference standard's value lies in its inherent heterogeneity, which is caused by a mix of surface water and groundwater habitats. These habitats create an intricate network that does not exist in a disturbed setting (Laubscher 2005; Wardrop et al. 2012). Thus, areas that are naturally more diverse in aquatic habitats tend to be more indicative of biological integrity. This is especially true of headwater systems. The goal of wetland biological assessment and the ICIs is to measure this integrity. Riverine systems, for example, need to be evaluated, not only on their ability to provide both stream and floodplain habitat, but also on the quality of both habitat types provided. This includes an evaluation at multiple scales and consideration of connectivity. Overall, communities definitely show responses to human disturbance and macroinvertebrates are good indicators for wetland assessments (their mobility and utilization of multiple aquatic habitat types adds information that is probably not apparent in plant assessments of condition) (Conklin 2003; Laubscher and Conklin 2004; Laubscher 2005). However, failure to consider the effects of natural variation (e.g., flow vs. flood pulses, seasonal vs. permanent hydroperiods, etc.) within a wetland subclass makes it difficult to differentiate between natural and anthropogenic effects. Analysis of reference-standard headwater and floodplain complexes confirmed that macroinvertebrate communities are significantly different within sites, typically at the level of the habitat unit. Thus, identifying these ecologically relevant habitat types and stratifying assessments and metric selections by habitat should improve macroinvertebrate ICIs. In other words, the best approach may be

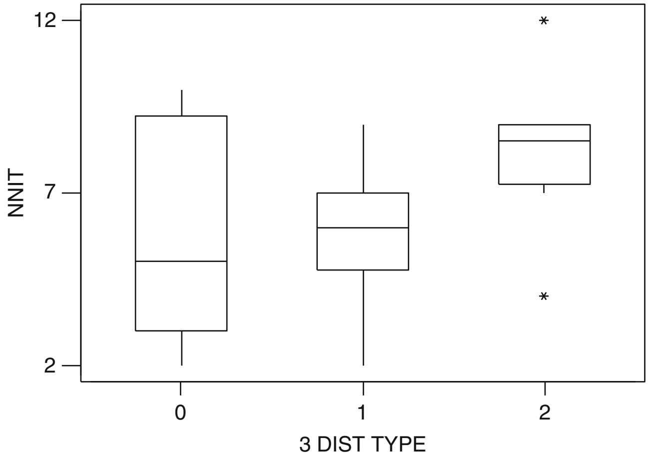

Fig. 10.11 Box plot showing response of a floodplain metric (*NNIT* number noninsect taxa) from the mainstem RICI (floodplain complexes) across three categories of human disturbance (0 = reference standard; 1 = moderately disturbed; 3 = severely disturbed or impacted)

to develop metrics for particular habitat types and determine which types are most prevalent for each HGM subclass. For example, a perennial depression ICI would include metrics associated with ephemeral (both seasonal and temporary) pools, while a headwater complex ICI may consist of these metrics plus metrics related to the active zone and floodplain (the actual metrics chosen would be determined during a site visit by evaluating the major habitat types present).

Although total ICI scores were significantly higher in reference-standard sites than impacted sites, these community responses are not necessarily linear. This along with the large amount of variation in the reference standard makes it difficult to diagnose specific stressors or impacts to the system and develop management plans for preventing and mitigating those impacts. Figure 10.11 illustrates this with the floodplain metric NNIT (number of noninsect taxa) from the mainstem RICI (also known as floodplain complexes). Two important considerations for evaluating metric performance are discriminatory power and scope of impairment (Barbour et al. 1996; Klemm et al. 2002). The former represents a metric's ability to segregate reference-standard sites from impacted sites and is scored based on the overlap between the interquartile ranges. Notice the overlap between the interquartile ranges for NNIT metric at different disturbance levels, demonstrating a lack of discriminatory power (Fig. 10.11). Many floodplain and other wetland metrics also performed poorly with regard to scope of impairment (Fig. 10.11). The relative scope of impairment provides information regarding an attribute's variability in the reference condition as compared to the range of impairment (USEPA 1998) and is measured for metrics that increase with increasing disturbance (e.g., NNIT) as the difference between the highest quartile of the reference-standard sites and the maximum overall value.

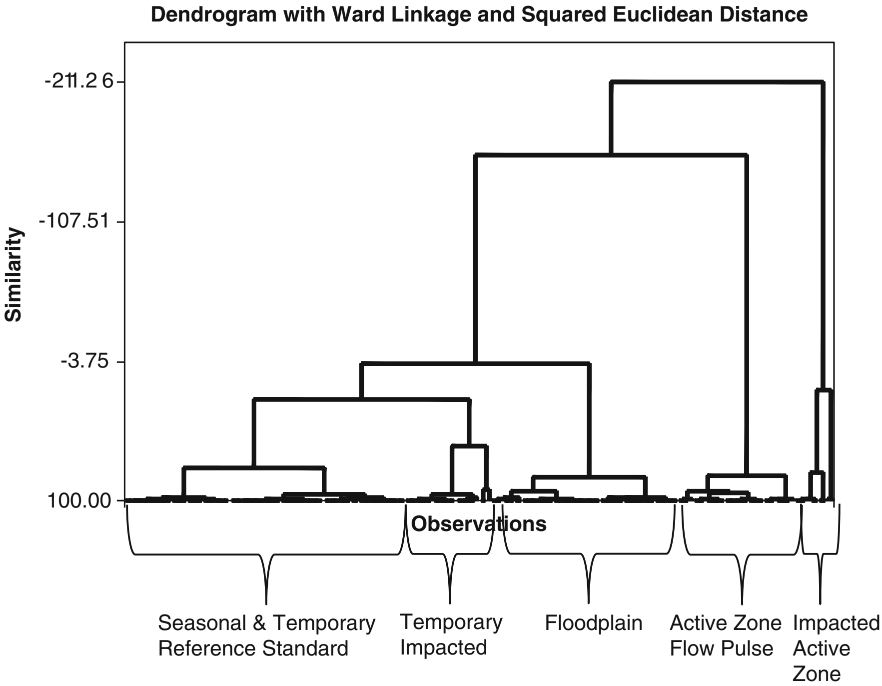

Fig. 10.12 Hierarchical cluster analysis of macroinvertebrate relative abundance data collected from various reference standard and impacted headwater and floodplain complexes

The high variability in the reference-standard values for floodplain metrics should not be surprising, when one considers that these metrics encompassed any non-baseflow habitat; thus, macroinvertebrate data collected from two different ecological sets (active zone and floodplain) were lumped together to determine the metrics. For example, applying the habitat classification to floodplain complexes can explain much of the variation in the reference-standard values of the NNIT metric, which tends to increase with increasing anthropogenic impact (Fig. 10.11). Noninsect taxa (e.g., snails, mollusks, aquatic worms) can become quite diverse and abundant in lentic floodplain habitats but typically only a few genera inhabit flooded areas (e.g., secondary channels) in the active zone. Thus, this metric should only be applied to habitats in the active zone.

Figure 10.12 illustrates the necessity of stratifying this data by habitat type before comparing results between reference standard and impacted floodplains. As the dendrogram shows, natural variation between habitat types exerts a stronger influence on macroinvertebrate community distributions in wetland and floodplain habitats than human disturbance. The biological data separated sites first by habitat type, then by disturbance. Active zone habitats, while clearly different from the other habitat types, are particularly responsive to human impacts and convert to an entirely different assemblage (impacted active zone habitats were the first to separate during the cluster analysis) (Fig. 10.12). This is basically due to the loss of flow pulses as streams become disconnected from their floodplains and the habitats maintained by these frequent, low magnitude flood events disappear and are replaced by deep, scour pools that are created during less frequent, higher magnitude floods. Although the floodplain habitat type didn't clearly distinguish between reference standard and impacted sites, after I analyzed the data separately (sans seasonal, temporary, and active zone habitat data) there was a definite shift in the community as floodplains became increasingly impacted. Reference-standard floodplain habitats were often dominated by the isopod *Caecidotea* and by mayflies or caddisflies with specialized life history adaptations to intermittent hydroperiods. As floodplain habitats became increasingly impacted (usually less connected to the main channel) the community shifted to a dominance of aquatic worms, midges, aquatic beetles, and snails.

10.4.5 Ecological Responses to Anthropogenic Disturbance in Riverine Systems

Figure 10.13 summarizes the major changes (and probable causes) in active zone, floodplain, and seasonal/temporary pool habitats of both headwater and floodplain complexes as land use shifts from primarily forested (reference standard) to agricultural and urban uses (impacted). In addition, the taxonomic shifts in the macroinvertebrate assemblages within these habitats are also included. Riffle habitats, being hydrologically stable (i.e., possessing a relatively permanent lotic hydroperiod), respond primarily to changes in water quality. This is one important reason why they are often the habitat of choice for stream biological assessments. Decreasing water quality is brought about by multiple changes to the instream habitat including increased embeddedness of riffle substrates from fine sediment during runoff events and lower dissolved oxygen levels and higher stream temperatures from loss of forest cover. These habitat changes are reflected by the community primarily through losses of intolerant EPT taxa which are replaced by more tolerant EPT taxa in moderately disturbed reaches and eventually by tolerant taxa from other insect and non-insect orders in severely impacted situations.

Flow pulse habitats in the active zone experience both increases and decreases in hydrologic connectivity (Fig. 10.13). Those closest to the main channel become more connected as a result of increased bank erosion and higher magnitude flood events that scour the channel, converting these habitats to baseflow habitats, thereby reducing habitat heterogeneity. Thus, the typical flow pulse taxa are replaced by common nonriffle taxa from the main channel. Flow pulse habitats farther away from the main channel become more disconnected, only receiving flood waters during high magnitude flood events and become either semipermanent or highly

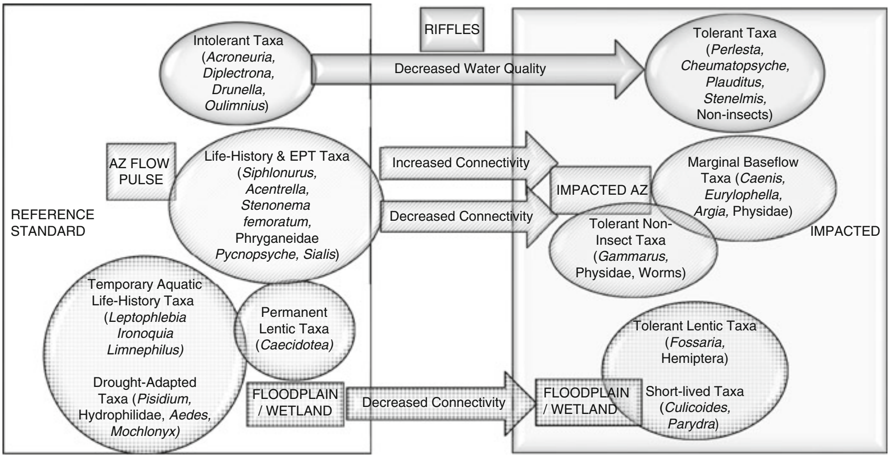

Fig. 10.13 Summary illustration of the major changes in habitat and macroinvertebrate structure from floodplain complexes as land use changes from forest to agriculture and residential. *Arrows* depict the likely source of change in each habitat type

ephemeral habitats. The former are typically in the form of deeply scoured pools and function much like aquatic floodplain habitats, but interestingly house an entirely different community. Here, taxa with life history adaptations to seasonally available flow pulse habitats are replaced by generalist taxa more tolerant to stagnant, lentic conditions. The latter, more ephemeral habitats support either aerial colonizers or taxa with rapid growth rates.

Groundwater-supported seasonal wetland habitats and temporary pools may lose both lateral connectivity with the main channel and vertical connectivity with subsurface flows (Fig. 10.13). In reference-standard complexes they support a wide variety of taxa, many rare and unique to specific wetlands. These taxa drop out as human activities impact the wetland, creating more uniform habitat characteristics and supporting taxa with shorter life cycles and greater tolerance to extreme environmental conditions.

10.5 Conclusions/Final Remarks

In order to preserve riverine biodiversity, we must pay special attention to areas along the river continuum where this biodiversity is maximized. Riverine headwater and floodplain complexes contain multiple wetland habitat types and represent hot spots of ecological diversity. Future research should involve the gathering of empirical evidence from complex riverine reaches regarding the diversity of aquatic habitats, their biology, and the underlying hydrologic processes responsible for their creation and persistence. Such an endeavor requires: (1) an assessment unit large enough to encompass the floodplain mosaic (i.e., minimum of reach scale); (2) consideration of the temporal effect on optimal hydrologic connectivity, aquatic habitat, and macroinvertebrate diversity (e.g., seasonal sampling); and (3) biological sampling of all aquatic habitat types within the assessment unit supplemented with hydrologic monitoring data.

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Chapter 11
Monitoring and Assessment of Wetlands: Concepts, Case Studies, and Lessons Learned

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Abstract Monitoring and assessment (M&A) have long been considered critical components of any resource management program where there is a need to evaluate progress and performance over time. Understanding the origins of current monitoring and assessment strategies and techniques for wetlands in the United States provides useful perspectives on how wetlands are both similar and different from other waters and allows us to take advantage of the lessons learned across all aquatic resources. We highlight several knowledge threads that significantly influenced how we approach M&A today, including legal mandates, tools developed to improve the management of resources, and scientific evidence of the utility of M&A informa-

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tion. We describe the role of regional forums in the evolution and development of these tools and in the building of support for their programmatic integration in the Mid-Atlantic Region (MAR). We then tell the story of their use and application at a variety of spatial scales, including site-level mitigation applications in Pennsylvania, watershed application in the Upper Juniata Watershed, regional application in the MAR, and, finally, national application in the National Wetland Condition Assessment (NWCA). We document the lessons learned, and present an example of promising future use of M&A data in the construction of Tiered Aquatic Life Use (TALU) Standards for wetlands.

11.1 Introduction

Monitoring and assessment (M&A) have long been considered critical components of any resource management program where there is a need to evaluate progress and performance over time. A number of major US environmental programs are built on a template of ecosystem-based management, which generally emphasizes four main principles: (1) integration of ecosystem components with resource uses and users, (2) focus on sustainable outcomes, (3) avoidance of deleterious outcomes, and (4) use of an adaptive approach wherein experience leads to more effective management. Within the last decade, the adaptive approach has been developed, articulated, and institutionalized to varying degrees (e.g., Thom 1997, 2000; Thom et al. 2005). Monitoring is foundational to adaptive management, providing measures of management performance and ecosystem response and leading to an increased understanding of the ecosystem and effective management mechanisms. The value of M&A information is recognized in the design of major regulatory frameworks. For example, the Federal Water Pollution Control Act of 1972, Public Law 92–500, commonly referred to as the Clean Water Act (CWA), specifies a need to monitor, compile, analyze, and report on water quality data, broadly defined (CWA§106(e)(1)). Wetlands are included because they are "waters of the U.S." Thus, there is both a management imperative and a legal basis to monitor and assess wetlands at a variety of spatial scales, from watershed to nationwide.

Understanding the origins of current monitoring and assessment strategies and techniques for wetlands in the United States provides useful perspectives on how wetlands are both similar and different from other waters and allows us to take advantage of the lessons learned across all aquatic resources. The threads of M&A approaches for wetlands in vogue today can be traced primarily from policies and activities initiated by the U.S. Environmental Protection Agency (USEPA) and the U.S. Army Corps of Engineers (Corps) in the late 1980s and early 1990s that stimulated a significant record of applied research by agency and academic scientists. It is not our intent to exhaustively list every individual or organization that contributed to the expansion of our knowledge base on wetlands science, management, and monitoring—there were many, and we benefited from the many publications and conversations that occurred. Rather, our intent is to illustrate the value and utility of wetland M&A by:

1. Highlighting several knowledge threads and efforts that significantly influenced how M&A is typically approached today
2. Telling the story of their use and application at a variety of spatial scales in the Mid-Atlantic Region (MAR)
3. Documenting the lessons learned
4. Presenting some promising future uses of M&A data

11.2 Initial Knowledge Threads: Establishing a Monitoring Framework

The current framework for monitoring wetlands was initiated as a response to legal mandates for the protection of resources and human health, such as the CWA, and the need to manage resources effectively. As monitoring programs and tools were developed to respond to these needs, the scientific evidence that M&A was worth the effort appeared in various forms and the momentum for M&A began to build in earnest.

11.2.1 Legal Mandates for M&A

Monitoring and assessment are essential for any wetland regulatory program to evaluate the performance of permitting, mitigation, and compensation. Although there are legal mandates and guidance for M&A, agency resources are often depleted prior to the M&A phase.

The CWA of 1972 addresses the need for monitoring in §305(b) and §303(d). States are required to report on the status of their water-related activities with information compiled from M&A data. Progress was initially made with streams and rivers with the intention of adding other waters as methods were devised and tested. States are expected to develop and adopt ten elements that comprise an overall water M&A program (USEPA 2003) because, "Broad-based, integrated *monitoring and assessment* programs inform decision makers, target restoration activities, and help us address significant stressors." (USEPA 2006a, p. 102).

11.2.2 M&A in the Management of Resources

Inability to respond to legal mandates often stimulated development of tools for better management of wetlands. Assessment of cumulative impacts is a case in point.

In 1988, the newly formed Wetlands Research Program of the USEPA, convened a workshop to address the ever-elusive topic of how to measure cumulative impacts to wetlands (Preston and Bedford 1988). The workshop initiated discussions and projects concerning how to address cumulative impacts on a watershed basis, how to measure wetland function and condition, and how to develop measures of biological integrity (Preston and Bedford 1988), which eventually led to progress in state M&A programs.

USEPA's Wetlands Division began a concerted effort to build M&A capacity within state wetlands programs in 2000 by establishing two national priorities: (1) assist states and tribes to develop wetland monitoring programs; and (2) improve the success rate of compensatory wetlands mitigation. Between 2003 and 2006, USEPA developed guidance and adopted the three-tiered approach for monitoring wetlands, urging states and tribes to include ten elements in their programs (USEPA 2006a). The three-tiered approach as described by Brooks et al. (2002), and further refined in USEPA's Elements Letter (USEPA 2006a) has the following components:

- *Landscape assessment* (Level 1) uses remote sensing data and field surveys to inventory wetlands and riparian areas
- *Rapid assessment* (Level 2) uses field diagnostics to assess condition of wetland sites
- *Intensive assessment* (Level 3) provides the quantitative data to validate rapid methods, characterize reference condition, and diagnose the causes of wetland condition observed in Levels 1 and 2

More recently, the EPA's guidance called for four core elements in a successful wetlands program (reduced from ten for operational convenience); i.e., M&A, regulatory activities, restoration and protection, and water quality standards (USEPA 2012). M&A is the first core element and is key to tracking performance of any regulatory or management program. Competitive funding through the Wetland Program Development Grants Program provides incentives (CWA; §104(b)(3)). This source of funding has generated dramatic progress within some regions (e.g., Mid-Atlantic and New England), and in some states (e.g., California, Ohio, and Montana), bringing the nation closer to full implementation of wetlands protection programs.

Yet another thread can be traced back to the late 1980s and a goal to restore wetlands. The need for restoration/creation became apparent from three sources: mitigation of impacts or losses related to permitting activities managed by the Corps through §404 of the CWA; restoration of waters designated as impaired from a water quality perspective; and the recognition that the nation needed to curb the extensive losses of wetlands (e.g., Dahl 2011) and restore what had already been lost or degraded. In 1987, the USEPA called for the establishment of a National Wetlands Policy Forum, which was charged with making recommendations for national policy on the protection of wetlands (NRC 1995). The Forum's central recommendation was revolutionary, calling for "no net loss" and "long term gain" in area and function of wetlands (The Conservation Foundation 1988). The first phrase became policy under the administration of President George H. W. Bush, and has continued under every US President since that time.

Even with policy support, efforts to foster long-term gains in wetland area foundered despite scientific evidence that wetlands mitigation was not at all adequate (e.g., Kusler and Kentula 1990; Kentula et al. 1992; Zedler and Callaway 1999; Gwin et al. 1999). Calls for change continued to ring out, beginning with a National Research Council (NRC) report on restoration of all waters (NRC 1992). Again, in a 2001 report, the NRC indicated that the goal of no net loss of wetland function was not being met due to poor mitigation policy and implementation (NRC 2001). In 2002, an interagency National Wetlands Mitigation Action Plan was released, outlining specific tasks needed to improve the integrity of mitigation wetlands (USACE 2002). To help correct this record of poor mitigation performance, the USEPA and the Corps jointly developed and issued the Mitigation Rule of the CWA (33 C.F.R. Parts 325 and 332 and 40 C.F.R. Part 230) (USEPA 2008). The new rule requires mitigation to be carried out in a landscape context using the best available science, to the extent appropriate and practicable. Under the new rule, states must devise measureable and enforceable standards to be used in assessment of mitigation wetland performance during regular monitoring periods. This guidance recommends using many of the approaches and tools described in this chapter.

11.3 Building Support for M&A at Multiple Scales

By the beginning of the new millennium, the mandate for comprehensive wetlands monitoring and assessment had never been stronger, and a number of technical tools had been developed. However, no integrated, transferable, or scalable approach to M&A had emerged. The primary reason for the diverse collection of M&A methods was that the efforts had not occurred through any one model of funding and/or development. Hydrogeomorphic (HGM) classification and functional assessment models had been primarily regional efforts, under the direction and support of the Corps and the USEPA. Development of biological assessment methods had followed suit. They were composed primarily of regional efforts, funded by various sources, and represented to some degree by the USEPA-supported Biological Assessment of Wetlands Working Group (BAWWG). In contrast, assessment of wetlands in a watershed had been mainly represented by projects in the Nanticoke (Whigham et al. 2007) and Upper Juniata (Wardrop et al. 2007a, b) watersheds, funded by USEPA's Environmental Monitoring and Assessment Program (EMAP). Many smaller efforts had taken place across the country, and both the regional/local focus and spectrum of funding sources had made integration of a comprehensive monitoring and assessment program and its technology transfer difficult, if not impossible.

What emerged from these disparate efforts was a clear need for a forum to facilitate the development and implementation of wetland monitoring strategies. Moreover, the forum could not be effective if it chose to tackle specific monitoring issues on a national basis. The range of wetland types and management issues would dilute such an effort past its point of utility. Such a forum, therefore, needed to address issues on a regional basis. Additional reasons for the formation of a regional wetlands workgroup, most specifically in the MAR, were also in play. Wetland monitoring protocols to meet CWA requirements needed to be developed as a result of lawsuit settlements in Pennsylvania and Delaware. The lawsuits had illuminated the need for an interstate and interagency effort to determine how wetlands monitoring, and restoration could be integrated (e.g., in the development and implementation of Total Maximum Daily Loads, or TMDLs). The growing presence of volunteer monitoring networks, such as the Pennsylvania Organization for Watersheds and Rivers, needed a designated source of technical expertise. Therefore, the overarching goal became the support of a forum to facilitate the development and implementation of wetland monitoring strategies, including elements of a comprehensive wetland monitoring program that met the needs of the Mid-Atlantic States.

Specific models existed for such a forum. The previously mentioned BAWWG, an *ad hoc* national working group formed by USEPA in mid-1990s, had been established for the technical and feasibility review of biological assessment tools, specifically the development of Indices of Biologic Integrity (IBI) (Karr et al. 1986). The BAWWG met periodically during the late 1990s and early 2000s and was led by Susan Jackson and Doreen Vetter of USEPA. Convening this group brought varied scientific and taxonomic experts to the table to develop biological assessment tools to a point where they could be implemented in monitoring programs. The effectiveness of this forum is evidenced by the publication of the Methods for Evaluating Wetland Condition modules (e.g., USEPA 2002), a series of 14 white papers that provide a blueprint and toolbox for the use of biological assessment tools in M&A programs at a variety of scales. Perhaps a more important outcome from the BAWWG was the growth and development of a network of scientists and managers that understood the goals of wetlands M&A and who, collectively, populated the academic, agency, and consulting landscapes with a series of related approaches to M&A.

Regionalization of the BAWWG approach had been identified as a need by the group. At the first national BAWWG conference held in Orlando, FL, in May of 2001, a consensus was reached as to the need for work in biological assessment to continue, primarily through regionalization of approaches. The reasons stated above spoke to the need for a regional workgroup specifically in the Mid-Atlantic. The New England BAWWG (NEBAWWG) served as an early model for the role of such a group in regional wetland and related aquatic issues.

11.3.1 Regional Forums for Monitoring and Assessment: The MAWWG Example

Using the example provided by NEBAWWG, the Mid-Atlantic Wetland Workgroup (MAWWG) was initiated in 2002 with funding provided by USEPA. MAWWG experienced early and immediate success due to a number of factors. Academics and agency personnel from Pennsylvania, Ohio, and Delaware already had strong ties to the national BAWWG and with each other. In addition, EPA-funded M&A projects had already been conducted in Delaware, Maryland, Ohio, Virginia, and Pennsylvania.

The primary objective of the MAWWG was, and still is, to provide a forum to facilitate the development and implementation of wetland monitoring strategies, including elements of a comprehensive wetland monitoring program that met the needs of the Mid-Atlantic States (i.e., wetland monitoring programs to be implemented at the state level). Primary goals for the MAWWG are:

1. Provide the technical support necessary for improved coordination of surface water and wetland monitoring programs, with the eventual long-term incorporation of wetlands into traditional water quality monitoring programs (e.g., CWA § 305(b), 303(d), 319, and 106)
2. Regionalize existing monitoring and assessment tools for wetlands, such as HGM classification and functional assessment and biological assessment
3. Use monitoring and assessment tools to improve restoration and mitigation
4. Provide training for regulatory personnel in monitoring and assessment methods
5. Provide a source of information on monitoring and assessment tools through a workgroup web site

Over its 10 years of existence, the MAWWG membership has been composed of participants from nine states: Delaware, Maryland, New Jersey, New York, North Carolina, Ohio, Pennsylvania, Virginia, and West Virginia. These states represent 9% (~69 million ha) of the nation's contiguous land area and 10% (~4 million ha) of the nation's wetlands (excluding Alaska and Hawaii). The tools and products of both the individual states and the group (MAWWG) are made available through the group's website (http://www.mawwg.psu.edu). The material on the website includes a range of bioassessment and functional assessment tools (including an online calculator for the Floristic Quality Assessment Index (FQAI)), protocols developed at a regional scale for a Mid-Atlantic Wetland Condition Assessment, the results of this assessment, and a reference wetlands database to improve mitigation design and performance. The core states of MAWWG, their representatives, and the various academic and agency partners are presented in Fig. 11.1. The following sections detail how the various tools were developed and successfully deployed at state and regional levels, with the support of MAWWG.

11.3.2 MAWWG and the Implementation of M&A

At its initiation, the MAWWG members had a wide range of experience with incorporation of M&A into regulatory or non-regulatory programs. The motivation for wetland M&A varied across the states. Pennsylvania was anxious to embark on statewide condition assessment monitoring (partially due to a legal mandate discussed previously), while Delaware's approach was directed more towards improving the effectiveness of restoration. Maryland intended to develop water quality standards for wetlands; West Virginia was initiating protection of its highest quality sites; and Virginia's need for support of permitting decisions was becoming critical. Each of these purposes required unique information on the status of the resource

Fig. 11.1 Member states of the Mid-Atlantic Wetland Workgroup (MAWWG) and their representatives

and its primary threats. The collective need was for an approach to appropriately sample the resource, assess its function and/or condition, and report the results in a way that was helpful to the program of interest. A timeline with major milestones in the implementation of M&A by MAWWG, selected state products that were shared within the group, and the major collaborative products that emerged are presented in Fig. 11.2. The following sections detail MAWWG's experience in each of the major areas of M&A.

11.3.2.1 Survey Design

A major impediment in the assessment of wetlands was the lack of a method to obtain a statistically valid sample suitable for making inferences about a population of wetlands at a specified spatial scale. For example, how many wetlands did one need to characterize in a small watershed to make statements about the overall condition of wetlands or the level of function provided? Additionally, the issue of identifying an appropriate survey design was critical to answering the ever-present question on the degree of cumulative impacts due to wetland loss. M&A of wetlands was (and remains) complicated by the fact that, even if resources are available

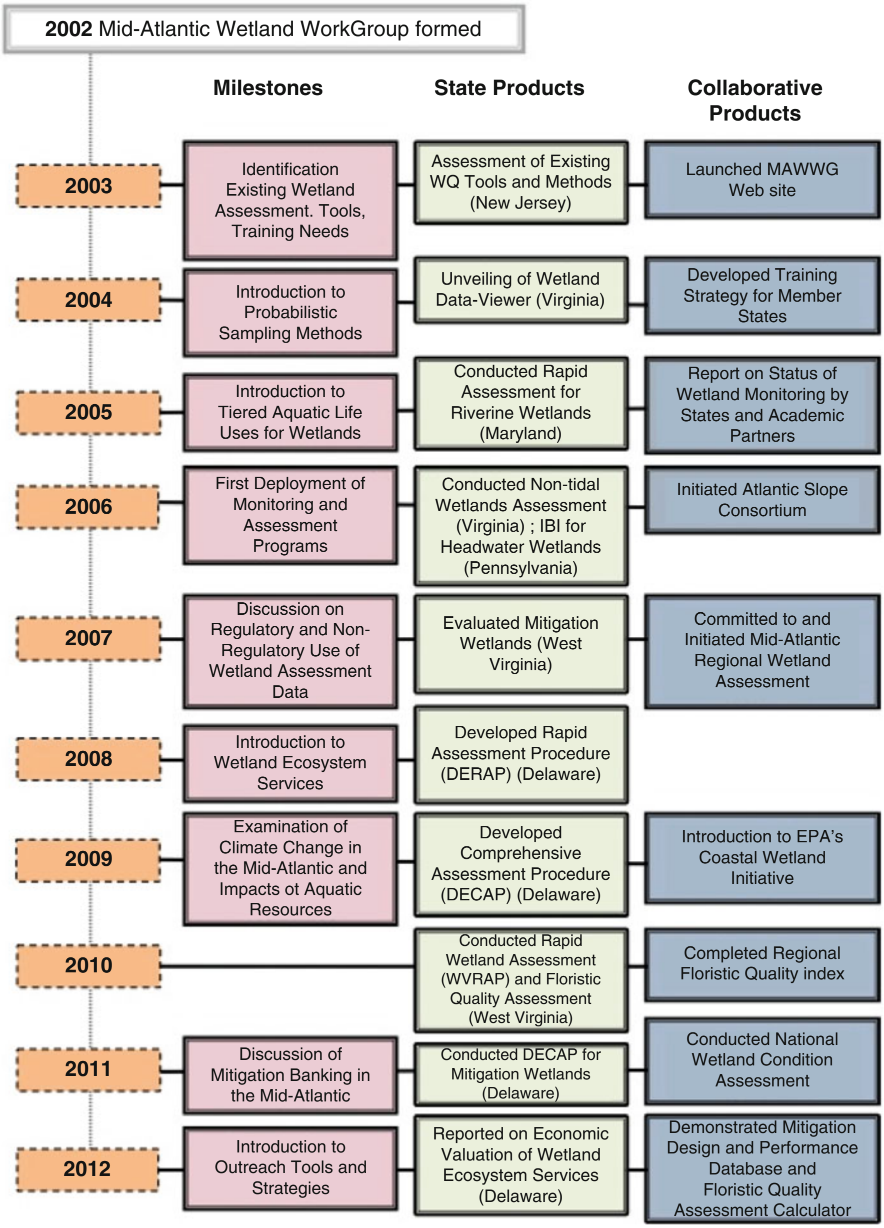

Fig. 11.2 Timeline of major milestones, selected state products, and collaborative products of MAWWG

for all wetlands to be sampled, a high percentage of wetland area nationwide exists on private property, making any approach requiring a complete census nearly impossible due to access considerations. Thus, a survey design for selecting sites to provide valid data for developing accurate estimates for the entire population or area of interest became a priority requirement for the further evolution of M&A.

The elimination of a census as a survey design puts one on a path to a probability survey, defined as a survey in which every element (wetland) has a known probability of being selected for assessment, and the inferences derived from assessing a sampled subpopulation can be applied to the entire population. Because wetlands are distributed across the landscape as discrete elements, linear features, and as a matrix for other systems (e.g., the Everglades), the survey design must include a spatial component. There are a number of approaches to spatial survey design, and the expertise required to choose the one most appropriate for a particular use went beyond the expertise of most state agencies. Fortunately, the answer arose from the efforts of USEPA's Environmental Monitoring and Assessment Program (EMAP), which had long been tasked with developing the science needed to assess the state of the nation's aquatic resources at various spatial scales. EMAP was charged with answering a suite of monitoring questions about the nation's waters: what is the overall quality, to what extent is it changing over time, what is causing the problem, how might we fix it, and how effective are our management techniques? EMAP's approach to answering these questions on a national scale was directly applicable to answering the questions for any individual state.

At the fifth meeting of MAWWG (December 2004), Anthony R. (Tony) Olsen of USEPA Office of Research and Development introduced MAWWG to the Generalized Random Tessellation Stratified (GRTS) design (Stevens and Olsen 1999, 2000, 2004), which was initially developed as a way to sample other aquatic resources, such as streams, rivers, and lakes. Briefly, the GRTS design results in a spatially balanced sample with the points (i.e., locations selected from the sample frame) ordered so that sequential use of the points as study sites maintains spatial balance (i.e., the spatial density pattern of the sample closely mimics that of the resource). In other words, a list of wetland points would be provided to the field crew; the crew would then pursue access to each wetland in the list in order of the draw. Implementing GRTS as a statistical technique requires consideration of a number of factors, and two are of special note: the identification of the sample frame, and the identification of any desirable stratifications of the data. Both are illustrated below with examples of how MAWWG members addressed them.

The sample frame is the digitally mapped representation of the target population (in our case, wetlands of all types in the Mid-Atlantic) that is used to select the sample sites. The National Wetland Inventory (NWI) is a commonly used sample frame for wetlands, because NWI is the most complete digital map of wetland location, type, and extent that is nationally available. However, previous experience in many Mid-Atlantic States, including Pennsylvania, had suggested that NWI missed many small wetlands in forested portions of the landscape, resulting in significant under-coverage (Brooks et al. 1999). Wardrop et al. (2007a) present one approach to supplementing the NWI in an assessment of the condition of wetlands in the Upper Juniata Watershed in south central Pennsylvania.

Geologic structural and stratigraphic information in combination with floodplain maps were used to generate a map of areas with high probability of wetland occurrence, which were screened using recent aerial photography and then groundtruthed. The identification of these additional wetland areas resulted in an estimate of total wetland area in the watershed (2,123 ha) that was almost double that calculated from the NWI map alone (1,144 ha) (Wardrop et al. 2007a).

Another aspect of survey design is the identification of subpopulations of wetlands that may vary in important ecological characteristics, such as size classes, vegetation type, or wetland type and should be included in the reporting on the results of the survey. Stratification, as used in this case, is the process of identifying these relatively homogeneous subgroups and obtaining a representative sample from each. Stratifications can either be an explicit part of the survey design, or can be applied after the sample has been obtained. An example of the former is the approach taken by the Pennsylvania Department of Environmental Protection (PADEP) when planning for their rotating basin assessment of wetland condition in the Commonwealth. PADEP chose surrounding land cover as a criterion for defining four subpopulations of wetlands and utilized the proportion of (or lack of) disturbed cover and natural cover in a 1-km radius circle surrounding each wetland to identify the subpopulations on the sample frame. PADEP's rationale for this approach was that wetlands in similar land cover contexts would be subjected to the same suite of stressors, and these stressors would likely negatively impact condition. The land cover class could be used as an organizing factor around which to prescribe the appropriate family of Best Management Practices (BMPs) that would improve overall wetland condition in the most effective manner. In an example of post-sampling stratification, an assessment of wetlands in the Nanticoke watershed in Delaware by Whigham et al. (2007) encountered a sizable number of privately owned sites for which access permission was denied or was neither explicitly given nor denied, raising concerns about the representativeness of the achieved sample. Because of the possibility of differing management practices between public and privately owned wetlands, and the potential to affect wetland condition, the sample was post-stratified on ownership (methods in Stevens and Jensen 2007).

In summary, the collaborative process between USEPA and MAWWG on survey design had significant implications for both groups that stretched far beyond the technical details of using GRTS to generate a sample. For MAWWG, the consideration of survey design issues forced reflection and discussion on an entire suite of questions that needed to be formulated into monitoring objectives, such as consideration of the amount of riverine wetlands (the most common HGM type) in low condition across an individual state. For USEPA, the demand for technical support for probability-based sampling from MAWWG members reinforced the importance of an effort to make software to create GRTS-based survey designs publicly available (http://www.epa.gov/nheerl/arm/analysispages/software.htm). In addition, questions of how best to report the results of a condition assessment also required the identification of wetland subgroups that may differ in the anthropogenic impacts to which they are subject and in the manner of their response to similar impacts. Thus, a renewed interest in classification followed in the MAWWG.

11.3.2.2 Classification: The Importance of Context

Monitoring information is often utilized in an administrative sense, ignoring its landscape and system context. For example, an inventory of the aquatic macroinvertebrates found in a stream gives few clues about why things are the way they are. Without the accompanying assessment of habitat conditions (e.g., inhospitable benthic conditions or poor water quality) and the human activities that created them, we are without direction in ameliorating a bad condition and restoring valuable function. We must, as Luna Leopold (1977) states, adopt a philosophy of water management that recognizes the hydrologic system as deeply interconnected and placed in the context of geography and climate.

Initial M&A efforts for wetlands provided a way to incorporate the landscape context, as recommended by Leopold (1977), in the design and analysis of M&A efforts. Classification systems were devised that are based on geography and climate, and on hydrogeomorphology. For example, there are the descriptions of ecoregions developed by Omernik (1987) and Bailey (1995) and the HGM classification of wetlands developed by Brinson (1993). Ecoregions exhibit similarities in the mosaic of environmental resources, ecosystems, and the effects of humans (Omernik 1995). They are areas with a relative homogeneity in ecosystems that differ from that of adjacent regions (Omernik and Bailey 1997). Specific to wetlands, Brinson's (1993) HGM classification places emphasis on hydrologic and geomorphic controls that are responsible for determining many of the functional aspects of wetland ecosystems.

Brinson's (1993) HGM classification system looked to properties of geomorphic setting, water source, and hydrodynamics to derive a set of classes of wetlands associated with their ecological functions. Not all wetlands provide the same functions or to the same level (e.g., wetlands in a floodplain setting provide storage of flood waters, while slope wetlands, which by Brinson's definition do not have contours that create a basin, do not). As stated above, HGM classification describes an approach to classifying wetlands that aids in distinguishing the functions that each type can perform and in the identification of the potential effects of anthropogenic disturbance. In contrast, the NWI utilizes a classification of wetlands and deepwater habitats developed by the U.S. Fish and Wildlife Service (USFWS) (Cowardin et al. 1979) wherein wetlands are defined by hydrology, soils, and vegetation in a way that supported the photo interpretation required to create the NWI maps. Therefore, NWI classification does not provide a clear crosswalk between wetland type and the type of function provided, as well as the potential impact by anthropogenic disturbance.

The HGM classification focuses on the drivers of wetland structure and function assures comparisons of "apples to apples," which has clear links to survey design. For example, if flood storage were of interest, only wetlands that were of an HGM type that likely stored floodwaters and were in a landscape position to receive floodwaters would be part of the target population to be assessed. Thus, the associated survey design would assure that only wetlands involved in the function of interest were included.

Whatever the classification scheme, wetland type can be described for an individual site or as a quantitative measure of the abundance of various wetland types at scales from watershed to global. Especially useful are "landscape profiles." These are generally referred to as compilations of the relative abundance of wetland classes defined in terms of the hydrogeologic factors that cause specific wetland types to form and support their functioning in the landscape. The concept of landscape profiles was introduced by Bedford (1996) and then made operational through the use of HGM classification by Gwin et al. (1999). Landscape profiles are critical tools for restoration, management, mitigation, and cumulative impact assessment of naturally occurring wetlands and their utility is widely documented (e.g., Johnson 2005; Wardrop et al. 2007a). For example, in the Upper Juniata watershed, a landscape profile showed that wetlands in the slope class dominated the watershed, followed by riverine types. The profile reflected the physical geography of the region, which has a majority of stream miles in first and second order, and contains abundant toe-of-slope settings with potential groundwater discharge. The profile also highlighted the probable occurrence of significant habitat and biogeochemical functions that are associated with these wetland types.

The reporting requirements of the MAWWG members generally indicated that a HGM approach of classification was necessary because of the reasons stated above. Fortuitously, an additional advantage of HGM classification is its open structure, which allows for regionalization. This is reflected in the abundance of HGM classifications across the country that are generally developed on a regional basis, e.g., the Mid-Atlantic (Brooks et al. 2011) and Oregon (Adamus 2001).

11.3.2.3 The Concept of Reference

Whatever classification is utilized, the next step towards reporting on the condition and/or function of wetlands is setting expectations of condition or function for any specified class and location. For example, what is the difference in likely carbon storage in a depression vs. a riverine wetland? These expectations serve as a benchmark or "reference" for making comparisons and evaluating degradation (e.g., see the discussion of attributable and relative risk in Van Sickle and Paulsen 2008). The concept of reference, as embodied in Brinson's original description of the HGM approach, is often considered one of the most profound legacies of his work (Brinson and Rheinhardt 1996, Chap. 2 of this book). In general, reference denotes a range of wetland conditions that can be correlated with a gradient of anthropogenic impact (Fig. 11.3). Reference standard refers to conditions at the least, or minimally, impacted sites, thereby providing the potential to develop a quantitative description of the best available chemical, physical, and biological conditions in the wetland resource given the current state of the landscape (see Stoddard et al. 2006 for a discussion of various definitions of reference). This conceptual framework and family of definitions is adaptable to any wetland type in any geographic setting.

The power of the reference concept in M&A cannot be overstated. It provides the grounding of either end of the condition/disturbance gradient (Fig. 11.3), as well as

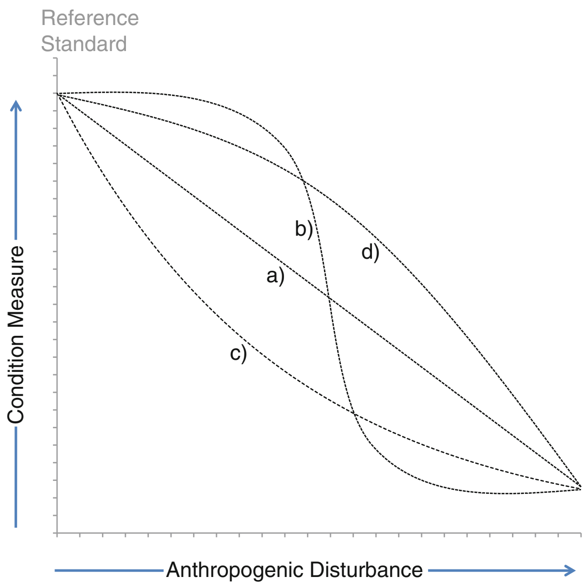

Fig. 11.3 Concept of reference wetlands along a gradient of anthropogenic disturbance with reference standard referring to conditions at the least, or minimally impacted sites: (**a**) linear response of condition to disturbance; (**b**) non-linear response of condition to disturbance; (**c**) and (**d**) potential envelope of reference wetland condition

defining the nature of the relationship (e.g., linear, and nonlinear with thresholds) and the variability in condition at any value of the disturbance gradient (e.g., a range of wetland condition exhibited at high levels of disturbance). It also allows establishment of three benchmarks important to the ultimate management of wetlands: minimally disturbed (condition in the absence of significant human disturbance), least disturbed (condition given the best available condition of the landscape, e.g., wetlands in an agricultural setting), and best attainable (the expected ecological condition of least-disturbed sites if BMPs are employed for some period of time) (Stoddard et al. 2006). All of the MAWWG members have invested significantly in the establishment of a collection of reference wetlands. Riparia at Penn State has consistently utilized its reference collection of 222 wetlands in developing monitoring tools such as HGM functional models and IBIs for macroinvertebrates, plant communities, amphibians, and birds.

11.3.2.4 Evolution of Assessment Tools

A family of assessment methods have allowed us to "connect the dots" between land use, stressors, and resulting ecological condition and functions. However, a major obstacle to implementation of M&A is how to balance the value of the information gathered and the cost of collecting it. The obvious limitation to wetland assessment posed by resource constraints has given rise to a multilevel approach, as currently presented by the three-tiered approach (see Sect. M&A in the Management of Resources 11.2.2) and implemented by a number of states. The level of effort appropriate for a monitoring effort depends on the resources available and the degree of confidence required in the results. As one would expect, the degree of confidence in the data and the reliability of decisions made using the data increase with greater level of effort.

One or more of the three tiers can be employed over a variety of scales (Brooks et al. 2006; Fennessy et al. 2007a; Wardrop et al. 2007a, b; Whigham et al. 2007), and each level can be used to validate and inform the others (Fennessy et al. 2007a). For example, Wardrop et al. (2007a) demonstrated how data from an intensive assessment can be used to evaluate and improve the use of a landscape and a rapid assessment method (RAM). Alternatively, Wardrop et al. (2007b) showed how models of wetland functions that form the components of an intensive assessment can be checked using the results of a landscape and rapid assessment. In another example of how components of the tiered approach work together, Sifneos et al. (2010) used data from an intensive assessment to calibrate a rapid assessment and then employed the resulting correlation between the methods and double sampling (a statistical sampling method) to demonstrate how to make decisions about the number of sites that could be sampled using a combination of both methods for a fixed cost.

Landscape Assessment (Level 1)

A landscape assessment can be accomplished in the office using readily available digital data and a geographic information system (GIS) and requires a low level of effort compared to a site assessment in the field. The most common approach involves the establishment of a reference standard landscape, i.e., the determination of the surrounding land cover that is correlated with a wetland in reference standard condition (see Sect. 11.3.2.3 for a discussion of reference). For example, Wardrop et al. (2007a) chose forested land cover as a reference standard landscape because: (1) it is judged to be the least altered and in the best condition, and (2) non-forested land cover is a surrogate for the stressors that affect wetland condition. Thus, the landscape assessment score measures departure from this reference standard landscape.

Another approach is that developed by Virginia Institute of Marine Sciences (VIMS) in cooperation with the Virginia Department of Environmental Quality (VADEQ). The Virginia Method seeks to utilize the landscape assessment to estimate the level of individual ecosystem services, such as maintenance of water quality and habitat provision, instead of as a general indicator of overall condition. The method assumes that these, individual services (e.g., habitat service or water quality service) are controlled by specific sets of wetland characteristics, and, should not be inferred to be maximized by a wetland in good overall condition. The model construction process is evidence-based and begins by first identifying the ecosystem service of interest; models have been formulated for water quality and habitat. The basic assumption underlying a model is that a wetland's capacity to perform the ecosystem service of interest is greatest when the system is not subject to any stresses that might degrade that performance. A literature search identifies these specific stressors shown to impact the ecosystem service of interest, e.g., modification of hydrology for water quality improvement. The last step is then the selection of landscape characteristics that are correlated with the occurrence of the identified stressor, e.g., the presence of developed land cover in the buffer surrounding a wetland is highly correlated with hydrologic modification.

This three-step approach (selection of an ecosystem service, determination of the stressors most likely to impact performance, and identification of landscape characteristics that are indicative of stressor occurrence) differs from the more general multi-level approach, as previously discussed, in two primary ways. First, the Level 1 Landscape Analysis provides a relative measure of individual ecosystem service provision instead of general condition. The second is that the VIMS Level 2 rapid assessment and Level 3 intensive assessment do not serve as individual measures of condition, but serve only to inform and validate the Level 1 Landscape Analysis. The result of the VIMS approach, as applied in Virginia, is a census-level assessment of mapped nontidal wetlands (approximately 222,000 wetland units) for water quality and habitat service by watershed, utilizing a GIS-based analysis of remotely sensed information. This information is directly applicable to status and trends reporting under CWA §305(b), and can be utilized in permitting programs to assess cumulative impacts to wetlands within watersheds.

Rapid Assessment (Level 2)

RAMs or Rapid Assessment Protocols (RAPs) are intermediate in intensity between remote, landscape approaches, and intensive site sampling. Rapid assessments are based on easily observable structural indicators at a site, and take, as defined by Fennessy et al. (2007a), less than a 4-h site visit by two people to assess wetland condition. They can be advantageous in implementing M&A programs because they require less time in the field and less taxonomic expertise than do comprehensive assessments, leading to substantial savings in costs and providing the opportunity to increase sample sizes. The structure of RAMs vary, ranging from methods such as the Penn State Rapid Assessment, which is based on stressors and buffer characteristics, to those like the Ohio Rapid Assessment Method (ORAM) that are made up of a combination of indicators based on wetland form and structure and of stressor checklists used to inform the user about causes of degradation. Methods are designed either to provide a single, integrative score to indicate condition or to provide estimates of a suite of wetland functions. The treatment of wetland types varies; many methods are suitable for use in all HGM classes while others have different versions of the method specific to each class. In all cases, RAMs must be calibrated using data collected at a set of reference wetlands, and they must be validated using results of intensive assessments to assure that the results are ecologically robust (Fennessy et al. 2007a).

RAMs have been used effectively in both surveys of ambient condition and as a means to implement regulatory programs. For example, Ohio used an assessment approach combining the GRTS probabilistic sampling design with existing rapid assessment tools, including ORAM and the Penn State Rapid Assessment, to evaluate the ecological condition of wetlands in the 1,300 km^2 Cuyahoga River watershed.

In an 8-week summer field season, four field crews sampled over 250 sites and generated a "report card" of ambient condition (Fennessy et al. 2007b). Alternatively, ORAM was developed in the first instance as a tool for making regulatory decisions for the purposes of implementing wetland water quality standards and establishing mitigation ratios for wetland impacts. Development of rapid assessment methodologies by a number of states has continued at a rapid pace since Fennessy et al. (2007a, b) provided a review of six individual methods, because they can be modified to suit an individual state's needs (e.g., ORAM for assisting in the establishment of mitigation ratios) and they provide a relatively low cost entry into wetland condition assessment.

Intensive Assessment (Level 3)

Ecological integrity is often assessed by documenting the state or rate of ecological processes such as productivity, respiration and/or the structure of biological communities (Fennessy et al. 2007a; Smith et al. 1995). This can be accomplished by either measuring those processes (such as primary productivity) directly or through the use of indicators (such as the metrics composing IBIs as descriptors of community structure).

Bartoldus (1999) prepared a manual describing and evaluating 40 wetland assessment procedures developed in the United States over the preceding 30 years, and USEPA updated it in a series of documents describing a variety of approaches to assessing the ecological integrity of wetlands (e.g., USEPA 2002). Additionally, methods are further delineated into those that provide one measure of the status of site (i.e., condition assessment) vs. those that may provide function-by-function measures (i.e., functional assessment). Both approaches evaluate the ecological integrity of individual wetlands by comparing the results of the assessment to the values found in an established set of reference wetlands, seek to maintain wetlands in their minimally disturbed conditions, and make only within-type comparisons. A number of assessment methods of either type are available. Biological assessments have been utilized widely as the basis for state assessments of condition (e.g., Ohio, Maine) while functional approaches have been more commonly used at basin scales, perhaps because of their roots as a regulatory approach in Army Corps of Engineers project assessments (e.g., Willamette Valley, Oregon; Columbia Basin, Washington; Wardrop et al. 2007b; Whigham et al. 2007).

Condition Assessment

Condition assessments are rooted in the notion of ecological integrity, which can be estimated using Level 3 approaches such as IBIs as well as Level 2 RAMs. IBIs are multimetric indexes focused on a specific taxonomic group (vascular plant communities, invertebrates, algae) that quantitatively assess change in the structure and composition of those communities that result from anthropogenic disturbance

(Burton et al. 1999; Mack 2007; Miller et al. 2006). Condition describes the extent to which a given site departs from the full measure of ecological integrity that is possible in a region, which is defined by the least-impacted or reference condition. It can be measured in terms of structure (for example, the types and abundance of organisms, which are affected by the ecosystem processes in which they are involved), or form (the arrangement of ecosystem components, which helps define how they interact).

Because the range of possible metric scores and the expectations for condition vary by wetland class, the HGM approach to classification is often used to group sites, making the comparison of scores more equitable (Stevenson and Hauer 2002; Mack 2007). Condition assessments combine multiple metrics into a single score to represent the status of a site, typically by the simple addition of the metric scores, thus providing a measure of where the wetland sits on the scale between full ecological integrity and highly impacted (poor condition). Scores in themselves have no absolute value, but allow comparisons to be made between sites, enable the compilation of the distribution of condition scores by wetland type on a watershed or regional basis, can be combined with the landscape profile for that region to produce a profile of condition (Kentula 2007), and can be used to establish performance standards, for example for mitigation projects. Ultimately, as a site deviates from reference condition, the provision of the ecosystem services that are typical of that HGM class is altered, although methods to quantify the relationships between condition and services are currently lacking.

Functional Assessment

The HGM Functional Assessment is a recent advance in wetland assessment protocols, allowing the estimation of ecological functions associated with wetlands of various types on a wetland-by-wetland basis. The method requires three steps: specification of the wetland type (classification), the recognition of the functions associated with the specified wetland type, and the estimation of the level of functioning (functional assessment). HGM functional assessment uses a suite of mathematical models to estimate the magnitude at which a wetland performs a suite of ecological functions associated with a specific wetland subclass (Smith and Wakeley 2001). Assessment at the site level allows for nesting and characterization at larger spatial scales such as a watershed (Wardrop et al. 2007b; Whigham et al. 2007). HGM assessments are developed regionally and require significant field data collection, and so are available for limited areas of the United States (Kentula 2007) and have not been utilized on a widespread basis. It is important to note that the HGM functional assessment method is assumed to provide a reasonable approximation of functional capacity. Functional assessment models rely heavily upon structural measurements, with a sometimes-tenuous connection to real function (Cole 2006). The connection is generally most tenuous for hydrology and biogeochemical functions because they are difficult to validate. The few studies that are available to relate HGM model results to quantitative measurements of function show varying success of the models to estimate function (Jordan et al. 2007).

Riparia at Penn State began efforts to produce a regional HGM classification, reference system, and functional assessment in 1993, following the guidance of Brinson (1993). Over a 10-year period, 222 reference wetlands were characterized, and the data were used in the construction and calibration of a suite of ten functional assessment models (Chap. 2). The use of these functional assessment models has been featured in numerous studies (e.g., Brooks et al. 2006; Miller et al. 2006; Wardrop et al. 2007b, 2011).

11.4 M&A in Action—Examples of Applications in the MAR at Multiple Scales

The ability to assess wetland condition with a range of resource investment has greatly increased the implementation of assessments over a variety of spatial scales. Wetland condition assessments are dependent upon either a complete census or a probability-based sample that allows estimates of the entire population of interest. Techniques such as landscape assessments (Level 1) can allow an estimate of wetland condition at larger scales (e.g., for all wetlands in a watershed or basin), due to the availability of remote sensing data and the ability to perform such desktop analyses. Thus, wetland condition can be expressed at the watershed scale as a distribution of the values for all individual wetlands in the area being assessed (Wardrop et al. 2007a).

RAMs (Level 2) can be similarly used, although the increased effort required for the field work means that, in general, a complete census of all wetlands in a watershed is not feasible. However, use of a probability-based design to select wetlands that can be assessed for a statistically valid estimate of the total population, in conjunction with a rapid technique, has led to widespread use of rapid assessments to provide condition estimates on a watershed or basin scale (Stein and Ambrose 1998). The probability-based design also allows assessment of condition at the national scale; the National Wetlands Condition Assessment in 2011 (http://water.epa.gov/type/wetlands/assessment/survey/index.cfm) assessed condition through intensive assessment at approximately 1,000 sites, providing an expression of wetland condition at regional and national scales.

11.4.1 The Site-Level

Application of M&A at the site level is perhaps the most common, and can provide information relevant to a wide range of site-level decisions including permitting, restoration, mitigation, and protection. One of the most powerful uses has been the ability to assess mitigation sites and natural sites while utilizing the same methods, allowing us to compare the former to the latter. Penn State's efforts to develop M&A tools that were appropriate to this specific task began as early as 1993 with the establishment of a set of reference wetlands that had the primary intent of collecting the data necessary to improve wetland mitigation design and performance (Brooks et al. 2002, 2004, 2006). Once the commitment was made to establish a reference set, wetlands were added annually for a decade by securing funds from a variety of sources, and were utilized to develop assessment tools for all three levels of effort—Landscape, Rapid, and Intensive (both HGM and IBI approaches). Further discussion and guidance for creating a reference set of wetlands is covered in Chap. 2. By the late 1990s, a full suite of tools was available that could be utilized to compare natural and mitigation sites.

It is critical to use the same methods and protocols to assess mitigated and restored wetlands as those used to characterize naturally occurring reference wetlands; only then can the data be comparative, and useful in advancing the practice of mitigating wetlands. In addition, one needs reference data from an array of wetland types such that an appropriate set of data can be used to compare "apples to apples." Mitigation and restoration projects should be designed to mimic the characteristics of a particular type, presumably the same as the type of wetland being replaced. In some situations, a decision may be made to create a wetland corresponding to another type, perhaps to replenish the excessive loss of that type from a watershed. In either case, the target wetland type should be designated so that any studies of performance will use data from a matching reference type.

By utilizing the same assessment methods and protocols, Gebo and Brooks (2012) were able to show that mitigation and restoration projects in Pennsylvania, even those from mitigation banks, were performing at levels of function significantly below that of natural reference wetlands of the same type. As described in Chap. 12 and in Gebo and Brooks (2012), despite repeated calls over the past two decades to improve the design and performance of mitigation projects, only incremental improvements have occurred. By working with USEPA, PADEP, and agencies of other MAR states, Riparia at Penn State has assembled an interactive database of reference wetlands data, searchable by ecoregion, state, and HGM type (http://www.riparia.psu.edu). The intent is to provide practitioners with essential measurements that will assist in designing mitigation projects that more closely align with reference wetlands of the same type, and to provide suitable benchmarks for evaluating performance and success. Within an M&A program, using data from reference wetlands in the manner described here can bring us much closer to replacing wetland area and function in-kind.

11.4.2 Watershed-Level: The Upper Juniata Watershed

In 1998, USEPA scientists from both Region 3 and the Western Ecology Division collaborated to sponsor the first assessment of wetlands at the watershed scale utilizing an EMAP approach. The work was intended to serve as a scalable and transferable model of wetland assessment that would, hopefully, make wetland monitoring routine (Kentula 2007). Two watersheds in the MAR were selected, the Upper Juniata located in the Ridge and Valley physiographic province, and the

Nanticoke located in the Coastal Plain. A tremendous amount was learned during both assessments, as presented in a special feature of the journal Wetlands in 2007.

While the case for monitoring wetlands on a watershed basis had been strong, attempts to institutionalize it had been almost nonexistent because (1) methods for the assessment of wetland condition that are easily implemented and scientifically defensible had been lacking; (2) it was not clear how to obtain a representative sample of wetlands on a watershed basis, given the heterogeneous distribution of the resource and uncertainties in gaining access; and (3) the cost had been perceived as inordinately high (Wardrop et al. 2007a). The availability of a probability-based survey approach, HGM classification and functional assessment, condition assessment, and a three-tiered approach (all described in previous sections) came together to address these deficiencies. Since these pieces have already been described, the results of their application in the Upper Juniata are of relevance here, as an illustration of what can be learned and gained by their application at the watershed scale (details can be found in Wardrop et al. 2007a, b). Three primary points are discussed: the extent and character of the resource, the use of multiple tiers of assessment to inform one another, and the interplay of landscape and site-specific factors in the interpretation of functional assessment results.

The application of the GRTS design, along with the Landscape and Rapid Assessments, provided the first description of the wetland resource and its ecological condition in the Upper Juniata. One of the first questions regarding the resource was, quite simply, how much wetland acreage was present in the watershed, and of what type. Previous work showed that NWI may miss over half of the smaller wetlands in forested portions of the watershed, resulting in significant under-coverage, and we were interested in testing a method that might supplement the NWI. We utilized geologic structural and stratigraphic information in combination with floodplain maps to generate a map of areas with high probability of wetland occurrence; these high probability areas were sampled in conjunction with the NWI. The result was the first statistically-determined difference in wetland area predicted by the Riparia and NWI maps. Total wetland area in the Upper Juniata watershed was estimated as 2,123 ha (95% c.i. = 1,743, 2,503) using the 81 points from the Riparia map that had wetlands. By comparison, the total wetland area calculated from the NWI map was 1,144 ha. A primary outcome of the assessment was the quantitative confirmation of the under-representation of the resource by the NWI; namely, the addition of the Riparia map to the site selection process increased the estimate of wetland resource in the Upper Juniata almost twofold.

Since the Upper Juniata provided us with the first opportunity to utilize all three tiers of assessment at a significant number of sites (83), we were interested in how results from each tier of assessment could be used to inform the others. Thus, we used one of the components of the intensive assessment to calibrate and refine the landscape and rapid assessments. Specifically, we used the results of the FQAI to illustrate how the evaluation could be done because it had proven to be a reliable measure of wetland condition (the FQAI is described in detail in Chap. 6). Classification and regression tree (CART) analysis was used to evaluate (1) whether the results of the landscape and rapid assessments correspond to those from the intensive assessment (i.e., do they group sites along a condition gradient supported by ecological data) and (2) whether the categories of condition based on the results of the landscape and rapid assessments (four condition categories had been established) align with categories of condition specified by quantitative ecological data from the intensive assessment. In general, the analysis showed that both the landscape and rapid assessments assign sites to the highest and lowest categories of condition, but sites in the middle have a limited range of FQAI values that do not as clearly define groups. The CART results also indicated that our initial delineation of condition categories for both the landscape and rapid assessments should be more stringent (Table 11.1). For example, our highest condition category from the landscape assessment had been defined as sites with greater than 85% forested cover in a 1-km radius circle surrounding the site, and the CART results indicated that the highest condition was present at sites with greater than 89% forested cover. When both the landscape and rapid assessment scores were used as predictor variables, CART chose the rapid assessment results over the results of the landscape assessment, indicating that the rapid assessment better explained the variation in the response variable (FQAI) than the landscape assessment. This result is notable, since it demonstrates how the level of confidence in the results increases as one changes from a fairly general landscape assessment to more quantitative assessments (i.e., one looks "under the trees").

Finally, we employed the family of HGM functional assessment models (as described in Chap. 2) to provide a measure of the potential functional performance of a single wetland for up to 11 functions, depending on the subclass. Performance of each function is expressed by a Functional Capacity Index (FCI) score ranging between 0 and 1. A score of 1 indicated the site was performing the function at levels comparable to reference standard; a score of 0 indicated the site was not performing the function. We then reported on the distribution of FCI scores across all wetlands in the watershed by constructing cumulative distribution function (CDF) plots for the wetland population (all sampled wetland types), as well as individually for the slope and riverine classes. CDF plots allow estimation of what percent of the wetland area of the population is less than or equal to a particular FCI score. The CDF plots for the entire wetland resource (Fig. 11.4) are fairly linear over most of the distribution for all functions, indicating that the FCI scores are evenly distributed over the population. Several of the plots flatten at the upper and/or lower ends of the curves indicating that a very small proportion of the wetland area had the highest and lowest scores. However, the CDFs can also be utilized to assess whether the results of an individual functional assessment model are in agreement with the results of the landscape and rapid assessments. For example, the range of FCI scores for Characteristic Hydrology in the Upper Juniata wetlands is 50% of reference standard or higher (Fig. 11.4). This result was at odds with the Rapid Assessment findings that hydrologic alteration was a common stressor in the watershed, affecting on average 53% of the resource (Wardrop et al. 2007a, b). Either the hydrologic alterations did not affect the hydrologic functioning of the wetlands or, more likely, the model does not detect the likely effects. Findings such as these indicate that selected functional models require reassessment and revision.

Table 11.1 Ecological condition of the wetland resource in the Upper Juniata watershed based on the landscape assessment ($n=463$) and on the rapid assessment ($n=80$) and using the condition category definitions generated by the CART analysis

Condition category	Landscape assessment score	Percent of resource	Area of wetland (ha)	Rapid assessment score	Percent of resource	Area of wetland (ha)
Highest	≥89% FLC	3.6 (2.4, 4.8)	76 (51, 101)	≥87	2.9 (0, 5.8)	62 (0, 124)
High	≥82% and <89% FLC	4.0 (2.6, 5.4)	85 (55, 115)	≥58 and <87	28.8 (17.9, 39.7)	611 (380, 842)
Medium	≥74% and <82% FLC	12.5 (9.6, 15.4)	265 (203, 327)	≥42 and <58	14.3 (7.0, 21.6)	304 (149, 459)
Low	≥47% and <74% FLC	36.5 (32.8, 40.2)	775 (696, 854)	<42	54.0 (42.9, 65.1)	1,146 (910, 1,382)
Lowest	<47% FLC	43.4 (39.3, 47.5)	922 (835, 1,079)			

Values are means followed by the 95% confidence interval in parentheses. *FLC* Forested Land Cover. From Wardrop et al. (2007a)

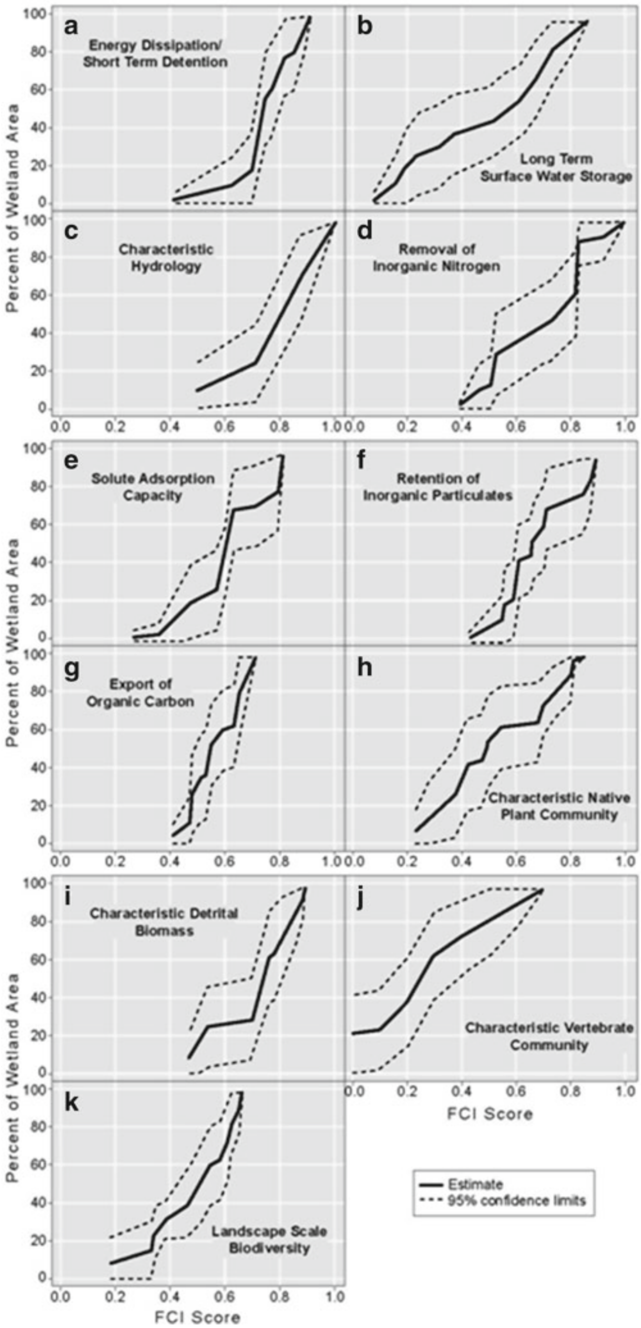

Fig. 11.4 Distribution of hydrogeomorphic (HGM) Functional Capacity Index (FCI) scores across all wetlands in the Upper Juniata watershed using cumulative distribution function (CDF) plots (Wardrop et al. 2007b)

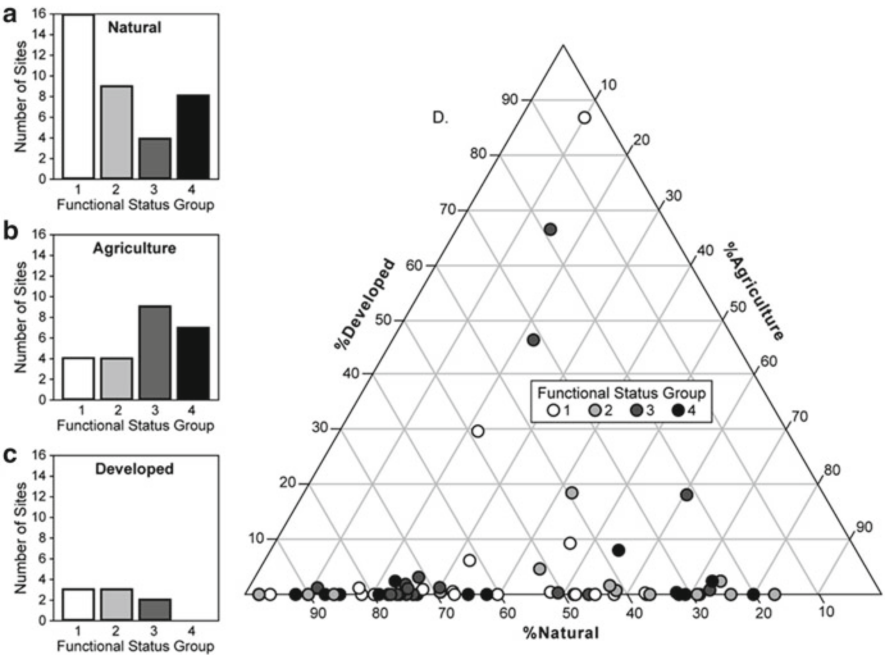

Fig. 11.5 Distribution of the 69 riverine and slope sites in the Upper Juniata watershed by Functional Status Group (FSG) within the (**a**) natural, (**b**) agriculture, and (**c**) developed reference domains. (**d**) Shows the relationship between FSG and percent land cover. Each circle represents one site. Reproduced from Wardrop et al. (2007b)

Finally, we were interested in whether clear groupings of sites with similar functional score profiles were present (e.g., a group of sites exhibiting high FCI scores across all functions), and if these groups were correlated with surrounding land cover classes. The FCI scores for the Characteristic Plant Community, Detrital Biomass, and Vertebrate Community Functions were chosen for the analysis because they either represented functions that were measured directly or had been validated. Clustering of the 69 riverine and slope sites for these three functions resulted in the formation of four Functional Status Groups (FSG) representing combinations of high, medium, and low mean FCI values for the three functions. Groups 1 and 2 represented a relatively high-functioning group of sites, but were differentiated by an exceptionally high Plant Community Function in Group 1 that differs significantly from the low value for the same function in Group 2. FSGs 3 and 4, with a combined total of 30 sites, represented a relatively low functioning group of sites and were differentiated by a significantly high Vertebrate Community Function in Group 3.

We used a ternary plot to visually represent the sites relative to their land cover setting (Fig. 11.5); these diagrams have three axes, one each for the percentage of agriculture, developed, or natural land cover surrounding the site. The figure clearly shows that sites of any given FSG are distributed across a variety of land cover compositions. We took a closer look at sites within a given category of surrounding land cover by establishing land cover-based reference domains, *sensu* Brinson and Rheinhardt (1996). Sites with surrounding land cover of >50% natural cover are termed "Natural," sites with >50% surrounding agricultural cover are termed "Agricultural," and all remaining sites are termed "Developed."

Sites of all FSGs appear in each reference domain, with some notable differences in distribution (Fig. 11.5). Sites in the Natural Domain are much more likely to be in the higher functioning FSGs, while sites in the Agricultural Domain are dominated by sites in FSGs 3 and 4, with an overall low level of functioning. Sites in the Developed Domain are equally distributed across FSGs 1, 2, and 3. What is surprising about this result is the realization that surrounding land cover does not completely control functional performance. The information obtained during the rapid assessment proved to be a valuable diagnostic tool because of the inclusion of information on the quality of the buffer associated with the sites as well as the stressors present. For example, what distinguishes a site in FSG 1 (highest level of functioning) and in developed land cover is the fact that it has an intact buffer. This type of information has potentially significant utility in restoration and management, since it provides a template of a high-functioning site that does so in spite of its context.

11.4.3 Regional-Level: The Mid-Atlantic Regional Wetland Assessment

As per Fig. 11.2, MAWWG decided to embark on a regional condition assessment in 2007, with funding from USEPA. The decision was a result of a number of factors, including: (1) the desire to be a regional pilot for the National Wetland Condition Assessment (NWCA) (described in the following section), (2) the management utility of a landscape and stressor profile for the entire region, as well as each individual state, (3) the opportunity to build state capacity in the various M&A tools, and (4) the construction of an assessment protocol that could be applied across the region and subsequently adopted by states, if appropriate. This project, which is in the final stages of analysis, used a combination of tools that had been developed by a number of the states and academic partners. The VIMS Landscape Assessment, which results in an estimate of potential water quality and habitat ecosystem service and was described above, was performed on all NWI polygons for freshwater wetlands in the MAR (about two million sample points). A RAP that could be applied across the entire MAR was developed using a synthesis of the Delaware, Penn State, and VIMS approaches and was applied at approximately 400 points obtained using the GRTS design (Fig. 11.6). The MAR rapid assessment was designed to provide a regional landscape profile, various stressor profiles, and an assessment of condition.

Following a training session to help standardize field methods, two field teams of two or three persons per team, one from Riparia and one from VIMS, conducted the sampling. Field sampling was conducted throughout the region during two summers, 2008 and 2009. Each of the field sites consisted of a wetland assessment area with a 40-m radius circle, surrounded by a 100-m buffer. Our goal was to have a

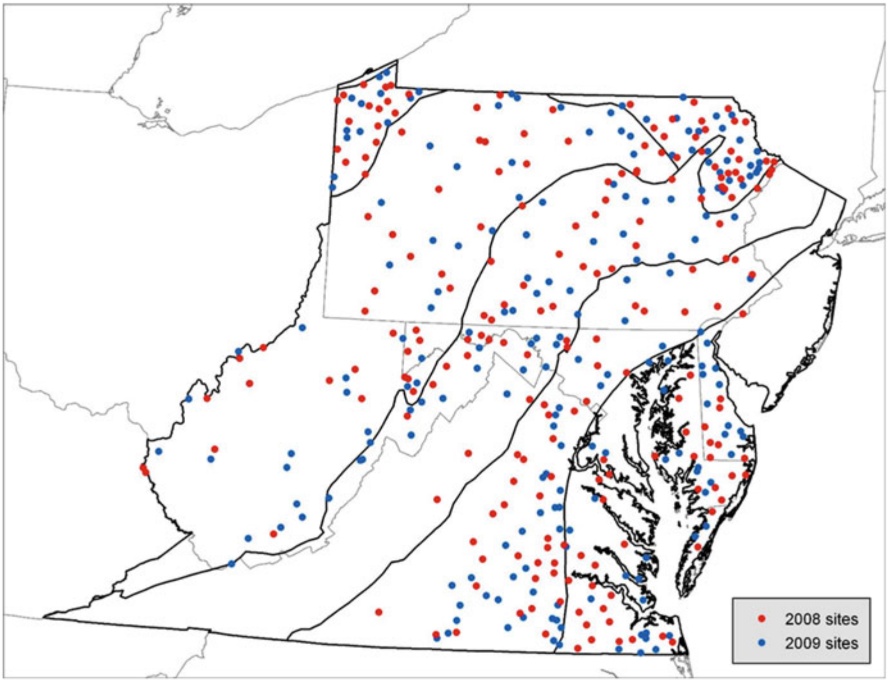

Fig. 11.6 Spatially balanced sample across the Mid-Atlantic Region for a condition assessment of wetlands conducted in 2008–2009 using the Unified Mid-Atlantic Rapid Assessment Protocol (Brooks et al. unpublished)

sufficient number of assessed sites in each of the five major ecoregions (80 sites in each ecoregion), and where possible, make comparisons across the more common wetlands types (e.g., riverine). The landscape profile (Fig. 11.7) shows that riverine wetlands dominate HGM types across the MAR. A simple tally of stressors recorded from the wetland assessment area shows that ecoregions are being affected differentially (Fig. 11.8). Results of this study will be posted at http://www.riparia.psu.edu/MARbook when available.

11.4.4 National-Level: The National Wetland Condition Assessment

The NWCA is part of the USEPA's National Aquatic Resource Surveys (NARS). The 2011 NWCA is the first national assessment of wetlands and the fifth in a series of NARS assessments, after streams, rivers, lakes, and coastal systems. The assessments will be conducted every 5 years, resources permitting, to report to Congress and the nation on trends in the condition of the nation's aquatic resources.

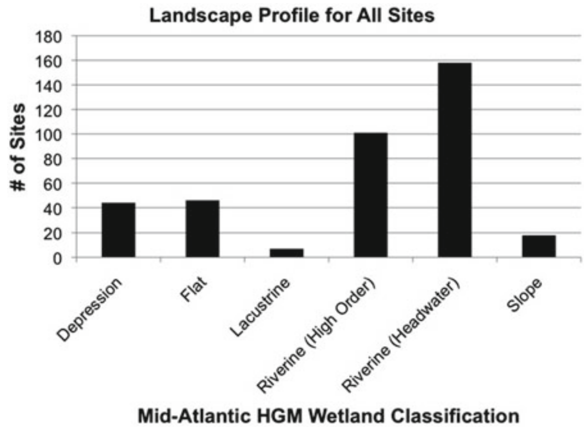

Fig. 11.7 Landscape profile of HGM wetland types provides a representative distribution of freshwater wetlands for the MAR

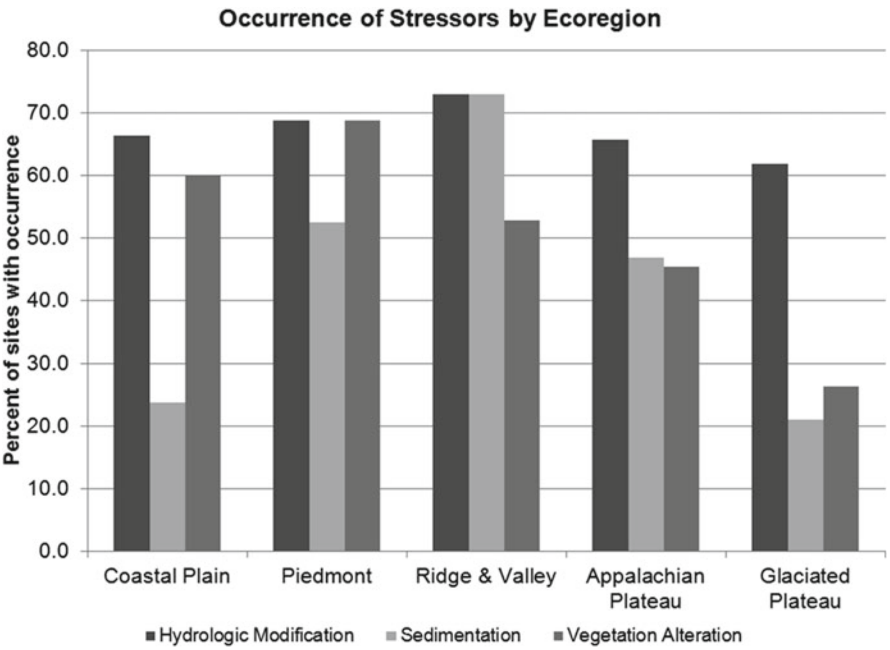

Fig. 11.8 Percent of sites with occurrence of stressors (hydrologic modification, sedimentation, vegetation alteration) by ecoregion in the MAR

The NWCA was designed to build on and augment the achievements of the USFWS's status and trends (S&T) reporting which characterizes changes in wetland acreage across the conterminous United States (e.g., Dahl 2011). Paired together, the NWCA and S&T reporting will provide the public and government agencies tasked with the management of natural resources with comparable, national information on wetland quantity and quality (Scozzafava et al. 2011). The NWCA is designed to produce detailed information on wetland quality by wetland type and region of the United States, providing insight into the implications of the changes in area reported by the USFWS S&T program. An overview of the NWCA and a presentation of frameworks for reporting the results in the context of the other NARS assessments are found in Kentula et al. (2011).

The fact that wetlands were added to the NARS is due in no small part to M&A efforts in the Mid-Atlantic as described in the sections above. In particular, it's interesting to note that the relationship between USEPA's Mid-Atlantic Regional Office in Philadelphia, Pennsylvania, and the wetland programs of the Mid-Atlantic States that led to the formation of the MAWWG also was instrumental to developing the science needed for implementation of the NWCA. Art Spingarn of USEPA's Regional Office was the person who made the assessments of the Upper Juniata and Nanticoke watersheds happen. Spingarn built support with the wetland managers from the states of Delaware and Pennsylvania, interacted with the scientists who would conduct the studies, protected the funding from attempted cuts, provided technical review, and did everything that needed his skills and attention to assure that the assessments were done. The assessments of the wetland resources in the Upper Juniata and Nanticoke watersheds and the subsequent assessment of the MAR convincingly demonstrated that the wetland scientific and management communities could cooperate to conduct an assessment of wetland condition at large scales and were ready to take on the challenges of planning and implementing the first NWCA.

Components of the NWCA were the same as tools developed for MAWWG and being used by the state wetland programs in the MAR. The following brief description of the NWCA details how those tools were used in the 2011 assessment.

The 2011 NWCA sample design was linked to the design used for the S&T reporting to assure that comparable information on wetland quantity and quality is produced. Both efforts used an ecological definition of wetlands, specifically:

> *Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. Wetlands must have* ***one or more*** *of the following three attributes:*

- *at least periodically, the land supports predominantly hydrophytes;*
- *the substrate is predominantly undrained hydric soil; and*
- *the substrate is non-soil and is saturated with water or covered by shallow water at some time during the growing season of each year (Dahl 2006).*

The target population was defined as: Tidal and nontidal wetlands of the conterminous United States, including certain farmed wetlands not currently in crop production. The wetlands have rooted vegetation and, when present, open water less

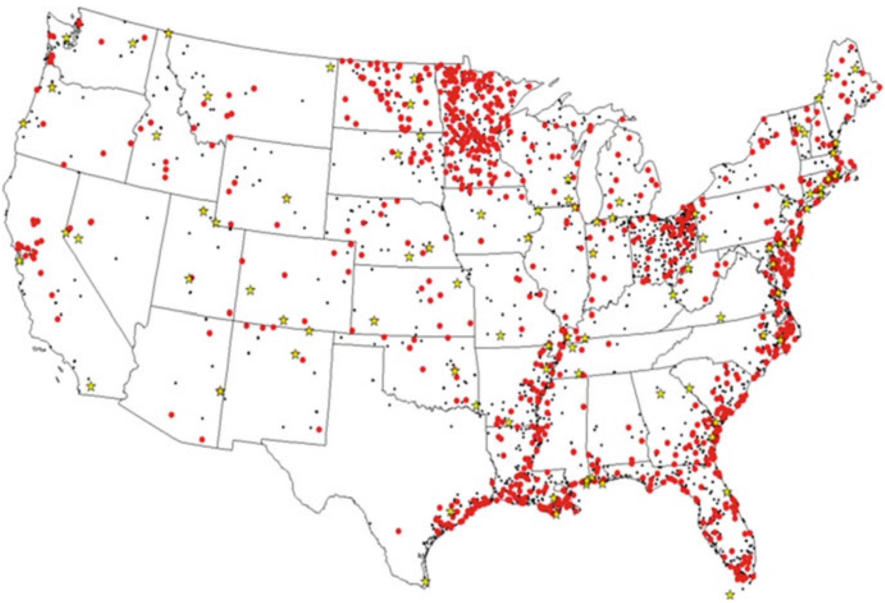

Fig. 11.9 Map of the conterminous United States showing all the points from the NWCA sample draw. • = the primary sample points; • = oversample points (for use if the primary points are not sampleable) ☆= revisit sites (primary sites that are resampled for quality assurance purposes)

than 1 m deep (USEPA 2011a). The Target Population is composed of seven of the wetland classes used in the S&T reporting, i.e., Estuarine Intertidal Emergent, Estuarine Intertidal Forested/Shrub, Palustrine Forested, Palustrine Shrub, Palustrine Emergent, Palustrine Unconsolidated Bottom/Aquatic Bed and Palustrine Farmed. The classes are a modification of the system developed by Cowardin et al. (1979).

A spatially balanced probability survey design was used (Stevens and Olsen 1999, 2000, 2004) to generate enough sample locations (hereafter points) to assure a target sample size of 900 (Fig. 11.9). The 2005 USFWS S&T sample plots augmented for better coverage on the Pacific Coast were used as the sample frame. Specifically, the frame was composed of 4-m^2 plots containing mapped wetlands, deepwater habitat, and uplands. Points were drawn from the wetland areas.

The NWCA was designed so that wetland condition could be reported by wetland type for the nation and by aggregated ecoregions based on the Omernik Level III Ecoregions (Omernik 1987; USEPA 2011b). USEPA Regions and major river basins are among the additional reporting units being considered. The ability to report on additional geographic units will depend on the number of sites sampled per unit.

A definition of reference condition is used to quantitatively describe the standard or benchmark against which to compare the current condition measured in the assessment (Stoddard et al. 2006). The NWCA, as done in previous NARS assessments, has defined reference as least disturbed and good condition as greater than or

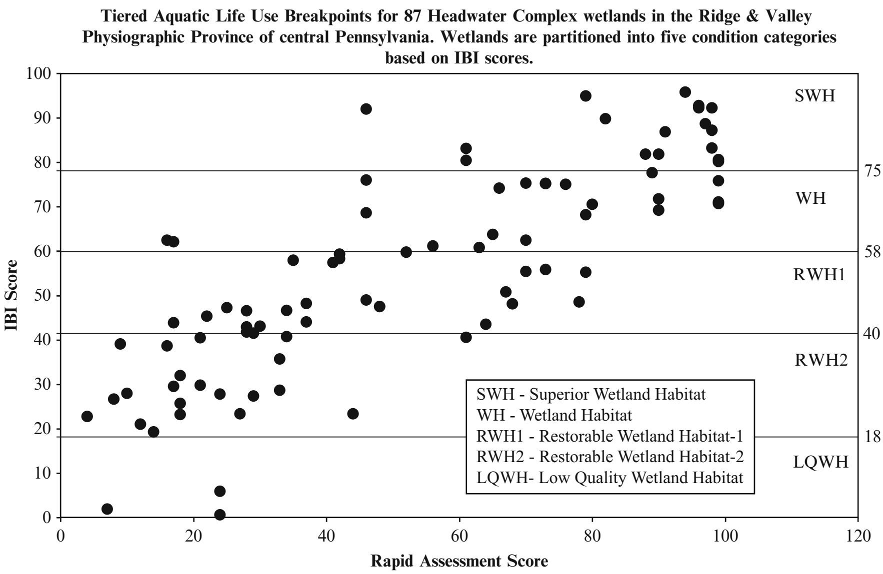

Fig. 11.10 Potential future use of M&A data in the construction of Tiered Aquatic Life Uses (TALUs)

equal to the 25th percentile of values observed in the reference population (USEPA 2006b, 2009). Candidate reference sites were recommended to the NWCA by the States, the National Estuarine Research Reserve System, the National Park Service, the USFWS National Wildlife Refuge System and the US Forest Service. The 1,141 candidate sites were evaluated using a series of screens that used aerial photography and digital land cover data to identify sites that were part of the target population, able to be sampled, and likely to be least disturbed. This resulted in 150 recommended reference sites distributed across ecoregions and wetland classes targeted for sampling in the 2011 NWCA. Identical field sampling and laboratory protocols were used for the recommended reference sites and 900 probability sample points. Additional screening will be performed post field sampling to assure that the field data collected support the pre-sampling evaluation of least disturbed and to identify any sites from the probability sample meeting the definition of reference (e.g., see the screening process described in Herlihy et al. 2008).

The NWCA used all components of the three-tiered approach. As described above, a landscape assessment was employed to screen potential reference sites and to evaluate points as to suitability for sampling (e.g., was the wetland part of the target population) (USEPA 2011a). A rapid assessment was developed for national application and used in 2011 to provide data for an initial evaluation of its performance. The primary data set for the NWCA was generated using an intensive assessment. NWCA protocols were designed to be completed by a four-person field crew during one day in the field. The field crew sampled a 0.5-ha assessment area (AA) and an area immediately adjacent to the AA (i.e., the buffer). The indicators used and a brief description of the sampling approach is presented below; the detailed protocols used in the 2011 NWCA are found in the Field Operations Manual (USEPA 2011c).

- *Vegetation* was characterized by collecting plant data in plots systematically placed across the AA
- *Soils* data were collected in four soil pits and include an on-site description of the soil profile and collection of four types of soil samples (chemistry, bulk density, stable isotope, and soil enzymes) for laboratory analysis
- *Hydrologic* data included an assessment of hydrologic sources and connectivity, indirect evidence of hydroperiod, estimates of hydrologic fluctuations, and documentation of hydrology alterations or stressors
- When standing water was present in the AA, *water chemistry* samples were taken and analyzed for general surface water conditions, various chemical analytes, and evidence of disturbance
- *Algae* samples were collected from sediments (benthic samples) and from the surface of vegetation stems and leaves (epiphytic samples)
- The presence of *stressors* was measured in the AA and buffer

Reporting on the 2011 NWCA will follow the format established in the NARS Wadeable Streams and Lake Assessment reports (USEPA 2006b, 2009). The results of the assessments are presented in four categories. First, the extent of the wetland resource will be described in ways that will inform and augment the USFWS S&T reporting, in particular, the report for 2004–2009 (Dahl 2011). For example, the NWCA will document the frequency and location of S&T mapping errors. In addition, the NWCA will provide information on the occurrence and condition of various wetland types in the Palustrine Unconsolidated Bottom Class, especially those commonly known as freshwater ponds. Freshwater ponds had the largest percent increase in area nationally of any wetland type between 1998 and 2004 (Dahl 2006) and increased 3.2% between 2004 and 2009 (Dahl 2011). The resulting shift in wetland types from vegetated wetland to those dominated by open water can involve changes in ecological structure and function in the affected landscape (e.g., see Gwin et al. 1999; Magee et al. 1999; Shaffer and Ernst 1999; Shaffer et al. 1999; Magee and Kentula 2005). To better capture the nature of the of any changes associated with increase in area of ponds, S&T added descriptive categories to the 2004–2009 report (Dahl 2011), which were also tracked in the 2011 NWCA.

Second, the NWCA will report on the ecological condition of the nation's wetland resource. Vegetation is the NWCA's primary indicator of condition with algae and soil providing additional information. The vegetation data collected are suitable for the development of various condition indices, especially IBIs (e.g., Miller et al. 2006), an Observed vs. Expected index (e.g., Carlisle and Hawkins 2008; Van Sickle 2008), and a Floristic Quality Index (FQI) (e.g., Lopez and Fennessy 2002; Rooney and Rogers 2002; Mathews 2003; Bourdaghs et al. 2006; Miller and Wardrop 2006). Although Vegetation IBIs and FQIs have been developed for a number of states and regions and for a number of wetland types (see Mack and Kentula 2010 for a review); they are not available for the nation or for all states or regions and will require additional information and research for use in NWCA reporting.

Third, the NWCA will report on the area of the target population affected by biological, chemical, and physical stressors, thus recognizing the connection between the presence of stressors and wetland condition. The use of stressor data are consistent with current approaches to assessment. For example, some RAMs use only stressors as indicators of wetland condition (e.g., the Delaware RAM Jacobs 2007), the Penn State Stressor Checklist (Wardrop et al. 2007a) and models comprising an HGM intensive assessment use stressors as variables (e.g., Wardrop et al. 2007b; Whigham et al. 2007). The estimates of the relative extent of stressors generated from the NWCA are a measure of how common a stressor is and can be reported for the nation and ecoregions and by wetland class.

Finally, the NWCA will explore the relationship between stressors and ecological condition through the concepts of relative and attributable risk. Relative risk is an expression of the likelihood of having poor ecological condition when the magnitude of a stressor is high vs. low (Van Sickle and Paulsen 2008). Attributable risk provides an estimate of the proportion of the population in poor condition that could be reduced if the effects of a particular stressor were eliminated (Van Sickle and Paulsen 2008). An example from the Wadeable Streams Assessment (USEPA 2006a) illustrates how these measures of risk can illuminate subtleties in the M&A data. The relative risks for all stressors in the West region are consistently larger than for the nation and other regions while the extent of streams in poor condition are consistently lower. This suggests that although the stressors are less widespread in the West, the region's streams are particularly sensitive to the stressors detected.

The above description of the reporting anticipated from the 2011 NWCA demonstrates the comprehensive nature of the data that can be generated from M&A and suggests uses in resource management. Reporting on the extent and condition of the resource can be used to track effectiveness of regulation and management practices by geographic region and/or wetland type. Alternatively, the estimates of the extent of stressors can identify the emergence of new threats to wetland condition, while the use of relative and attributable risk helps to prioritize management actions by stressor, geographic region, and/or wetland type. The number of examples above coming from work done in the Mid-Atlantic is notable and the potential uses for NWCA data echo the objectives of the MAWWG.

11.5 Lessons Learned

The success of M&A in the Mid-Atlantic required innovations in three areas: technical tools, programmatic opportunities and applications, and partnerships. Lessons learned for each are:

Tools

- It would be difficult to imagine M&A without all of the technical advances of the last 20 years, most notably Mark Brinson's contribution of the HGM Classification. However, the point to be made is that no one tool was the basis for rapid progress in M&A; it was that the field allowed one tool to inform the development of another. For example, IBI and HGM functional assessment each had dedicated proponents, but clearly profited by the cross-pollination of concepts. The concept of reference substantially informed the construction of the stress gradient used in the development of IBIs; in turn, many of the condition assessment indicators (such as plant-based IBIs) became critical variables in functional assessment models

- The development of a robust collection of condition assessment tools, representing a range of resource requirements (e.g., three tiers of assessment) was critical for use and application. Users varied widely in both the resources available for assessment and the requirements for the precision of the resulting data, and the availability of tools to inform a myriad of decisions hastened the refinement of the entire toolbox. In addition, tools from each level could be used to refine each other, as has been evident in a number of efforts and discussed in this chapter

Programmatic applications

- The realization that condition assessment data could be broadly utilized across both wetland regulatory and/or non-regulatory programs led to a blossoming of uses. Mid-Atlantic States used condition assessment data for assessment of mitigation programs (Pennsylvania), permitting and cumulative impact decisions (Virginia), water quality standards (Maryland), restoration performance and design standards (Delaware), and protection (West Virginia). The willingness of the leadership of various state programs to utilize these tools played a key role in their use, which led to tool refinement, which led to broader usage
- Consistent use of the same M&A methods in various parts of a wetlands program leveraged the value of the information. For example, a landscape profile of wetland types obtained during condition assessment could be utilized as a template for wetland creation; assessment of mitigation wetlands utilizing the same protocol as that used for condition assessment of natural wetlands provided a fair assessment of the success of the mitigation projects

Partnerships

- Regional forums such as MAWWG allowed state programs to stand upon one another's shoulders, and allowed the development of both tools and their applications to proceed in a cost-effective manner. As evidence, many of the state's condition assessment tools resembled one another, and provided a regional consistency that ultimately led to the Mid-Atlantic Regional Wetland Assessment. Successes, as well as challenges, were freely shared, and states that had not developed certain tools could gain them rapidly since they could pick and choose parts of existing methods or adopt them in their entirety. In another dimension, the academic/regulatory partnership of Riparia and USEPA Region 3 (via the leadership of Regina Poeske) was reflected in the agenda of MAWWG meetings, which split meeting time almost equivalently between technical issues and programmatic ones
- Partnerships between MAWWG and external groups, such as USEPA's EMAP, were invaluable in providing a level of technical assistance that had been previously unavailable to state programs. For example, the investment of time

and expertise by both Mary Kentula and Anthony Olsen of USEPA resulted in numerous applications of the GRTS protocol to large, probability-based surveys in a number of states, as well as region-wide in the Mid-Atlantic Regional Wetland Assessment.

11.6 Future Uses of M&A

Most states in the MAR have begun moving away from program development into implementation of statewide monitoring and assessment programs. The focus for state wetland monitoring programs, therefore, has shifted from the development of tools to estimate wetland condition to integrating ecologically meaningful data into water quality standards. This translation of data into standards is the final step in institutionalizing state monitoring and assessment programs for wetlands.

Davies and Jackson (2006) developed a framework for interpreting the ecological condition of wetlands and other habitats from empirical data termed the Biological Condition Gradient (BCG). The BCG provides a common foundation for states to develop water quality standards for wetlands in the form of tiered aquatic life uses (TALUs) (US Code title 33, section 1,251 (b), 1,313) with each tier corresponding to a different level of ecological integrity.

In cooperation with the PADEP, Riparia developed a prototype of a BCG for Headwater Complex Wetlands in the Ridge and Valley Physiographic Province. Using vegetation data collected from 87 wetlands, five tiers of habitat condition were described: SWH, WH, RWH1, RWH2, LQWH (see below). CART analysis of IBI scores (Miller et al. 2006) for each wetland was used to derive breakpoints, thus providing a quantitative basis for each tier. Once tiers are established and codified, prescriptive measures can be promulgated to protect the highest tiered wetlands, prevent further degradation of wetlands in the middle tier, and restore the functions and values of those in the lowest tiers.

Thus, the knowledge treads and efforts of the past three decades have coalesced into a varied tapestry composed of a plethora of assessment tools, programmatic applications, and partnerships generated from a common objective to improve protection and management of the wetland resource based on sound science. The anticipated future of the monitoring and assessment of wetlands promises a rich and exciting design to come.

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Chapter 12
Wetlands Restoration and Mitigation

Robert P. Brooks and Naomi A. Gebo

Abstract For decades, scientists, managers, policy makers, and practitioners have sought to improve the design and performance of mitigated and restored wetlands. Progress has been made, but further improvements are needed. In this chapter, we provide a historical context, review the mitigation process, summarize the literature on mitigation and restoration of wetlands, and make the case for using natural reference wetlands as templates for designing mitigation and restoring projects and assessing their performance. Two case studies conducted by Riparia at Penn State are used to demonstrate the value of a reference-based approach. A comparison of scores from Habitat Suitability Index models between reference and created wetlands shows that the latter are either not equivalent, with created sites scoring lower, or habitats are shifted toward species in the wildlife community that favor open water or emergent conditions. In the second study, scores of hydrogeomorphic (HGM) functional models are compared between reference wetlands and mitigation sites, showing that average performance is often significantly lower for several functions across multiple HGM types. Finally, we describe how a set of variables from Riparia's database of reference wetlands can be used to improve the outcome of mitigation and restoration projects.

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12.1 Introduction

Striving to improve the *performance* of wetland projects has been a goal of *mitigation*, *restoration*, *creation*, *construction*, and *enhancement* efforts since the inception of these practices (see Sect. "Glossary" for definitions of underlined terms). The call to improve the performance of mitigation projects began in earnest with the release of The Conservation Foundation's report on Protecting America's Wetlands: An Action Agenda in 1988, which recommended both a no net loss policy for existing wetland area and function, and a long-term gain in wetland area and function. This report also stated the need for developing technical guidance for designing and replacing wetlands and their inherent functions, but for years, the "no net loss" portion has been applied to acreage only (not function), and the "gain" portion of the recommendations has not been effectively applied to either acreage nor function.

The state-of-the-science in wetlands restoration and creation was summarized in an edited volume by Kusler and Kentula (1990). This was soon followed by the National Research Council's report (1992) that called attention to the gaps in our knowledge about restoring wetlands and other aquatic ecosystems. A variety of works aimed at guiding practitioners on how to "build a better wetland" (Cole et al. 1997) followed, such as Hammer (1992) and Marble (1992). Yet, the focus of these and most other publications was on the creation, restoration, or enhancement of many freshwater, emergent marshes, for which *design* and construction techniques are well established (e.g., Cole et al. 1996). Thus, the majority of wetlands were of this type, whether they were built as mitigation projects, as *voluntary*, incentive-driven projects on private lands, or as wildlife enhancements designed and constructed by conservation organizations, such as Ducks Unlimited (www.ducks.org).

What was obviously needed was a process by which more in-kind replacement could be proposed and designed. To achieve this, two things were necessary: a classification system that had a functional basis, and a process by which one could recognize relevant models for restoration or creation. These needs were met by a series of papers such as Brinson and Rheinhardt (1996), which recommended the study of comparable, natural, reference wetlands to guide the process of designing and constructing mitigation projects. The concept was that wetlands classified differently either by their hydrogeomorphic (HGM) characteristics (i.e., water sources, hydrodynamics, landscape position, Brinson 1993) or their vegetation characteristics (e.g., aquatic bed, emergent, shrub, trees, Cowardin et al. 1979), would vary in their design parameters and construction specifications.

The limits of replicating or replacing wetlands "in kind" (of the same type) or "off site" (some distance from the wetland being replaced, but usually within the same watershed) and their associated ecosystem services have been extensively cited and debated (e.g., Race and Fonsec 1996; Mitsch and Wilson 1996; Zedler and Callaway 1999, National Research Council 2001, U.S. General Accounting Office 2002; Environmental Law Institute 2004, 2005; Hoeltje and Cole 2007; Hossler et al. 2011), culminating in the release of the so called "Mitigation Rule" by the U.S. Environmental Protection Agency and the U.S. Army Corps of Engineers in 2008 (33 C.F.R. 332.3(c); USEPA 2008). These revisions encourage states to carry out mitigation in a watershed context, prioritizing mitigation projects on a watershed basis to the extent appropriate and practicable. States are expected to establish monitoring programs and measureable performance standards for mitigation wetlands. At present, the science and practice of restoration and mitigation are on the cusp of demonstrating how these sites can be more like their natural counterparts, and thus, deliver the level of structure and function that the profession and public expect.

This chapter provides a summary of research conducted by Riparia (http://www.riparia.psu.edu) that focuses on providing information that can improve practice. Following a brief synopsis of the state-of-the-science, we address specific measures related to both the design of projects and evaluation of their performance. Because understanding terminology precisely is a key to assessing wetlands mitigation, a glossary of *italicized* terms is provided at the end of this chapter for the convenience of readers. Throughout this chapter, we will use "mitigation" when referring generically to *restoration*, *creation*, *construction*, and *enhancement*.

12.2 The Mitigation Process

Mitigation and restoration activities should not be conducted in a vacuum, where the landscape and wetland types are not known. Based on studies conducted by personnel from Riparia and others from the early 1990s, we began to recommend that wetlands mitigation be conducted using a defined process to assess the type and location of wetland restoration or creation, in order to achieve maximum likelihood of effective function (Kentula et al. 1992; Brooks 1993). We were influenced, in part, by findings of Gwin and Kentula (1990) and Kentula et al. (1992b), which demonstrated that wetlands created for purposes of mitigation, were not mimicking natural wetlands found in the landscapes of Oregon where those studies were conducted. As a consequence, the profile of natural wetlands (i.e., wetland abundance by wetland type) found in a given landscape would likely shift to a new profile comprised of dissimilar or unrecognizable types of wetlands (e.g., Bedford 1999), with a resultant shift in functions and values provided by those wetlands.

To further the compatibility of mitigation decision-making and current wetlands science, Brooks et al. (2006) diagrammed an overall planning process where the general objective was to have no net reduction in ecological integrity. Restoration was considered as the last part of a sequence that would likely involve an inventory of existing wetland resources and assessment of target resources, before prioritizing sites for restoration based on their landscape position, conservation status, and restoration potential (Table 12.1).

Table 12.1 Integrated tasks for wetland monitoring matrix (WMM): inventory, assessment, and restoration at three levels of effort

	Inventory	Assessment	Restoration
Level 1: Landscape	Use existing map resources (NWI) of wetlands for priority watersheds	Map land uses in watersheds; compute landscape metrics and initial condition	Produce synoptic watershed maps of restoration potential with multiple sites
Level 2: Rapid	Enhance inventory using landscape-based decision rules classify by NWI and HGM types	Rapid site visit and stressor checklist; determine condition based on human disturbance score	Select sites for restoration; examine levels of threat from surroundings
Level 3: Intensive	Map wetlands intensively for a portion of area; verify inventory; classify by NWI and HGM types	Apply HGM and IBI models to selected sites to assess condition based on reference sites and data	Focus on specific sites for restoration; design projects with reference data sets using performance criteria matrices

Modified from Brooks et al. (2006)

The mitigation process, as recommended, consists of seven major steps (Brooks 1993):

1. Conduct a functional assessment of the wetland to be impacted (assuming there is a need for direct replacement), considering the functional needs for the region of interest.
2. Set site-specific objectives for the project in cooperation with stakeholders, which could include agency personnel, landowners, cosponsors, and/or citizen groups.
3. Select and acquire access to a suitable site.
4. Design conceptual plans based on site conditions and project-specific objectives with input from stakeholders.
5. Prepare construction plans, specifications, and budget.
6. Implement construction and maintenance activities.
7. Prepare as-built condition plans for baseline information, and implement monitoring protocols for evaluation reports.

An underlying tenet of our work has been the critical need to mirror methods used to assess wetland condition (step 1) vs. those used to measure performance (step 7). In plain language, with few exceptions, one must measure the same parameters in the same way when assessing natural wetlands and when evaluating performance of mitigation projects (Kentula et al. 1992; Brooks 1993; Brooks et al. 2005, 2006). Inherent in this tenet, is that practitioners need design and performance criteria, specific to different wetland types that are based on measurements obtained from natural reference wetlands of the intended type, in order to construct and monitor projects, respectively. Unfortunately, design and performance criteria, based on specific types of wetlands occurring in different ecoregions, have not been widely available.

12.3 State-of-the-Science in Wetlands Restoration and Mitigation

Performance "curves", as a concept, were recommended by Kentula et al. (1992a) to document the progression of ecological function(s) within a mitigated wetland over time against findings for reference wetlands within a region. In the mid-1990s, actual performance matrices were compiled for the hydrologic, soil, vegetation, and wildlife components of wetlands in central Pennsylvania in a report that was narrowly distributed, and hence, those data were not widely used. These matrices were structured to provide detailed information (e.g., means, ranges, and species) by HGM subclass to aid in designing mitigation projects for specific wetland types. In addition, when characteristics of natural wetlands were compared to those of mitigation sites, large differences were found that highlighted the poor performance of the mitigation sites.

Further studies by Riparia personnel, reported in Bishel-Machung et al. (1996), Stauffer and Brooks (1997), Cole and Brooks (2000), Cole et al. (2001), Brooks et al. 2002, Campbell et al. (2002), Walls et al. (2005), and Cole et al. (2006), demonstrated the consistent failure of mitigation sites to replicate the structure and function of natural reference sites. For example, mitigation sites had soils with coarser texture, and lower amounts of organic matter, silt, and clay. This was most likely caused by the common practice of excavating to subsoil, without replacing topsoil removed from the sites. Soil bulk densities were higher in mitigation sites, reflecting inappropriate construction practices, such as allowing compaction by heavy, earth-moving machinery. Comparatively, the Munsell chroma of matrices from soils of mitigation sites were brighter than those of reference wetlands, suggesting that insufficient time had transpired for saturation or inundation to occur, which would force the reduction of iron that leads to duller colors. Campbell et al. (2002) found that in created sites up to 18 years since construction, organic matter failed to accrete over time so as to match that of comparable HGM types. They also found that vascular plant richness and total cover were both greater in reference versus created wetlands, and invasive plants were more prevalent in the latter. Basin *morphometry* also varied, with reference wetlands displaying more complex perimeter-to-area relationships than in mitigation sites. This points to the tendency of creating geometric shapes during the wetland construction practices because they are less expensive and simpler to build.

Cole and Brooks (2000) concluded that while created wetlands can meet jurisdictional requirements, their hydrologic behavior is not necessarily the same as that defined by a naturally occurring HGM subclass. Differences in subclass have implication for function. Cole et al. (2002) found that for specific HGM subclasses and settings in Pennsylvania and Oregon, wetlands, which depend on surface water additions (for one HGM subclass), are more likely to have different wetland functions than wetlands that are hydrologically supported by regional water tables (for a different HGM subclass). Such variations in hydrologic regime can lead to differences in the water depth and/or duration of soil saturation, and thus change a wetland's dependence from that of one dominated by anaerobic soil conditions to one

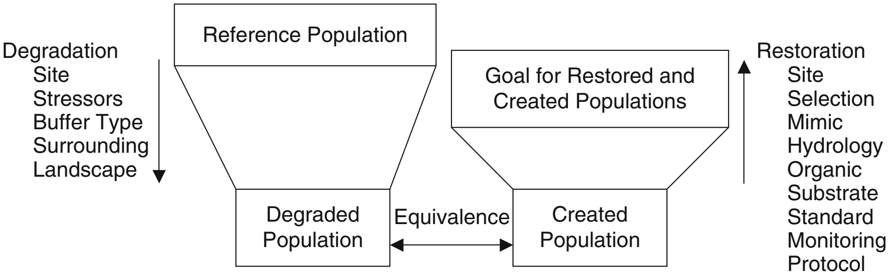

Fig. 12.1 Conceptual model of wetland degradation and restoration showing the equivalence of characteristics for populations of degraded and created populations, and how data from reference wetlands could be used to improve the performance of mitigation projects (modified from Brooks et al. (2005))

reflective of aerobic conditions. This would certainly have an effect on the formation of hydric soil indicators.

This body of work led to the formulation of a conceptual model of how the state-of-the-practice for creation, restoration and mitigation projects was resulting in wetlands that were equivalent in condition and/or function to highly degraded natural wetlands (Brooks et al. 2005). A conceptual model of the issues (Fig. 12.1), along with examples of how mitigation sites failed to match the characteristics of reference wetlands, led to their suggestion to "build a better wetland" by using reference wetlands to generate design and performance criteria that are specific to different types of wetlands.

More recently, Moreno-Mateos et al. (2012) compared over 621 wetland mitigation projects worldwide to 556 reference wetlands, concluding that recovery was slow and incomplete. The net result over time is a net loss of wetland ecosystem services. Larger restoration projects in warmer climates, and those controlled by the dynamics of rivers and tides, approached the level of ecosystem services provided by natural reference sites more rapidly than others. The focus of their meta-analysis was on restored wetlands ($n=401$), and less so on created wetlands ($n=220$, those built from scratch, which are typical of mitigation projects).

In their study, the recovery of ecosystem services in all major categories was always less than those of comparable reference wetlands even after significant periods of time, ranging from <10 to >100 years; wildlife and fisheries, aquatic insects and other invertebrates, and plants do not reach full functional equivalency. Biogeochemical functions also failed to reach the levels found in natural reference wetlands; soil organic matter averaged 62% of reference wetland values and nitrogen accumulation still only averaged 74% of reference wetland values after 50–100 years, and was substantially less over shorter time periods.

Similarly, Gebo and Brooks (2012) compared HGM functional assessments of 222 reference wetlands (spanning an anthropogenic disturbance gradient) from Riparia's Pennsylvania collection to 72 mitigation wetlands sampled in 2007 and 2008 from three categories—Pennsylvania Wetland Replacement Program sites,

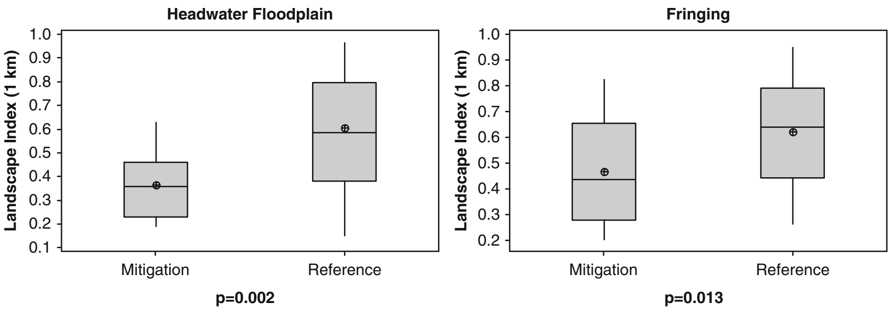

Fig. 12.2 Boxplots (mean, median, standard deviation, range) depicting the difference in Landscape Index score between reference and mitigation wetlands; headwater floodplain and fringing wetlands subclasses (Gebo 2009)

Pennsylvania Department of Transportation mitigation banks, and permit required compensatory mitigation sites. Overall, mitigation sites displayed lower potential to perform characteristic wetland functions than reference wetlands. Depressions show the greatest discrepancy, while fringing sites along lakes showed the least amount of difference from reference scores. The majority of mitigation sites fell within the range of reference function for their HGM subclass, indicating that they are at least equivalent in functional capacity to some naturally occurring wetlands. However, creating wetlands that function at a lower level like those of disturbed natural wetlands should not be considered an optimal mitigation or restoration endpoint (Brinson and Rheinhardt 1996, Zedler 1996).

The data reported here show some examples of functional assessments where mitigation projects scored lower than reference wetlands (Gebo 2009; Gebo and Brooks 2012). Gebo (2009) examined both the landscape setting (Fig. 12.2) and site-level conditions (Fig. 12.3) for mitigation projects and reference wetlands. For both landscape and site comparisons, most functions of mitigation sites scored significantly lower than those of reference wetlands.

For the majority of mitigation wetlands studied by Gebo (2009) and Gebo and Brooks (2012), fewer than 10 years had passed since initial site construction. Hossler et al. (2011) found that created wetlands, even those monitored several decades after construction, were not reaching equivalent patterns of nutrient cycling when compared to natural wetlands, thus raising concerns about long-term success. Other authors have expressed similar performance concerns regarding spatial patterns and temporal lags for mitigation projects designed to meet functional equivalency with natural wetlands (e.g., Gutrich and Hitzhusen 2004; Bendor 2009).

Overall, low functional capacity at mitigation and restoration sites is likely tied to continued problems of attaining hydrologic equivalence. This conclusion is supported by the finding that fringing sites, associated with adjacent deep water aquatic systems, had the most consistently high level of functional potential of all the mitigation sites studied in Pennsylvania (Gebo and Brooks 2012). Trying to mimic the hydrologic regimes of groundwater supported wetlands or mature floodplain forests

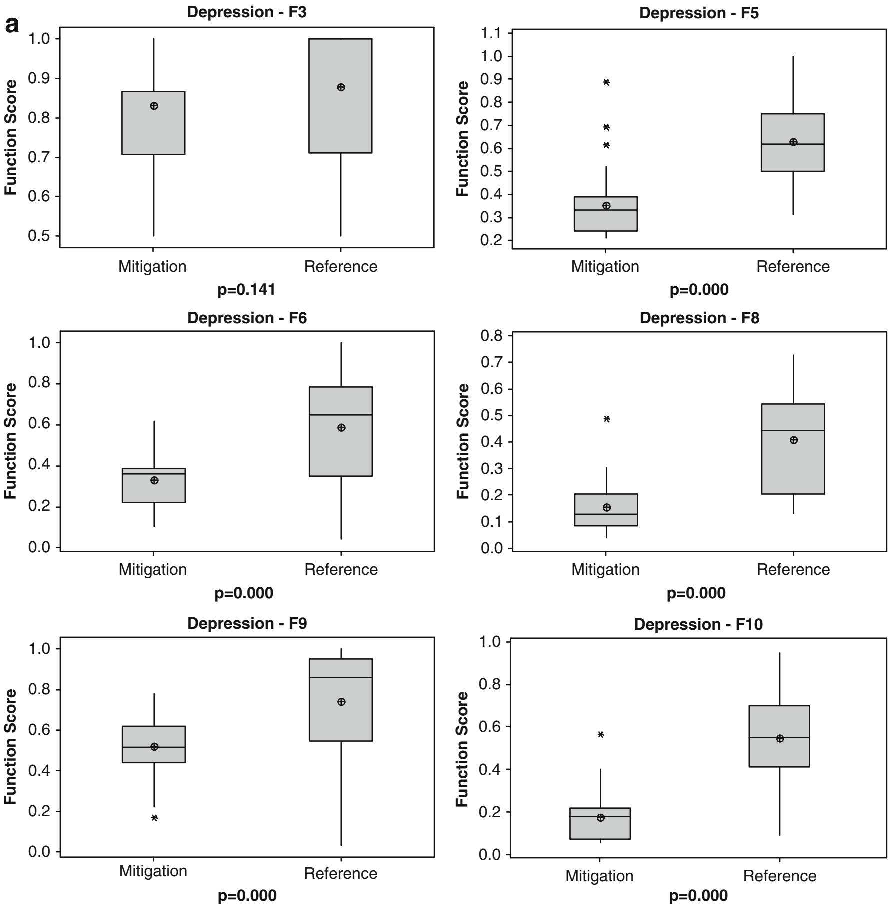

Fig. 12.3 (**a**–**d**) Boxplots (mean, median, standard deviation, range) comparing selected functional capacity index scores for mitigation sites and reference wetlands classified as: (**a**) depression (permanent, seasonal, and temporary), (**b**) fringing (lacustrine), (**c**) headwater floodplain (riverine upper perennial), and (**d**) slope. Functions included here are: F1 (energy dissipation/Short-term surface water detention), F2 (long-term surface water storage), F3 (Maintain characteristic hydrology), F5 (removal of imported inorganic nitrogen), F6 (solute adsorption capacity), F7 (Retention of inorganic particulates), F8 (export of organic carbon), F9 (maintain characteristic native plant community composition), F10 (maintain characteristic detrital biomass), F11 (vertebrate community structure and composition) (see Brooks 2004, Gebo 2009)

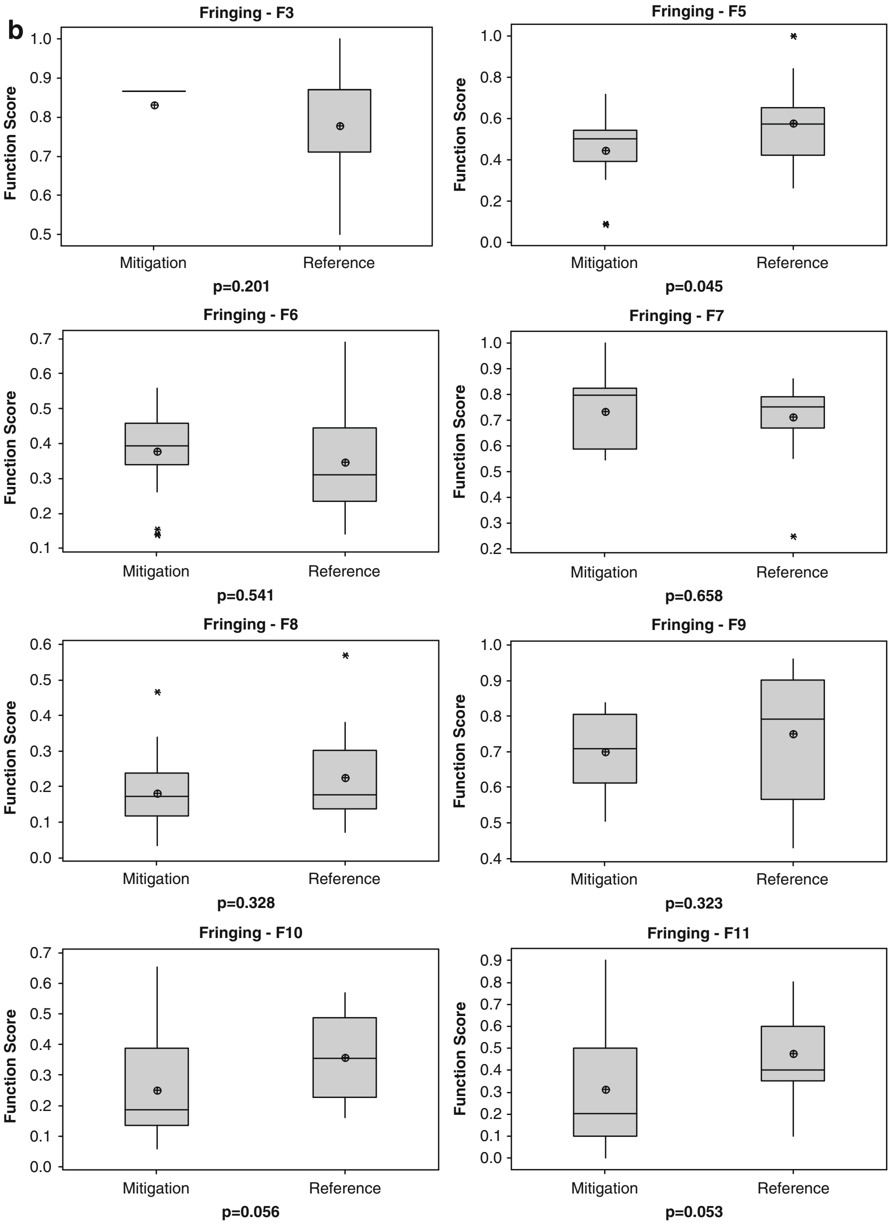

Fig. 12.3 (continued)

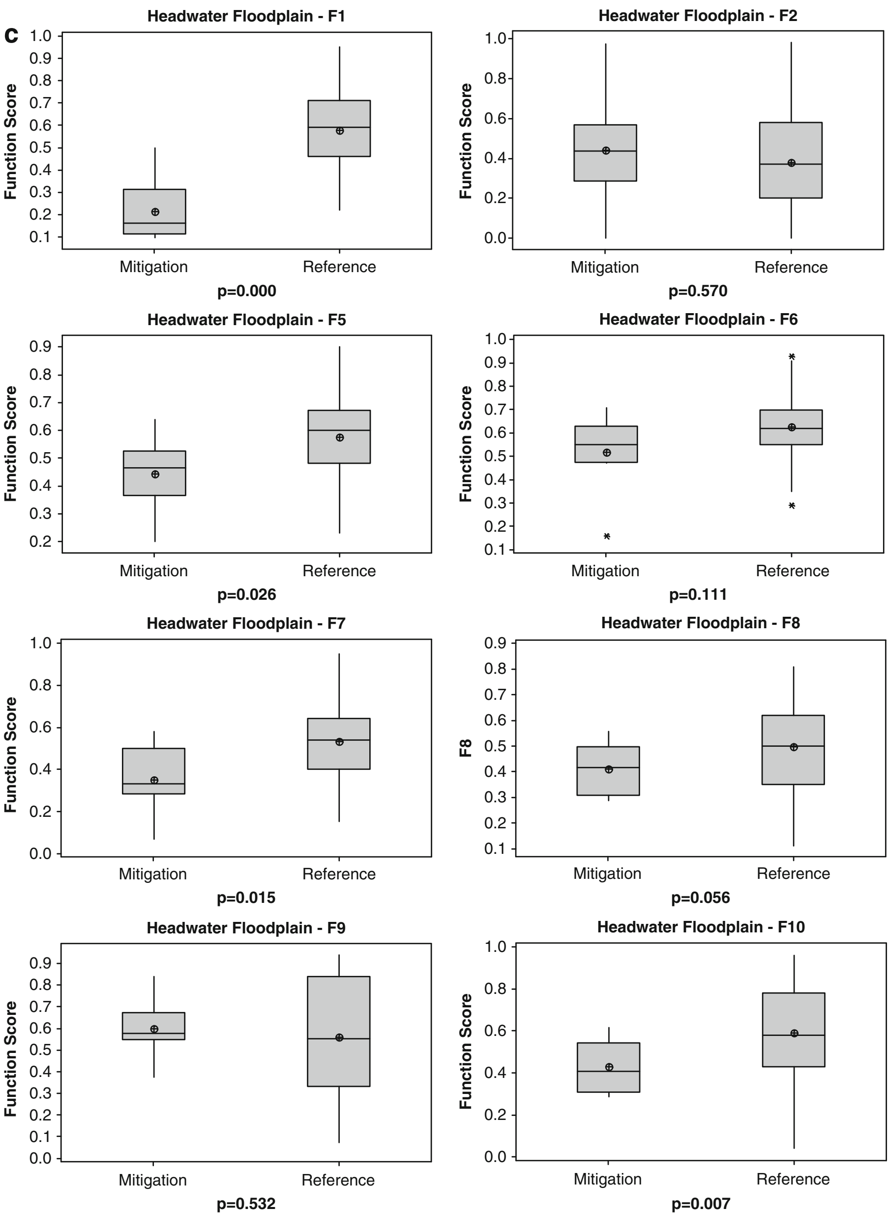

Fig. 12.3 (continued)

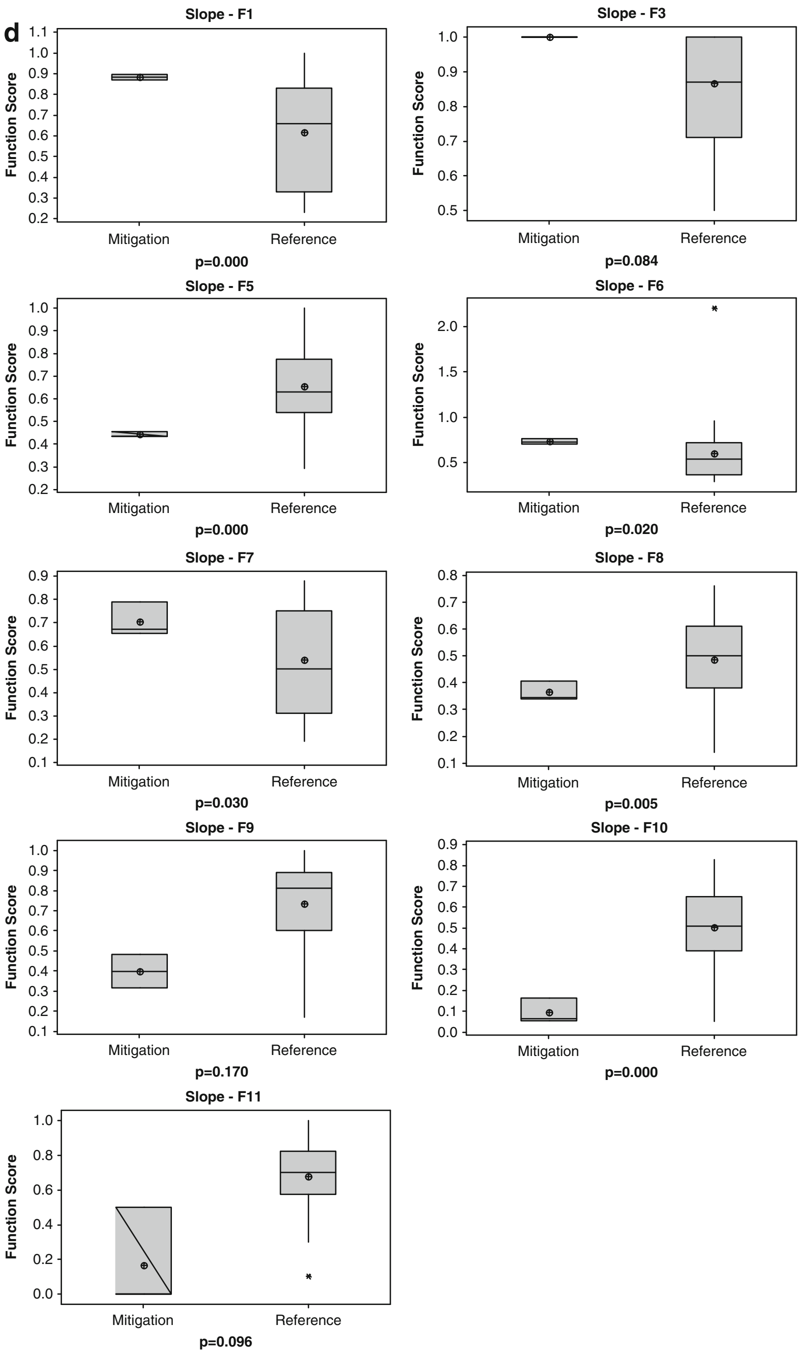

Fig. 12.3 (continued)

is far more difficult, and less likely to succeed, than creating emergent marshes using surface water sources.

12.4 Wildlife Habitat Community Profiles for Natural Reference and Created Wetlands

Wildlife managers and environmental professionals commonly use Habitat Suitability Index (HSI) models to evaluate potential habitat by individual species in single habitat types. During Riparia's studies of natural reference wetlands, we devised a standard means to compare habitat suitability across multiple types of freshwater, inland wetlands of the northeastern US. We developed a wildlife community habitat profile (WCHP) composed of ten species chosen to represent a range of taxa, trophic levels, and habitats (Brooks and Prosser 1995) (Table 12.2). HSI numerical scores (0–1 range) for each of the ten individual species were placed along a vegetation and moisture gradient from open water to forested wetlands, to create a unique wildlife profile. We then compared profiles between reference wetlands ($n=38$) and created wetland projects ($n=12$). Reference herbaceous wetlands were distinguished from reference wooded sites based on significant differences in HSI scores for each species comprising the profile. Species that use emergent wetlands scored equally well on reference herbaceous wetlands and created herbaceous sites, suggesting that wildlife habitat functions can be replaced reasonably well during

Table 12.2 Ten wildlife species used in the wildlife community habitat profile to evaluate reference and mitigation wetlands with Habitat Suitability Index (HSI) models

Common name	Scientific name	Taxonomic group	Trophic level
Open water (with some emergents allowed)			
Bullfrog	*Lithobates catesbeiana*	Amphibian	Carnivore
Muskrat	*Ondatra zibethicus*	Mammal	Herbivore
Emergent (with some open water or shrubs allowed)			
Meadow vole	*Microtus pennsylvanicus*	Mammal	Herbivore
Red-winged blackbird	*Agelaius phoeniceus*	Bird	Granivore
Scrub–shrub (with some emergents or forested wetland allowed)			
American woodcock	*Philohela minor*	Bird	Invertivore
Common yellowthroat	*Geothlypis thrichas*	Bird	Insectivore
Green-backed heron	*Butorides striatus*	Bird	Carnivore
Forested wetland (with some shrubs or emergents allowed)			
Wood duck	*Aix sponsa*	Bird	Herbivore
Wood frog	*Lithobates sylvatica*	Amphibian	Carnivore
Red-backed vole	*Clethrionomys* g. *gapperi*	Mammal	Herbivore

Table 12.3 Comparisons of median HSI scores for ten wildlife species among three wetland types

	Median HSI scores			Pairwise wetland type comparisons[b]		
Wildlife species	RH	RW	CH[a]	RH vs. RW	RH vs. CH	CH vs. RW[c]
Bullfrog	0.66	0.00	0.73	<0.05**	>0.15	<0.05**
Muskrat	0.63	0.27	0.73	>0.15	>0.15	>0.15
Meadow vole	0.61	0.50	0.60	>0.15	>0.15	>0.15
Red-winged blackbird	0.72	0.48	0.79	<0.05**	>0.15	<0.05**
Common yellowthroat	0.45	0.52	0.13	>0.15	<0.05**	<0.05**
American woodcock	0.37	0.44	0.20	>0.15	<0.05**	<0.05**
Green-backed heron	0.68	0.62	0.51	>0.15	0.07*	0.13*
Wood duck	0.32	0.33	0.34	>0.15	>0.15	>0.15
Wood frog	0.42	0.71	0.35	0.06*	>0.15	<0.05**
Southern red-backed vole	0.0.29	0.53	0.00	0.08*	<0.05**	<0.05**

[a,c]Wetland types and comparisons: reference herbaceous (RH), reference wooded (RW), created herbaceous (CH)
[b]Kruskal–Wallis, Bonferroni; p values are reported as >0.15 indicating no significance, *Significant difference for overall <0.15 (0.05 × 3) to >0.06 (0.02 × 3); **Highly significant difference for overall <0.05 (0.017 × 3)

creation of emergent marshes. Species dependent on forest and shrub wetlands scored poorly on created wetlands due to the absence of a wooded component on mitigation projects, which can only appear over time. This pattern is replicated for other functions, where herbaceous, emergent wetlands perform closer to the functional capability of natural wetlands than those dominated by woody vegetation.

The WCHP provides a consistent means to make quantitative and visual (by plotting a histogram of scores for the ten species) comparisons of habitat suitability among wetland types. Many of the variables assessed when applying the HSI models can be incorporated into designs for mitigation projects.

Median habitat scores for the bullfrog, wood frog, muskrat, meadow vole, red-winged blackbird, or the wood duck were not significantly different between reference herbaceous sites and created herbaceous sites. The common yellowthroat, American woodcock, green-backed heron, and southern red-backed vole, all showed significantly higher habitat values on reference herbaceous sites than on created herbaceous sites (Table 12.3).

Reference herbaceous sites scored significantly higher than the reference wooded sites for the bullfrog and red-winged blackbird, and reference wooded sites scored significantly higher than reference herbaceous sites for the wood frog and southern red-backed vole. No differences were observed in scores for species located in the middle portion of the vegetative profile; muskrat, meadow vole, common yellowthroat, American woodcock, green-backed heron, and the wood duck (Table 12.3).

Reference wooded sites had significantly higher scores than created herbaceous sites for species that require woody cover; common yellowthroat, American woodcock, green-backed heron, wood frog, and southern red-backed vole. Created herbaceous sites had significantly higher scores for the bullfrog and red-winged blackbird. No differences were found for the muskrat, meadow vole, and wood duck (Table 12.3).

Overall, reference herbaceous and created herbaceous sites provided equivalent wildlife habitat functions for six of the ten species. Habitat potential was poor for the four species which prefer some wooded cover; common yellowthroat, American woodcock, green-backed heron, and southern red-backed vole. If reference wooded wetlands are destroyed and replaced by mitigated wetlands dominated by herbaceous cover, a resulting shift in the wildlife community is likely to occur. Thus, while wetland practitioners are capable of producing equivalent habitat potential for species that require herbaceous, emergent marshes, there is little evidence of functional replacement of habitats for species that require the forest and shrub components of wetlands.

Given the overwhelming empirical evidence from these and many other studies by others that mitigation projects usually do not mimic the structure nor perform the functions of natural wetlands, and given the shift in policy accentuated in the release of the federal "Mitigation Rule," we focus the rest of this chapter on how to achieve the performance we desire.

12.5 Design and Performance Criteria

Following our basic premise of assessing mitigation projects and reference sites using comparable methods, we present a list of variables derived from assessments of natural wetlands that can then be used for evaluating the performance of wetland mitigation projects. The site data related to these measures are voluminous, and thus, are best served from a website (www.riparia.psu.edu) where characteristic measures can be selected by HGM types and for designated ecoregions. These initial data are primarily from Pennsylvania, but a summary of data from reference sites for the Mid-Atlantic Region is planned for distribution through the Riparia website. Because of their particular geographic origin, these data should be used with caution for other areas. Many of these variables, however, will have some relevance to wetlands of a particular type in many other physiographic regions. For readers interested in additional details about sampling and assessment methodologies, and how variables are scored and combined for HGM functional assessment models, refer to the appropriate sections of Brooks 2004 (Table 12.4) (relevant section available in pdf form at http://www.riparia.psu.edu)

Variables are divided into several categories:

1. Variables collected remotely or from GIS databases to assess the landscape around a site for assessment, design, or performance purposes
2. Variables collected at ground level primarily for design purposes
3. Variables collected at ground level for assessment or performance purposes

Once the purpose and objectives for a mitigation project are defined, site selection becomes a most critical next step. So, the first variables presented are used to assess the landscape surrounding a site. Mitigation projects located in an inappropriate place are likely to fail.

Table 12.4 Variables used to compare among natural wetland types and between reference wetlands and mitigation sites (see Brooks (2004) for additional details)

Variable	Design	Performance	Landscape
AQCON	X		
BIOMASS	X	X	
HERB% COV	X	X	
SHRB% COV	X	X	
TREE% COV	X	X	
CWD-BA	X	X	
CWD-SIZE	X	X	
EXOTIC		X	
FLOODP	[X]	[X]	
100FLOODPL			X
FWD	X	X	
GRAD	X	X	
HYDROCHA	[X]	[X]	
HYDROSTR			X
MACRO	X	X	
MICRO	X	X	
MPS			X
ORGMA	X	X	
REDOX		X	
REGEN		X	
ROUGH	X	X	
RDDEN			X
SDI			X
SNAGS	X	X	
SPPCOMP	X	X	
STR INDEX			X
NEAR DIST			X
TEXTURE	X	X	
UNDEVEL			X
UNOBSTRUC			X
URBAN			X
LDI			X
—			
WILDLIFE	X	X	X

HSI variables for ten species (Brooks and Prosser 1995; Brooks 2004)

Presumably, one aspect of effective project planning must include a decision on what type of wetland is to be built. Choosing a subclass based on both the physically oriented HGM system (Brinson 1993) and the vegetation-oriented Cowardin et al. (1979) system has worked well for our studies. We have developed a regional wetlands classification system for the Mid-Atlantic that pays homage to both systems, although HGM is emphasized (Brooks et al. 2011).

To emphasize the importance of location and landscape position, there are at least nine primary variables used to compute six metrics pertinent to selecting the location of a site in a landscape (see Brooks 2004 at http://www.riparia.psu.edu for definitions and sampling protocols):

1. Aquatic connectivity (VAQCON) is a composite variable comprised of three subvariables; occurrence of the site in a 100-year floodplain (V100FLOOD), stream density index (VSTR INDEX), and distance to the nearest National Wetlands Inventory mapped wetland (VNEAR DIST)
2. Gradient (VGRAD)
3. Number of hydrologic stressors (VHYDROSTRESS)
4. Average forest patch size within a 1-km radius circle (VMPS)
5. Road density with a 1 km radius circle around the site (VRDDEN)
6. Shannon diversity index (VSDI) for landscape categories within a 1-km radius circle around the site or Land Development Index (LDI) for a site
7. Undeveloped (VUNDEVEL) portions of landscape (a composite variable of VRDDEN and VURB)
8. Unobstructed portions of riverine floodplains (riverine types only; a composite variable of VRDDEN, VURB, and VHYDROSTRESS)
9. Percentage of urban land within a 1-km radius circle around the site (VURB)

Once a subclass is chosen, and a suitable location is secured, then the set of pertinent ground-based variables can be explored and translated into project-specific design criteria intended to produce a wetland that shares characteristics with its natural counterparts. Most importantly, the chosen location must have a hydrologic regime that provides the sources of water needed by that type of wetland, with sufficient quantities to meet frequency and duration requirements. For example, if on-site soils do not meet texture, nutrient, and/or organic matter content parameters, then it may be necessary to use soil amendments in appropriate quantities, usually computed volumetrically.

We recommend 17 variables collected at ground level for use in assessment, design, and performance purposes (see Brooks 2004 at www.riparia.psu.edu). Those variables with an "*" are only pertinent for measuring assessment and performance, as they are not particularly useful for design purposes. There are a few variables that are deemed to be important, but for which we do not have established field measurements to capture them, denoted by brackets [ ].

1. Biomass (VBIOMASS)—a metric composed of abundance and composition measures for herbaceous, shrub and tree strata within nested plots of different sizes.
2. Coarse woody debris—basal area (VCWD-BA)—measure of basal area for three diameter classes
3. Coarse woody debris—size (VCWD-SIZE)—abundance of three diameter classes
4. Exotic plants (VEXOTIC)—% of species list that are non-native
5. [Floodplain characteristics] (VFLOODP)—reserved until suitable measurements are developed
6. Fine woody debris (VFWD)—visual estimate of litter layer

7. [Hydrologic characteristics] (VHYDROCHAR)—reserved as a measurement; available hydrographs for designated HGM subclass should be examined
8. Hydrologic stressors (VHYDROSTRESS)—captured from the stressor checklist; obviously should be minimized for mitigation project planning and site location
9. Macrotopographic depressions (VMACRO)—number of topographic depressions in wetland (usually a floodplain) where standing water is more likely to occur
10. Microtopographic complexity (VMICRO)—used in the computation of VROUGH, an adaptation of Manning's roughness coefficient
11. Soil organic matter (VORGMA)—% soil organic matter usually in the top 5 cm of soil profile (amount at 20 cm depth may also be relevant)
12. Redoximorphic features (VREDOX)—Munsell chroma of soil matrix and mottles (if any) at 20 cm depth
13. Regeneration (VREGEN)—presence of dominant tree species in multiple strata
14. Roughness (VROUGH)—modified Manning's roughness coefficient
15. Snags (VSNAG)—density and diameter of erect dead woody material in three diameter classes
16. Species composition of flora (VSPPCOMP)—uses Floristic Quality Assessment Index (VFQAI) scores to reflect species composition of all vascular plants
17. Soil texture (VTEX)—measurement or observation of soil texture as a surrogate for mineral particle size and pore space

Additional characteristics that may be useful are detailed measures of the hydrologic regime, usually taken from automatic recording wells (see Chap. 4), and potential habitat characteristics, often derived from Habitat Suitability Models (HSI, see Brooks and Prosser 1995).

By designing mitigation sites with characteristics derived from reference wetlands of relevant HGM subclasses, practitioners are more likely to construct a project that will at least be on a performance trajectory to replace the ecosystem services of natural systems. As mentioned above, a searchable database based on reference wetlands is available at http://www.riparia.psu.edu, and we encourage users to design and construct restoration and mitigation projects based on these data. In time, we believe this will lead all of us toward "building better wetlands."

12.6 Glossary

Compensatory mitigation Creation, restoration, enhancement or preservation of a wetland designed to offset permitted losses of wetland functions in response to special conditions of a permit (National Research Council 2001)

Construction Activities resulting in the building of a wetland for restoration or mitigation purposes

Constructed wetlands Created for the primary purpose of contaminant or pollution removal from wastewater or runoff (Hammer 1997)

Creation Conversion of a persistent upland or shallow water area into a wetland (National Research Council 2001)

Design Plan for a mitigation project, usually based on measures of a wetland's intended structure and function

(Wetland) enhancement Refers to human activity that increase one or more functions of an existing wetland (National Research Council 2001)

Mitigation Similar to compensatory mitigation, but can include substitution of creation, restoration, enhancement or preservation of other aquatic or upland habitat types

Morphometry Topographic measures of a wetland's size, shape, slope and depth

Performance Measurable outcome of a mitigation project, usually based on assessment of a wetland's structure and function

(Wetland) preservation Refers to the protection of an existing and well-functioning wetland from prospective future threats (National Research Council 2001)

(Wetland) restoration To return a wetland from a disturbed or altered condition by human activity to a previously existing condition (National Research Council 1992)

Voluntary restoration Same as restoration, but landowner makes a conscious choice unrelated to permitting requirements

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Chapter 13
Policy and Regulatory Programs Affecting Wetlands and Waters of the Mid-Atlantic Region

James M. McElfish Jr. and Robert P. Brooks

Abstract Federal and state laws and policies determine which wetlands and waters are protected and which are not. More than a century of policy evolution has reflected growing understanding of the importance of wetland systems, while responding to economic and social pressures of a rising population with development expectations. Federal laws, chiefly the Clean Water Act, provide the most substantial regulatory framework governing what activities may take place in wetlands and under what conditions. The U.S. Army Corps of Engineers operates the federal permitting program, which allows filling of waters and wetlands under individual, nationwide, or general permits, subject to requirements for avoidance, minimization, and compensation for impacts. Supreme Court cases in the first decade of the twenty-first century have made the application of the Clean Water Act to wetlands more complex, requiring science to try to answer legal questions. In the Mid-Atlantic Region, state laws also regulate activities in many wetlands and waters, with most states operating permitting regimes in addition to the federal system. Finally, other federal programs and international agreements provide additional opportunities for wetland conservation.

13.1 Why Laws and Policies Matter

Laws and policies led first to activities promoting wetland loss and then later to wetland conservation, preservation, and restoration. Changes in American society, and related changes in our laws and policies, reflect a growing recognition of the

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important contributions of wetlands to water quality, wildlife habitat, hydrological integrity, recreation, open space, and ecosystem services.

Our increasing understanding of wetland functions and values since the middle of the twentieth century has led to changes in law to accommodate advances in scientific understanding. These wetlands laws and policies, in turn, create a demand for wetlands science. Regulation writers, government officials, consultants, developers, agricultural agency personnel, land use planners, wildlife managers, and many others operate at the intersection of wetlands science and law. Wetlands scientists can identify whether a given area is regulated or not, what functions it performs, and how a wetland can best be conserved, restored, mitigated, or managed.

Changes in law, new decisions by the federal and state courts, and growing experience in wetlands conservation, demand that wetlands scientists pursue an ongoing engagement with the world of law. Trying to conserve and restore wetlands with unchanging laws would be like a medical doctor trying to heal patients using only medical textbooks and instruments from the past: it would be better than having no text or technology at all, but it would not reflect the best practices that science can bring to bear. As new challenges arise (such as climate change's effect on freshwater systems), and new knowledge arrives (such as the contribution of small and isolated wetlands to nutrient cycling or animal life history), so too laws and policies will need to be able to respond. In the Mid-Atlantic Region (MAR), wetlands laws and policies must be able to adapt as our understanding changes.

13.2 Historical Review of Wetland Exploitation, Conservation, and Protection

When considering the status of wetlands protection in the twenty-first century, it is instructive to examine the evolving path that brought us to the current conservation and regulatory regime. The following summary generalizes over the past century with respect to wetlands drainage, conservation, protection, and awareness. The names for the early phases of conservation history were, in part, devised by James Trefethen in his book, An American Crusade for Wildlife (Trefethen 1975), but are pertinent to the wetlands story. As can be seen, concerns for wetlands have often paralleled other phases of environmental awareness and protection. Relevant laws, regulations, and actions are listed in Table 13.1.

Before 1890—Period of Exploitation

- Massive drainage of wetlands due to Swamp Land Acts (1849, 1850, 1860)

1890–1910—Period of Conservation

- Large parcels federally protected for natural and cultural resource values
- Pelican Island, Florida established as first National Wildlife Refuge (wading birds) under Executive Order by President Theodore Roosevelt
- Dam building and public works seen as part of conservation

Table 13.1 Relevant federal laws, orders, regulations, guidance, and programs influencing protection of wetlands in the Mid-Atlantic Region, listed chronologically (see Mitsch and Gosselink (2007), Table 14.1 and text for a more extensive list)

Item or action	Year	Responsible organization
Rivers and Harbors Act	1899	U.S. Army Corps of Engineers
Migratory Bird Treaty Act(s)	1913, 1916, 1918	U.S. Fish and Wildlife Service
Fish and Wildlife Coordination Act	1934	Federal water resource agencies
National Environmental Policy Act	1969	All Federal agencies and Council on Environmental Quality
Convention on Wetlands of International Importance (Ramsar Convention)	1971	International Contracting Parties
Federal Water Pollution Control Act amended as Clean Water Act	1972, 1977, 1987	U.S. Army Corps and USEPA
Coastal Zone Management Act	1972, 1990	U.S. Dept. Commerce—NOAA
Endangered Species Act	1973	U.S. Fish and Wildlife Service/NOAA
Executive Order 11990—Protection of Wetlands	1977	All federal agencies
Executive Order 11988—Protection of Floodplains	1977	All federal agencies
Food Security Act Swampbuster provisions	1985	U.S. Dept. Agric., Natural Resource Conservation Service
North American Waterfowl Management Plan	1986	U.S. Fish and Wildlife Service and Canadian Wildlife Service
Wetlands Delineation Manuals	1987, 1989, 1991	All federal agencies
No net loss policy	1988	All federal agencies
North American Wetlands Conservation Act	1989	All federal agencies
Wetlands Reserve Program	1991	U.S. Dept. Agric., Natural Resources Conservation Service
Compensatory Mitigation for Losses of Aquatic Resources (Mitigation Rule)	2008	U.S. Army Corps and USEPA

1910–1940—Period of Maturing Conservation (Science begins to inform conservation)

- Enactment of migratory bird protection laws with an emphasis on waterfowl
- Enactment of "Duck Stamp Act" (1934) providing a source of funds for wetland acquisition for National Wildlife Refuges
- Establishment of soil conservation service, techniques professionalizing conservation, and education on private lands

1940–1960—Period of Industrialization

- Economic development
- Public subsidy of wetland loss
- Public drainage projects

1960–1990—Period of Environmental Awareness (& Wetlands Inventory)

- Documented substantial historic losses in wetland area and function
- Passage of Clean Water Act (CWA) in 1972 with provisions for wetland protection
- Legal definition of wetlands formulated; delineation manual developed
- Recognized functions and values incorporated into federal and state regulations
- Recognition of wetlands in federal Farm Bill legislation
- Developed Water Quality Standards, primarily for streams and rivers
- Wetland classification system (NWI) implemented and mapping initiated
- Early standardization of wetlands assessment with the Federal Highway Method (Adamus 1983) and subsequent state efforts
- Founding of Society of Wetland Scientists (1980) and Association of State Wetland Managers (1983)
- Regulatory and education efforts advanced to increase protection of wetlands

1990–2005—Period of Environmental Maturation (& Stewardship)

- Delineation methodology standardized by returning to a modified 1987 manual
- Development of CWA 404 procedures and standards, coordination between EPA and Corps of Engineers
- Watershed reporting (CWA Sections 305(b) and 303(d) lists) of stream condition
- Wetlands recognized and considered as a heterogeneous resource
- Attention to achieving "no net loss" of regulated wetlands under federal policy
- Development and spread of wetland mitigation banks to provide compensatory mitigation for permitted wetland losses
- Developed hydrogeomorphic (HGM) approach (classification, reference, and functional assessment), and other assessment approaches.
- Retrenchment in CWA coverage of wetlands under Supreme Court decisions (2001, 2006)

2005–2010—Period of Wetlands Monitoring and Assessment

- Wetland definitions and delineation methodology remained essentially unchanged, but federal jurisdictional determinations far more complex
- Issuance of Compensatory Mitigation for Losses of Aquatic Resources (Mitigation Rule) to address wetlands and other aquatic ecosystems by watershed
- Implementation of assessment approaches and indication to assess condition over large geographic areas (regions, states, watersheds)
- Concept of ecosystem services begins to replace terms of functions and values
- Planning for the first National Wetlands Condition Assessment (2011 launch)
- Development and implementation of Water Quality Standards for wetlands by states progresses
- Watershed reporting of wetland condition by states to USEPA progresses (due by 2014)
- Potential effects of wetlands on climate change, and impacts on wetlands by climate, becomes an issue of global concern

13.3 Regulatory Programs

13.3.1 Introduction

The goals of most regulatory programs affecting freshwater wetlands include public health, hydrological and ecological integrity, habitat conservation, water supply, and others. The federal CWA declares a goal to "restore and maintain the chemical, physical, and biological integrity of the nation's waters," for example (33 U.S.C. §1251(a)). (U.S.C. stands for "United States Code," the official compilation of federal laws enacted by Congress. The CWA is found in volume 33). Various state laws are aimed at protecting the "waters of the state," or specifically at preventing pollution or degradation. Typical forms of regulation include permit requirements for certain activities (such as dredging or filling of wetlands), but just as typically contain exceptions and exclusions (frequently for practices associated with agriculture, e.g.). The types of wetlands subject to regulation vary as well, and so it is important to examine legal definitions closely.

13.3.2 Federal Regulation

Federal regulation has dominated the wetlands regulatory landscape since the early 1970s. The key provisions are discussed below, but the field is complex, involving numerous federal agencies, several major laws, and hundreds of pages of detailed regulations, and many hundreds more of interpretive "guidance" documents and standard operating procedures. Federal court cases also affect the interpretation and application of the laws that regulate activities in wetlands (Strand and Rothschild 2009). Both the U.S. Environmental Protection Agency and U.S. Army Corps of Engineers maintain useful websites addressing the relevant regulatory programs.

13.3.2.1 Clean Water Act §404

Federal laws provide a substantial part of the regulatory protections for wetlands, which may also receive some protection from state and local governments. The Federal Water Pollution Control Act, more generally known as the Clean Water Act (or CWA), establishes the primary federal framework for regulation of water quality. The CWA is important in the freshwater wetlands context because it requires those seeking to fill wetlands to first obtain a permit from the Army Corps of Engineers under regulations jointly established by the Corps and the Environmental Protection Agency (33 U.S.C. §1344).

The CWA applies to "navigable waters," defined as "waters of the United States, including the territorial seas." 33 U.S.C. §1362(7). Such waters have for decades been interpreted to include many, if not most, wetlands. Indeed, the

Supreme Court has ruled that waters need not be "navigable in fact" in order to come within the Act's jurisdiction, and that waters and wetlands adjacent to navigable waters are covered by the Act. *United States v. Riverside Bayview Homes, Inc.*, 474 U.S. 121 (1985).

However, following two Supreme Court decisions in the early part of the twenty-first century, the CWA's ability to regulate activities affecting isolated wetlands, ephemeral and intermittent streams, and some headwaters streams, and their associated wetlands is now in considerable doubt. In 2001, the Supreme Court decided *Solid Waste Agency of Northern Cook County v. U.S. Army Corps of Engineers, 531 U.S. 159 (2001)*, commonly known as the *SWANCC* case. In a five-to-four ruling, the Court concluded that Congress had not intended the federal CWA to reach "isolated ponds, some only seasonal" that were located wholly within one state, where the sole basis for federal jurisdiction was their use as a habitat by migratory birds. After *SWANCC*, waters and wetlands deemed to be isolated are, for the most part no longer protected by the CWA.

Five years later, the Supreme Court again addressed the jurisdictional scope of the CWA, in *Rapanos v. United States,* 547 U.S. 715 (2006). This awkwardly divided decision lacked a majority opinion. *Rapanos* established two different rules for determining whether wetlands (and, perhaps, other waters) are jurisdictional for purposes of the federal Act. Justice Scalia's opinion (on behalf of four justices) would find CWA coverage for a wetland only where the wetland has a *continuous surface connection* with a *relatively permanent* body of water that is connected to traditional navigable waters. Justice Kennedy's concurring opinion in *Rapanos* would find CWA coverage for wetlands where there is a *significant nexus* between the wetlands and downstream waters—i.e., where the wetlands, "either alone or in combination with similarly situated lands in the region, significantly affect the chemical, physical, and biological integrity of other covered waters more readily understood as 'navigable.'" Thus, the Corps of Engineers and EPA are required to engage in complex jurisdictional determinations, in addition to determining whether a specific water meets the "wetland" definition in the regulations. Numerous lower court decisions in the years after *Rapanos* have indicated that wetlands and waters are subject to CWA jurisdiction if they meet *either* the adjacent surface connection or the significant nexus test (Environmental Law Institute 2012). Based on their interpretation of the Court's multiple opinions (there were actually five separate opinions in *Rapanos*, none commanding a majority), the Corps and EPA issued a joint guidance document in 2007, finalized in 2008, to guide their field offices in applying the CWA (USEPA and US Army Corps 2008b). In 2011, the Corps and EPA proposed an updated guidance, further interpreting the jurisdictional tests (USEPA and US Army Corps 2011). The 2008 guidance will be used until an updated version is adopted.

In general the agencies will assert jurisdiction over wetlands and waters as follows:

The agencies will assert jurisdiction over the following waters:

- Traditional navigable waters
- Wetlands adjacent to traditional navigable waters

- Non-navigable tributaries of traditional navigable waters that are relatively permanent where the tributaries typically flow year-round or have continuous flow at least seasonally (e.g., typically 3 months)
- Wetlands that directly abut such tributaries

The agencies will decide jurisdiction over the following waters based on a fact-specific analysis to determine whether they have a significant nexus with traditional navigable water:

- Non-navigable tributaries that are not relatively permanent
- Wetlands adjacent to non-navigable tributaries that are not relatively permanent
- Wetlands adjacent to but that do not directly abut a relatively permanent non-navigable tributary

The agencies generally will not assert jurisdiction over the following features:

- Swales or erosional features (e.g., gullies, small washes characterized by low volume, infrequent, or short duration flow)
- Ditches (including roadside ditches) excavated wholly in and draining only uplands and that do not carry a relatively permanent flow of water

The agencies will apply the significant nexus standard as follows:

- A significant nexus analysis will assess the flow characteristics and functions of the tributary itself and the functions performed by all wetlands adjacent to the tributary to determine if they significantly affect the chemical, physical, and biological integrity of downstream traditional navigable waters.
- Significant nexus includes consideration of hydrologic and ecologic factors (USEPA and US Army Corps 2008b)

Section 404 of the CWA establishes a permit program, administered by the U.S. Army Corps of Engineers under guidelines developed by the EPA, to regulate discharges of dredged and fill material into the waters of the United States (including wetlands that meet the definitions) 33 U.S.C. §1344. However, the CWA exempts from 404 permitting "the discharge of dredged or fill material from normal farming, silviculture, and ranching activities," as well as maintenance of certain structures, maintenance of drainage ditches, construction or maintenance of farm roads or forest roads or temporary roads for moving mining equipment constructed in accordance with specified best management practices.

Federal regulations provide detailed requirements for avoiding unnecessary fills where alternatives exist, minimization of remaining impacts, and compensatory mitigation of any unavoidable impacts. Avoidance, minimization, and compensatory mitigation are known as the mitigation "sequence."

Corps of Engineers Section 404 permits are issued by the relevant Corps district (there are 38 across the country), and are subject to a technical review process and opportunity for public review. Section 404 permits can be applied for and issued as *individual* permits; these undergo individual review by the district, including a jurisdictional determination if needed, and application of the federal standards for review and mitigation. There is also a process under the CWA that allows certain low-impact routine activities to be addressed by a general permit that does not require individual review. The Corps has adopted (and every 5 years must review and readopt) "nationwide permits" that establish standard conditions for activities that occur frequently and for which the Corps has determined that activities are "similar in nature, will cause only minimal adverse environmental effects when performed separately and will have only minimal cumulative adverse effect on the environment." 33 U.S.C. §1344(e). Corps districts may also adopt *general* permits to address certain kinds of common activities, including statewide programmatic general permits to improve coordination with state permitting programs, for example. The Corps estimates that it processes 4,500–5,000 individual permits each year, while about 40,000 regulated actions are covered by nationwide permits and another 45,000 by general permits including statewide programmatic general permits.

Section 404(c) authorizes EPA to prohibit, restrict, or deny the discharge of dredged or fill material at a specific site whenever it determines, after notice and opportunity for public hearing, that such use of the site would have an "unacceptable adverse effect" on municipal water supplies, shellfish beds, and fishery areas (including spawning and breeding areas), wildlife, or recreational areas. This "veto" authority, used only on rare occasions, provides a regulatory backstop to Corps actions that EPA believes will not be consistent with environmental conservation of the waters of the United States.

States are authorized to "assume" the 404 permit program and operate in lieu of the Corps upon meeting appropriate conditions, but only New Jersey and Michigan have done so. States that have not assumed the 404 program nevertheless often coordinate their 401 review (see below) and often coordinate their independent administration of their own state-enacted wetlands protection laws with the Corps of Engineers permit program.

Because the Section 404 permit is a federal action, permit actions by the Corps are subject to environmental impact review under the National Environmental Policy Act (NEPA), discussed below. Being federal, this permit may also trigger consultation under the Endangered Species Act (ESA), also discussed below.

13.3.2.2 Clean Water Act §401

Section 401 of the CWA requires states or interstate agencies with jurisdiction to review applications for federal permits and licenses and to certify that the federally authorized actions will not violate adopted state water quality standards. 33 U.S.C. §1341. No federal license or permit may be granted until the state certification has been obtained, or waived by state inaction.

This "401 certification" process gives states an opportunity to review proposed permitting actions subject to Corps of Engineers 404 permits. Where relevant water quality standards apply, states can use their certification authority to deny or impose conditions upon approval of the federal permit.

In addition to review of individual permits, states also apply Section 401 review and certification to the adoption of both nationwide permits and general permits, and may deny certification to any that violate state water quality standards. As a result, certain "nationwide permits" adopted by the Corps do not apply in specific states where certification has been denied, or may apply only with conditions imposed by the state.

About half the states use their 401 certification programs as their sole or primary means of regulating activities in freshwater wetlands. However, because Section 401 applies only to activities where there is a federal permit or license, this authority cannot be used if the water or wetland in question is not subject to federal CWA jurisdiction. In the MAR, only Delaware, West Virginia, and the District of Columbia depend primarily on their 401 programs to address freshwater wetlands; the other states apply 401 but also have their own freshwater permitting programs under state laws (discussed below).

13.3.2.3 Rivers and Harbors Act §10

In addition to the Clean Water Act 404 program, the Army Corps of Engineers also has significant authority over maintaining water transportation and navigation of the nation's waterways. In "any of the waters of the United States," an obstruction to navigation, such as a pier, jetty, or other structure, or the modification of the course, condition or capacity of a waterway or navigation terminus is prohibited unless it is authorized by permit from the Corps. 33 U.S.C. §403 (originally Section 10 of the Rivers and Harbors Act of 1899). This Section 10 permit is also subject to Section 401 certification by states. But, unlike the Section 404 program, the Section 10 program cannot be "assumed" by states, and is administered solely by the Corps.

13.3.2.4 Executive Orders

Several Executive Orders, issued by the President to direct the discretionary actions of federal agencies, have been significant in addressing wetlands. Executive Orders are not enforceable by outside parties, but serve to shape the actions of executive agencies. Executive Order 11990, "Protection of Wetlands," issued in 1978 and amended in 1988, makes wetland protection a responsibility of all federal agencies. It directs that agencies "minimize the destruction, loss or degradation of wetlands, and … preserve and enhance the natural and beneficial values of wetlands." It also directs federal agencies, to the extent allowed by law, to avoid undertaking or providing assistance for new construction in wetlands unless there is no practicable alternative, and all practicable measures are taken to minimize harm.

Executive Order 11988, "Floodplain Management" requires federal agencies to evaluate the effects of their actions on and in floodplains, and to consider alternatives and minimize impacts. "Each agency shall provide leadership and shall take action to reduce the risk of flood loss, to minimize the impact of floods on human safety, health, and welfare, and to restore and preserve the natural and beneficial values served by flood plains in carrying out its responsibilities" for acquiring, managing, and disposing of federal lands and facilities; providing federally undertaken, financed, or assisted construction and improvements; and conducting federal activities and programs affecting land use, including but not limited to water and related land resources planning, regulation, and licensing activities.

Both of these orders can be used to encourage federal agencies to take actions (or avoid actions) that may not necessarily be compelled by regulations or permit provisions, but that result in better outcomes for wetland and floodplain areas.

13.3.2.5 Endangered Species Act

The ESA protects and requires the recovery of species listed as endangered or threatened. 16 U.S.C. §1533. Many species listed as threatened or endangered area wetland-dependent. Pursuant to Section 9 of the Act, it is illegal for any person to "take" any endangered species. 16 U.S.C. §1538. "Take" is defined as to "harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct." 16 U.S.C. §1532 (Endangered plants are separately listed; however, listed plants enjoy lesser protections under Section 9).

Section 7 of the Act prohibits any federal agency from authorizing, funding, or carrying out any action that may jeopardize the existence of a listed species or result in the "destruction or adverse modification" of their critical habitat. It requires agencies to "consult" with the U.S. Fish and Wildlife Service (or National Oceanic and Atmospheric Administration for some species) to determine that the action will not jeopardize such species or habitat. The Section 7 consultation requirement frequently comes into play in connection with evaluation of a CWA Section 404 permit in an area with known occurrences of listed species. Because of the federal permit, the consultation requirement is triggered.

Since 1982, FWS and NOAA have had the authority under Section 10 to allow the taking of a listed species by nonfederal entities for activities that may cause incidental harm to a listed species, if the permittee agrees to develop a habitat conservation plan (HCP). 16 U.S.C. §1539(a). One of the conditions of the permit, known as a §10 "incidental take" permit, is that the applicant will, "to the maximum extent practicable, minimize and mitigate the impacts of such taking." HCPs must identify the impact on the listed species, the steps the applicant will take to monitor, minimize, and mitigate those impacts, and the funding available to implement the plan. HCPs were first adopted primarily to allow individual projects to proceed with appropriate mitigation and safeguards. More recent HCPs have attempted to address broader-based regional planning issues and, in some cases, multiple species in one plan. An example of a HCP for the federally threatened bog turtle is presented in Chap. 9.

Many listed species have specific water needs (including for temperature and seasonal water quantity). When water usage or wetland modification is incompatible with those needs, the ESA can limit water use as well as limit modification of the wetland or aquatic habitat.

The ESA declares a policy to avoid water conflicts through federal-state cooperation. 16 U.S.C. §1531(c)(2). It also requires the Fish and Wildlife Service to consult with states "before acquiring any land or water, or interest therein, for the purpose of conserving" listed species. 16 U.S.C.1535.

13.3.2.6 National Environmental Policy Act

The NEPA of 1969 requires federal agencies to undertake a comprehensive assessment of any “major federal action significantly affecting the quality of the human environment” (42 U.S.C. §4332). Major federal actions include federal leases, permits, funding and other approvals as well as actions taken directly by the federal government. Issuance of a Section 404 or Section 10 permit is subject to NEPA. NEPA does not require a federal agency to select the environmentally preferable outcome, but does require that the decision maker develop the information that makes clear the environmental consequences of its action. NEPA is designed to produce “informed” decisions. The Corps of Engineers is responsible for carrying out NEPA responsibilities for its permit programs.

Under the Council on Environmental Quality’s NEPA regulations (40 CFR 1500–1508) Federal agencies must prepare an environmental impact statement (EIS) detailing the impacts of the proposed action, any adverse environmental effects, alternatives to the proposed action, the relationship between local short-term uses of the environment and the maintenance and enhancement of long-term productivity, and any irreversible and irretrievable commitments of resources involved in the proposed action should it be implemented. If an EIS is required, the lead agency will hold a public scoping meeting to identify issues and then will prepare a draft EIS, accept public comments, and prepare a final EIS.

The regulations provide for preparation of a briefer Environmental Assessment (EA) by an agency if it is uncertain whether an EIS will be needed. EAs that result in Findings of No Significant Impact are frequently used by federal agencies to determine not to prepare an EIS, often by identifying mitigation that will keep the environmental effects below the threshold of significance. Federal agencies may adopt “categorical exclusions” (CEs) for certain categories of actions they have determined “do not individually or cumulatively have a significant effect on the human environment.” CEs can only be adopted after development of a record, public comment, and approval by CEQ.

NEPA review is generally used to integrate compliance with other environmental provisions, including the ESA, federal Executive Orders, and state and local environmental laws.

13.3.2.7 Coastal Zone Management Act Federal Consistency

The Coastal Zone Management Act (CZMA) establishes a voluntary program within the U.S. Department of Commerce (and implemented by the National Oceanic and Atmospheric Administration) that offers cost-sharing grants to coastal states, including the Great Lakes states and US territories, to develop and implement coastal zone management programs (16 U.S.C. §§1453, 1455). In addition to these financial incentives, the CZMA directs the federal government to delegate “federal consistency review” authority to each coastal state that has a NOAA-approved coastal zone management program (16 U.S.C. §§1454, 1356). Federal consistency review empowers states to review proposed federal agency activities (including permits and licenses) and to ensure that they are consistent with the enforceable policies of the state's coastal program. This power of review, the financial incentives, and the voluntary nature of the CZM Program have led 34 of the 35 eligible states and territories to participate in the Program, including all of the Mid-Atlantic states.

The statutory authority of the CZMA is confined to the "coastal zone" as defined by the state. States have the authority to designate the inland boundary of their coastal zone, which varies by state. Regardless of the size of the state's coastal zone, federal consistency review applies to any federal activity that may affect the coastal zone, whether or not the activity occurs in it. Activities performed by, on behalf of, requiring a permit from or receiving financial assistance from a federal agency, that are reasonably likely to affect the coastal zone, must comply with the enforceable state policies identified in the state's NOAA-approved coastal zone program.

Thus, consistency review will apply to Section 404/10 permits issued by the Corps within the coastal zone, as well as to activities supported by federal agencies that affect resources within the coastal zone, including freshwater wetlands.

13.3.2.8 Swampbuster Regulation

In 1985 Congress added provisions to the Farm Bill providing that persons who "converted" wetlands to produce agricultural commodity crops would become ineligible for federal agricultural payments and related benefits. This "swampbuster" provision has been carried forward, adjusted, and strengthened in subsequent Farm Bill legislation (16 U.S.C. §3821). Some of the definitions used in the swampbuster program (such as "prior converted cropland" and "farmed wetlands") have influenced the CWA Section 404 program and definitions. In general, however, it is important to recognize that one of the regulatory influences affecting wetlands on agricultural lands is the eligibility for agricultural support programs. However, if a farmer does not receive federal agricultural benefits, swampbuster will have no regulatory effect on wetland activities.

Because the swampbuster program first appeared in the 1985 Farm Bill, "prior converted croplands" are lands that were formerly wetlands and were cropped before December 23, 1985, and no longer meet wetland criteria. These lands are not subject to swampbuster restrictions. "Farmed wetlands" are wetlands that were cropped or altered prior to December 23, 1985, but that continue to meet wetland criteria. These lands may continue to be farmed in the same way they were before, with exceptions allowing further changes that produce only "minimal effects" on wetland functions. If a farmer becomes ineligible under swampbuster, but the wetland conversion was in good faith and without intent to violate the law, the law allows the farmer to engage in restoration within 1 year to avoid liability.

13.3.3 State Wetlands Regulation

States play a significant role in the regulation of activities in wetlands. Many of them operate their own permit programs, and may apply these programs to cover waters and wetlands that are not subject to federal regulation, as well as those that are. States also play a role in the review of federal permits for consistency with water quality standards as discussed below.

13.3.3.1 State Regulation Dependent on CWA Section 401

Nationally, about half the states rely solely or primarily on their Section 401 certification powers to protect freshwater wetlands—meaning that their ability to regulate, condition, or deny activities in these wetlands depends upon whether the Corps of Engineers has jurisdiction. In the MAR, however, most of the states have state laws that directly regulate activities in some or all freshwater wetlands, and hence, are not limited to Section 401 reviews (Environmental Law Institute 2008b, 2011).

Delaware and West Virginia rely on 401 in the absence of state freshwater wetlands laws. However, West Virginia has occasionally asserted jurisdiction over wetlands under its general water quality laws even where the Corps has found no jurisdiction under *SWANCC*. North Carolina also relies on Section 401 for freshwater wetlands, but in the aftermath of the *SWANCC* decision by the U.S. Supreme Court, its legislature modified North Carolina's existing 401 program to apply similar standards (under state law) to geographically isolated wetlands that fall outside federal jurisdiction. Essentially, North Carolina has enacted a limited freshwater wetlands program to pick up waters that are not covered by Corps permitting (Environmental Law Institute 2008b).

13.3.3.2 State Regulation Implementing State Wetland Laws

More common in the MAR are state laws that establish permit programs that directly regulate activities in freshwater wetlands. These permit requirements apply whether or not a Corps permit is needed, and in fact, many activities require that permits be issued both by the Corps (under CWA 404 and subject to state 401 certification) and by the state environmental agency (under state law). In most states the review processes are coordinated in order to avoid duplication of effort, and indeed, often there is a common application that serves both purposes even though the decisions are independent.

There are gaps in state regulation, however. Delaware has no freshwater wetlands permitting law. New York regulates activities in freshwater wetlands that are 5 ha (12.5 acres) in size or larger, and certain other wetlands. West Virginia lacks a wetlands regulatory program but occasionally invokes its water quality law to address activities in wetlands that escape Corps regulation. Table 13.2 reviews state regulatory programs for wetlands in the MAR.

Table 13.2 Freshwater Wetlands Programs in Mid-Atlantic states (Environmental Law Institute 2011, used by permission)

State	Authority	Waters covered
Permitting Program Covering Most Freshwater Wetlands in the State		
New Jersey	Freshwater Wetlands Protection Act	Freshwater wetlands and their buffers
	N.J. Stat. Ann. tit. 13:9, ch. 9B	Freshwater wetland definition similar to federal definition (N.J. STAT. ANN. §13:9B). The Pinelands Protection Act (N.J. Stat. Ann. §§13:18A-1), Hackensack Meadowlands Reclamation and Development Act (N.J. Stat. Ann §13:17–9), and Highlands Water Protection and Planning Act (N.J. Stat. Ann. §§13:20–1) provide additional protection for freshwater wetlands
Pennsylvania	Dam Safety and Encroachments Act 32 Pa. Cons. Stat. §693.3	Watercourses, streams, or bodies of water and their floodways wholly or partly within or forming part of the boundary of the state. Bodies of water include any natural lake, pond, reservoir, swamp, marsh, or wetland
Virginia	State Water Control Law and Nontidal Wetlands Act	State waters and nontidal wetlands
	Va. Code Ann. §62.1-44.5	Covers both waters that are regulated under the CWA and activities in nontidal wetlands that are not subject to regulation under the CWA. Federal wetland definition (VA. Code Ann. §62.1-44.3)
Permitting Program for Freshwater Wetlands, but with defined exceptions based on wetland type, size, or class		
Maryland	Nontidal Wetlands Protection Act Md. Code Ann. [Envir.] §5-902(b)	All non-tidal wetlands. MD. Code Ann., Envir. §5-901(h)(1). However, isolated wetlands of less than 1 acre and cumulative impacts of less than 5,000 square feet are exempt from permit and mitigation requirements, but require a letter of exemption (Md. Code Ann., [Envir.] §5–906)
New York	Freshwater Wetlands Act N.Y. Envtl. Conserv. Law §§24-0101	Wetlands outside the Adirondack Park greater than 12.4 acres in size and those less than 12.4 acres if they are deemed of "unusual local importance," including a 100 ft buffer. Within the Adirondack Park boundaries, wetlands greater than 1 acre in size or located adjacent to a body of water, including a permanent stream, with which there is free interchange of water at the surface. Jurisdiction over wetlands that are less than 12.4 acres in size and not of "unusual local importance" is up to the discretion of local governments. Definitions vary for wetlands outside and within the Adirondack Park. Wetlands are defined as lands and submerged lands commonly called marshes, swamps, sloughs, bogs, and flats supporting aquatic or semiaquatic vegetation (with further provisions for what constitutes wetland vegetation)
	Water Resources Law N.Y. Envtl. Conserv. Law §15-0505	Navigable waters of the state, includes marshes, estuaries, tidal marshes, and wetlands that are *adjacent to and contiguous at* any point to any of the navigable waters of the state and that are inundated at a mean high water level or tide. Wetland definition included in the Freshwater Wetlands Act (N.Y. Envtl. Conserv. Law §§24–0101)

Permitting Program for Isolated Wetlands		
North Carolina	Control of Sources of Water Pollution, Discharges to Isolated Wetlands and Isolated Waters N.C. Gen. Stat. 143–215.1, 15A NC Admin. Code 02H.1301	The state regulatory definition of wetlands states that "wetlands classified as waters of the state are restricted to waters of the United States, as defined by the Federal Code of Regulations." (NC Admin Code 02T.0103(46)). However, North Carolina regulates isolated wetlands pursuant to the "discharges to isolated wetlands and isolated waters" regulations (15A NC Admin Code 02H.1301) adopted in October, 2001
Ohio	Isolated Wetland Law Ohio Rev. Code Ann. §§6111.02	The Isolated Wetland Law establishes three tiers of regulations based on wetland categories, defined according to their ecological significance and size. The three categories are associated with different levels of review, different criteria for approval or disapproval of a permit, and different mitigation requirements. There are no minimum size thresholds for wetlands protected under the Isolated Wetland Law. Statute uses federal wetland definition
No state wetlands law, but isolated wetlands regulated case-by-case		
West Virginia	Water Pollution Control Act W. Va. Code §22-11	Under West Virginia State Code (§§22-11-3(23)—definition of waters of the state) isolated wetlands are considered wetlands of the state. Applicants must obtain any necessary approvals from the state prior to conducting activities in isolated wetlands (see http://www.dep.wv.gov/dmr/handbooks/Documents/401%20-%20Cert%20-%20Revised401%20-%2009-08-08.pdf, Personal communication with W. VA. Department of Environmental Protection, Division of Water and Waste Management, to ELI staff, Oct. 15, 2010)

State wetland laws and programs sometimes specify avoidance and minimization requirements for their state freshwater wetland laws like the federal 404 program. But these vary from state to state. Maryland, for example, requires the applicant to demonstrate that the activity is water dependent and that there are no practicable alternatives, as well as that the activity has minimized the alteration or impairment of the wetland. Pennsylvania requires the showing of water-dependence and no practicable alternative for an activity affecting an "exceptional value" wetland, but applies a lesser standard for other wetlands (where avoidance or reduction of adverse impacts to the maximum extent practicable substitutes for the requirement of water dependency).

States with regulatory programs require compensatory mitigation, and many of them have embraced mitigation banks and other compensatory mitigation programs. Many have also articulated their own goals of "no net loss" or net gain of wetlands. Some states, such as Maryland, have had little demand for compensatory mitigation because of regulatory programs that strongly emphasize avoidance and minimization. Others have supported thriving wetland banks or in-lieu fee programs.

Nationwide about one third of the states have environmental impact assessment laws (so-called "little NEPAs"). These state laws often address decisions that are not subject to review under the federal NEPA. However, most state little NEPAs are limited in focus to a very small subset of state-funded or state-sponsored activities. Only six states (only New York in the MAR) have little NEPAs that apply to a significant set of private activities conducted under state or local permits and/or to local government decisions: California, Washington, New York, Massachusetts, Hawaii, and Montana. Unlike the federal NEPA, most of these state laws have substantive requirements directing the selection of environmentally preferable outcomes unless otherwise justified, and directing implementation of feasible mitigation.

13.3.4 Local Regulation

Local governments can regulate wetlands in some states, and local governments can regulate the upland areas surrounding wetlands ("wetland buffers") in virtually all states. As many as 5,000 local governments have adopted some regulatory measures to protect at least some wetlands within their borders (Kusler 2003). While federal and state regulations require developers and others to obtain permits, state and federal coverage varies substantially by wetland type, acreage, activity, and potential impact.

Where federal and state regulatory programs do not apply or where jurisdiction is doubtful, local governments can be a supplemental source of protective authority if they have enacted suitable protective provisions. In some states, particularly in New England, state-level wetland regulation is delegated to local wetland boards to administer. And, even where federal or state programs provide for permitting of activities in wetlands, local governments still have an interest in ensuring the compatibility of the land use that occurs on and around these lands in order to maintain control of their patterns of development, community character, tax base, demand for services, and response to hazards. Many local governments have used their zoning authorities and their land use development provisions to ensure that development activities do not occur within wetland buffer areas (Environmental Law Institute 2008a).

Local government regulations tend to follow four approaches: (1) they may apply to wetlands and waters either as defined in the ordinance or in the "waters of the state" definition for the applicable state; (2) they may define specific wetland types or classes of wetlands for local protection; (3) they may apply to riparian corridors and floodways (focusing on flood and stormwater control); or (4) they may cover specifically mapped wetlands identified on a reference map (including a local zoning map or overlay district, for example) (Environmental Law Institute 2008a).

A number of local governments throughout the MAR have adopted wetland ordinances, or wetland buffer requirements to protect these resources. These include Baltimore County, Maryland, Bensalem Township, Pennsylvania, and many others. Numerous model ordinances are available (Center for Watershed Protection 2008). Frequently ordinances will cover what activities are prohibited, permitted, or conditionally permitted; what is the size of a required buffer or setback from a protected wetland or stream; what performance standards, if any, apply; and what documentation must be submitted to demonstrate compliance (Environmental Law Institute 2008a).

Approaches to wetland and wetland buffer protection may include adoption of zoning districts where wetlands and waterways are present. Activities in these districts are more closely regulated, with requirements for setbacks of buildings and parking lots from the margins of waters and wetlands, requirements for mapping and management, and limitations on impervious surface. Other approaches include environmental protections built into subdivision ordinances and construction permits. These aim to accommodate desired development or redevelopment while applying methods that protect the key resources of the municipality (McElfish 2004).

13.3.5 Compensatory Mitigation: A Closer Look

Under the CWA's §404 program, Congress assigned the day-to-day authority for issuing permits to the Corps, but assigned responsibility for developing the environmental criteria for permitting (the §404(b)(1) Guidelines) to the EPA. In 1980, the §404(b)(1) Guidelines were adopted as regulations. In 1986, the Corps adopted a comprehensive mitigation policy that applied to permit actions under §404 and under §10 of the Rivers and Harbors Act. Compensatory mitigation guidelines issued by the Department of the Army and EPA in 1990 further set out the process for mitigation. These prescribed that mitigation for impacts to wetlands and aquatic resources must be pursued in sequence. The sequence is: (1) avoidance, (2) minimization, and (3) compensation for impacts that cannot be avoided or minimized. In 1995, the Corps issued guidance on wetland mitigation banks, addressing how they should be established, approved, and monitored in providing compensatory mitigation. Finally, in 2008, the Corps and EPA adopted compensatory mitigation regulations.

The Compensatory Mitigation Rule explicitly preserves the mitigation sequence. In keeping with past practice, the Rule states that compensatory mitigation may be achieved through the restoration, enhancement, establishment, and "in certain circumstances" preservation of similar aquatic resources. It specifies, however, that restoration should generally be the first option considered, and that preservation may only be used when certain specific criteria are met. The Rule creates standards for measuring compensatory mitigation performance against ecological performance standards and requires mitigation site selection to be carried out using a "watershed approach." The Rule also includes requirements for financial assurances, permanent protection, and other measures intended to ensure the long-term conservation and management of compensatory mitigation sites. In general, compensation must be at a ratio of greater than 1:1.

A principal objective of the Rule is to create equivalent standards for all compensatory mitigation mechanisms, extending many of the requirements created for mitigation banks under the 1995 Wetland Banking Guidance to in-lieu fee programs and permittee-responsible mitigation.

Wetland mitigation banks are entities that are established to sell wetland credits to permittees needing to meet compensatory mitigation obligations. Banks are approved by an interagency review team and must meet certain performance standards and procedural requirements. In-lieu fee programs are similar, but may not necessarily have the mitigation in place or even all the mitigation sites designated in advance; however in-lieu fee programs must also guarantee performance of mitigation and long-term management, like the banks. Permittee-responsible mitigation is the traditional approach, where the permit applicant found a mitigation site (or performed the mitigation on-site) and conducted the mitigation for the specific project.

While under the 2008 Rule, the mitigation plan requirements are not identical for all three mitigation types, they are, broadly stated: "objectives; site selection criteria; site protection instruments (e.g., conservation easements); baseline information (for impact and compensation sites); credit determination methodology; mitigation work plan; maintenance plan; ecological performance standards; monitoring requirements; long-term management plan; adaptive management plan; and financial assurances."

Due to perceived advantages of mitigation banking over in-lieu fee programs and permittee-responsible mitigation, the Rule institutes an overall preference for use of mitigation banks to fulfill Section 404 compensatory mitigation obligations. Mitigation banking is given the highest preference under the Rule because "development of a mitigation bank requires site identification in advance, project-specific planning, and significant investment of financial resources that is often not practicable for many in-lieu fee programs." Mitigation banks are additionally preferred over permittee-responsible mitigation because banks "typically involve larger, more ecologically valuable parcels, and more rigorous scientific and technical analysis, planning and implementation than permittee-responsible mitigation." In-lieu fee mitigation gets the second preference, with permittee-responsible mitigation being last. Corps district engineers are given authority to alter the Rule's preference when other forms of compensation are deemed ecologically advantageous (USEPA/Corps of Engineers 2008a).

In the most recent national survey, prior to the Rule, Corps districts reported there were 405 approved mitigation banks. This represented an 85% increase in the number of approved banks in 4 years and a 780% increase in the number of banks in 14 years (Environmental Law Institute 2006). The number of banks has continued to rise since the 2008 Compensatory Mitigation Rule.

Compensatory mitigation under Section 404 commands a large outlay of funds, in many respects dwarfing the conservation outlays of state and federal agencies. It is important, therefore, to ensure that mitigation projects (banks, in-lieu fee program) are well targeted. In a 2007 report, the Environmental Law Institute determined that private and public expenditures for such compensation under Section 404 amounted to $2.9 billion annually in the United States (Environmental Law Institute 2007).

Regulatory Definitions (33 CFR 328.3)

(a) The term *waters of the United States* means

1. All waters which are currently used, or were used in the past, or may be susceptible to use in interstate or foreign commerce, including all waters which are subject to the ebb and flow of the tide
2. All interstate waters including interstate wetlands
3. All other waters such as intrastate lakes, rivers, streams (including intermittent streams), mudflats, sandflats, wetlands, sloughs, prairie potholes, wet meadows, playa lakes, or natural ponds, the use, degradation or destruction of which could affect interstate or foreign commerce including any such waters: (1) Which are or could be used by interstate or foreign travelers for recreational or other purposes; or (2) From which fish or shellfish are or could be taken and sold in interstate or foreign commerce; or (3) Which are used or could be used for industrial purpose by industries in interstate commerce
4. All impoundments of waters otherwise defined as waters of the United States under the definition
5. Tributaries of waters identified in paragraphs (a) (1–4) of this section
6. The territorial seas
7. Wetlands adjacent to waters (other than waters that are themselves wetlands) identified in paragraphs (a) (1–6) of this section
8. Waters of the United States do not include prior converted cropland … Waste treatment systems, including treatment ponds or lagoons designed to meet the requirements of CWA (other than cooling ponds as defined in 40 CFR 423.11(m) which also meet the criteria of this definition) are not waters of the United States

(b) The term *wetlands* means those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas

13.4 Landowner Incentives and Public Protection Programs

Activities in wetlands are not only subject to regulation. They are also affected by numerous governmental programs incentives and mechanisms designed to encourage conservation, restoration, and maintenance of wetland functions.

Among these are programs in the "conservation titles" of the federal "Farm bill" legislation. These have changed names and forms over time, but often consist of lease payments and/or technical assistance or cost-share funds. The Wetlands Reserve Program allows farmers to offer to conserve and maintain wetlands in exchange for rentals from the U.S. Department of Agriculture, which may support permanent easements (agreements to not change the use from wetlands), 30-year easements, or 10-year restoration and cost-share agreements (16 U.S.C. 3837–3837f). The Wetlands Reserve Program is highly dependent on the availability of sufficient federal funds to support easement and activities on lands volunteered for participations in the program. In 2008, the Farm Bill added the Wetlands Reserve Enhancement Program, to achieve additional benefits from the conserved wetlands in collaboration with participating states.

In addition to the WRP, wetlands conservation is supported by publicly funded programs like the North American Waterfowl Management Plan and North American Wetlands Conservation Act, Partners for Fish and Wildlife, the Wildlife Habitat Incentives Program, the Environmental Quality Incentives Program, and others. Wetlands are also protected through direct land acquisitions by the U.S. Fish and Wildlife Service and state agencies. The Land and Water Conservation Fund Act of 1965, and the Duck Stamp Act, along with Pittman-Robertson funding provide public dollars used for conservation of wetlands, including lands important to waterfowl.

Many private organizations, such as Ducks Unlimited, and The Nature Conservancy, are engaged in wetlands conservation through acquiring easements on lands from private landowners and managing them for ecological purposes. Federal and state tax laws provide incentives for donations of easements (often allowing deduction of the value of the easement as a charitable contribution, and in some states allowing the value of the remaining land subject to the easement to be taxed at a lower rate for property tax purposes). Some states have set up state-managed land trusts to hold conservation easements (such as the Maryland Environmental Trust, and the Virginia Outdoors Foundation).

13.5 International Wetlands Protections

13.5.1 Migratory Bird Treaty

Although this book focuses on the MAR, there are connections to international aspects of wetlands protection and conservation. Historically, the Migratory Bird Treaty Act of 1918 (Migratory Bird Convention Act of 1917 in Canada), and the subsequent Migratory Bird Conservation Act of 1929 (authorized the acquisition and preservation of wetlands as waterfowl habitat) and the Migratory Bird Hunting Stamp Act of 1934 ("Duck Stamp Act," provided an additional source of funds to purchase habitat through the sale of stamps), provided international protection of waterfowl and their breeding, migratory, and wintering habitats between the United States and Canada. Other nations signed similar treaties at later dates. The result has been the incorporation of millions of hectares of wetlands, primarily into National Wildlife Refuges in the United States (>60 million ha in 551 units, http://www.fws.gov/refuges/history/) and Migratory Bird Sanctuaries in Canada (11.5 million ha in 92 units, http://www.ec.gc.ca/ap-pa/default.asp?lang=En&n= EB3D54D1-1). There are 50 National Wildlife Refuges in the MAR many of which include coastal wetlands, and a few conserving inland wetlands (http://www.fws.gov/refuges/).

13.5.2 Ramsar Convention on Wetlands of International Importance

An international treaty first adopted in 1971 in the Iranian city of Ramsar, now lists 1,953 Wetlands of International Importance in 160 nations (Contracting Parties) totaling over 190 million ha (http://www.ramsar.org/). Ramsar promotes "the wise use of wetlands," but is not a regulatory body. Contracting parties are expected to establish wetland reserves, monitor and manage them, and submit reports every 3 years. Ramsar has been a boon to developing nations throughout the world by raising awareness about wetlands, especially where environmental laws and regulations are not well established.

In 1986 the United States became a party to the Ramsar Convention, with the first site approved in 1986. As of August 2011, there are 30 designated sites in the United States, of which 3 are in the MAR. These consist of about 110,000 ha in the coastal regions of Virginia, Delaware, and New Jersey, and overlap with some of the National Wildlife Refuges previously mentioned. More wetlands in the region and throughout the United States would certainly qualify, but submitting an application is a voluntary activity and perhaps, the significance of such a designation could be more widely encouraged.

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Chapter 14
Conservation and Management of Wetlands and Aquatic Landscapes: The Vital Role of Connectivity

Robert P. Brooks

Abstract The aquatic landscapes of the Mid-Atlantic Region (MAR) provide important ecosystem services, including ecological functions and societal values, such as floodwater storage, public water supplies, recreational greenbelts, and habitats for a diversity of flora and fauna. The connectivity of aquatic habitats is critically important for protecting regional biodiversity. Impacts may be localized in nature, but as the aquatic and terrestrial portions of a watershed are altered, the viability of these connections, through riparian corridors and proximal patches of natural vegetation, can be negatively affected. Although natural processes can retard succession (e.g., severe floods, fire, disease and insect epidemics), in the northeastern USA natural disturbances typically create a quilt-like mosaic of recovering habitat patches comprised primarily of natural vegetation. In contrast, human-induced land use changes are more likely to result in larger and more permanent alterations over time, with a resultant loss of habitat. Maintaining connectivity among wetland, riparian, and stream habitats by protecting or restoring corridors among these habitats has proven to be a viable approach to conservation. The connectivity requirements for a range of taxa are reviewed, as are planning tools and programs for conserving and restoring connectivity among aquatic habitats.

14.1 Introduction

As emphasized throughout this book, aquatic landscapes are a collection of wetland, riparian, and riverine habitats connected by the movement of water, carbon, nutrients, and biota. Flora and fauna move within and among various habitats and,

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consequently, the biological integrity of a given area depends on factors that affect species movements, reproduction, and survival. This concept has been frequently discussed in various chapters, including Chap. 1—Aquatic Landscapes, Chap. 2—Hydrogeomorphic (HGM) Classification, Chap. 8—Birds, Chap. 9—Amphibians and Reptiles, and Chap. 10—Aquatic Macroinvertebrates.

From a conservation perspective, protecting the *full complement* of biodiversity within a system of bioreserves could be considered the "gold standard." As stated eloquently by E. O. Wilson, "We should judge every scrap of biodiversity as priceless while we learn to use it and come to understand what it means to humanity." (1992:351). Conservation and management goals to preserve biodiversity often include the protection, restoration, or creation of a highly interconnected system of core reserves and corridors (e.g., Bennett 2003; Gilbert-Norton et al. 2010). So, in this chapter, I review the habitat connectivity requirements for a variety of stream, wetland, and riparian taxa, with an emphasis on corridors. A listing of specific planning tools and selection of existing conservation programs is presented.

14.2 Landscape Patterns and Species Movements

Connectivity is a key concept in conservation science where goals typically are to protect and restore landscape patterns that promote habitat corridors for species, communities, and ecological processes in environments modified by human activities and impacts. Of the many definitions available for the term *connectivity* used in this context, Taylor et al. (1993) provide a simple one, "…the degree to which the landscape facilitates or impedes movement among resource patches." That is, daily use, dispersal, and migratory movements, by faunal (and floral) species that require more naturalistic habitats, are fostered by a higher level of continuous or near continuous *connectivity* among their required resource patches (e.g., breeding, foraging, resting, and wintering sites).

Connectivity among aquatic habitats has been shown to affect both faunal (e.g., Gibbs 1993; Calhoun and deMaynadier 2007) and floral communities. For example, movements of vulnerable species can be hindered by dams, dikes, and culverts (e.g., detrimental to dispersing bog turtles, *Glyptemys muhlenbergii*) and discontinuities among requisite habitats can affect reproductive success and genetic diversity. Thus, a review of how species benefit from highly interconnected aquatic ecosystems is informative.

Within a riverine network, where most freshwater wetlands are found in the MAR, fish and aquatic macroinvertebrates use different habitats at different times of the day, year, and phases of their life cycle. For example, aquatic macroinvertebrates move downstream with the water column, a process known as "drift." Invertebrate drift rates have been shown to have a diel periodicity, with higher rates at night and peaks near dusk and dawn (Waters 1965). Vertebrate predators have been shown to respond to these diel drift patterns (Griffity 1974; Hughes 1998). Drift has been classified as active or passive depending on whether species intentionally enter the drift as a dispersal mechanism in response to food availability or predation risk, or accidentally with flow (Allan 1995). However, although the relative importance of these two types of drift is still debated, the evidence is clear that drift is an important process in headwater streams and larger rivers.

Fish also move longitudinally within the stream network. The most obvious examples are taxa that migrate upstream to breed, including many species of salmonids (trout) and catastomids (suckers). However, because all fish vary dramatically in size from embryo to adult, most species exhibit complex life cycles and habitat use patterns over the length of their life cycles that are mediated by migration (Schlosser 1995a). Moreover, many fish species have diel and seasonal migratory behaviors in response to food availability and natural variation in temperature and flow (Albanese et al. 2004). The size and distribution of mesohabitats (riffles and pools) within the stream channel have also shown to be important determinants of short-term fish movement patterns in headwater streams (Lonzarich et al. 2000).

Obviously, any physical barrier to downstream movement would affect drift and other movements by biota. Natural barriers such as waterfalls and beaver dams, and artificial barriers such as human constructed dams have been shown to affect invertebrate drift rates and fish migration (Radford and Hartland-Rowe 1971; Schlosser 1995a; Schlosser 1998). Moreover, invertebrate drift rates have been shown to correlate with increases in flow (Bosco and Perry 2000) and droughts (Cuffney and Wallace 1989), water temperature (Dudgeon 1990), light levels (Anderson 1966), and water quality (Wallace et al. 1987; Beltman et al. 1999). Urban and agriculture land uses in upstream portions of the watershed have been shown to indirectly influence fish migration patterns through their effects on stream flow, temperature, water quality, and distributions of stream habitat including the availability of hydrologic and thermal refugia (Schlosser 1995a; Roth et al. 1996; Pollino et al. 2004). Thus, in addition to the obvious effects of physical barriers, disturbances that alter flow, temperature, food availability, or water quality would also be expected to alter downstream movement of aquatic invertebrates and vertebrates (e.g., Snyder et al. 2003).

Lateral movements of species among upland, riparian, wetland, and stream habitats are equally important in riverine systems (see Chaps. 2 and 10). Several species of fish either breed in or use the aquatic beds found fringing along lakes, reservoirs, and meandering rivers for juvenile cover (e.g., bass, Centrarchidae). Others will use more densely vegetated wetlands that have at least some standing water (e.g., pickerel, Esocidae). Aquatic insects, by far the largest component of the macroinvertebrate community in streams in terms of diversity and productivity, spend most of their life cycles in the stream environment. However, many species emerge from the stream as winged adults to mate and lay eggs, mostly in large mating swarms. The biomass of aquatic insects emerging from streams represents a significant energy source for riparian birds, mammals, and spiders and, therefore, represents a return of a significant amount of energy from the stream, back to the riparian area in which it was originally derived (Jackson and Fisher 1986). In addition, dispersal and oviposition of winged adults is the principal route of recolonization of streams denuded by floods, droughts, or pollution (Sheldon 1984). Lateral movements by facultative invertebrates that adapt annually or seasonally to hydrologic conditions within rivers and their floodplains are, likewise, important (S. Yetter, pers. comm., see Chap. 10). Although there has been relatively little research into factors influencing adult dispersal of aquatic insects, some studies suggest that the amount and composition of riparian forests are important determinants of dispersal distance and colonization success (e.g., Collier and Smith 1998; Briers et al. 2002). Thus, disturbances that alter the structure and composition of forested wetlands and riparian forests, especially riparian banks, would be expected to reduce the capacity or rate in which headwater streams recover from additional natural or anthropogenic disturbances.

Semiaquatic species are also strongly influenced by the amounts, conditions, and spatial relations between upland, riparian, and stream and wetland habitats. In particular, most amphibian species require both terrestrial and aquatic habitats at various times during their life cycles. Some regionally important amphibian taxa such as several species of mole salamanders (*Ambystoma spp.)* and some anurans such as wood frogs (*Rana sylvatica*) spend most of their lives in terrestrial habitats, but use vernal pools (i.e., isolated depressions) and other wetland types for breeding and larval nursery habitat (Semlitsch 2000). Other species such as the red-spotted newt (*Notophthalmus viridescens*) spend most of their adult lives in aquatic habitats but spend 1–2 years in terrestrial habitats as immature efts (Forester and Lykens 1991). As a result of this biphasic life history, amphibians depend on relatively un-degraded terrestrial and aquatic components of the ecosystem to complete their life cycles. Typically, they use breeding pools for only a couple of weeks each spring before returning to adjacent forests for the remainder of the year. Eggs and larvae are only in pools for a month or two before metamorphs (newly emerged terrestrial forms) migrate to forests to forage in preparation for winter hibernation (Calhoun and deMaynadier 2007; Semlitsch and Skelly 2008) (Fig. 14.1). Moreover, the integrity of migration routes among habitats is critical in maintaining viable populations (Cushman 2006). The conversion of forest habitats to agriculture or urbanized landscapes and increased density of roads have all been shown to disrupt dispersal and migration corridors of amphibians (Gibbs 1993; Joly et al. 2001; Guerry and Hunter 2002). Snyder et al. (2005) found that pond use by all three species of mole salamander found in the Delaware River National Recreation Area was negatively correlated with primary roads.

In addition to aquatic and terrestrial habitat quality and intact migration routes, the size and isolation of breeding habitats have also been shown to be important landscape characteristics to amphibians (Julian 2009). In contrast to most other faunal groups, pond-breeding amphibians do not show a positive relationship between habitat size and assemblage diversity. That is, smaller wetlands and ponds are disproportionately important to this group of animals because they are typically ephemeral and consequently do not support fish and other vertebrates that prey on amphibian larvae (Snodgrass et al. 2000). These abundant, small wetlands also can function as stepping stones for dispersal and recolonization of locally extinct populations (Semlitsch and Bodie 2003), with individuals traveling distances of at least 300–1,000 m either between ponds or to and from foraging habitats (Patrick et al. 2006; Gamble et al. 2007; Rittenhouse and Semlitsch 2007; deMaynadier and Houlahan 2008). Despite their importance, small wetlands are usually more vulnerable to filling and draining associated with development because they typically lack

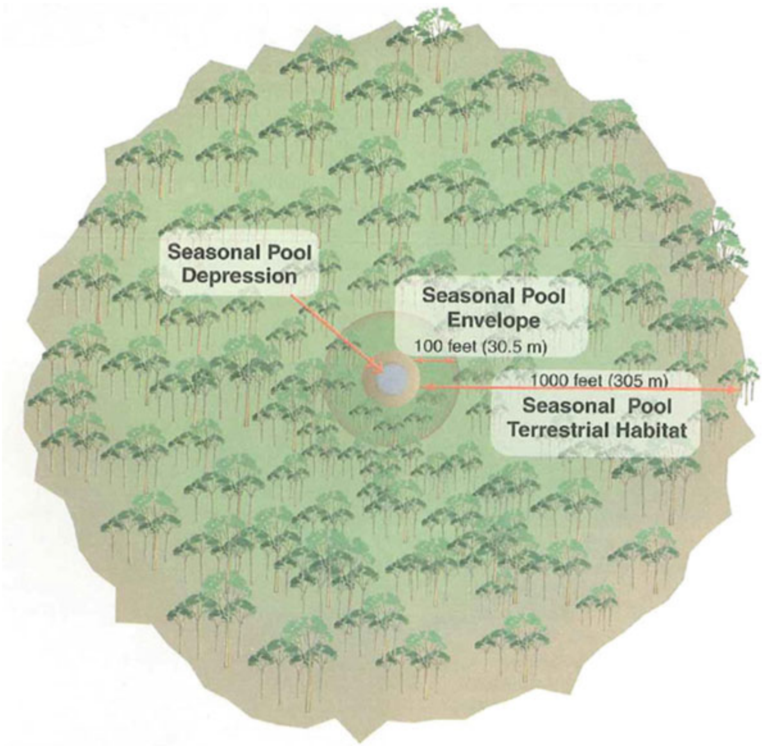

Fig. 14.1 Requisite habitats for pond-breeding amphibians around an isolated depression (vernal pool), showing the importance of maintaining extensive forest cover (Brown and Jung 2005)

state or federal regulatory protections (Semlitsch and Bodie 2003; Calhoun and deMaynadier 2007). Beyond direct habitat destruction, wetlands in general, and small wetlands in particular, are sensitive to changes in weather patterns.

A variety of birds and mammals use riparian areas as habitat, and several species and selected guilds have been shown to respond to degradation of these ecosystems (Croonquist and Brooks 1991, 1993; Brooks et al. 1998). The Louisiana waterthrush (*Parkesia motacilla*), one of the few obligate songbird species present in forested headwaters and wetlands throughout the region, serves as an integrative indicator of condition because of their dependence on interior forest habitat and clean, headwater streams (Prosser and Brooks 1998; O'Connell et al. 2003, see Chap. 8). Waterfowl, shorebirds, and other waterbirds use the mosaic of wetlands and waterbodies throughout the MAR. Beaver activities frequently alter the entire structure and function of headwater streams and wetlands, creating a variety of habitats for other species, including other semiaquatic mammals such as muskrat (*Ondatra zibethicus*), mink (*Mustela vison*), raccoon (*Procyon lotor*), and river otter (*Lontra canadensis*) (Fig. 14.2). The strong influence of the surrounding landscape on a wetland's or stream's ability to perform biologically related functions has become increasingly evident (e.g., Gibbs 1993; Wardrop and Brooks 1998; O'Connell et al. 2000; Strayer et al. 2003).

Fig. 14.2 Oblique aerial photograph of an extensive wetland complex created by decades of beaver activities in central Pennsylvania (photograph by R. Brooks)

14.3 Riparian Corridors

The dynamic processes that define the interface between terrestrial and aquatic ecosystems, including stream, floodplain, wetland, and upland components, necessarily result in a diverse collection of proximal habitat patches (Forman 1995). Not only are these habitat patches closely packed in space, but they are also subject to change over time through natural processes. The resultant collection of diverse habitats yields diverse biological communities. How humans interact with a landscape within the physical constraints of climate and geology defines land use. Patterns that arise along riparian corridors impact the structure and function of aquatic ecosystems, and hence, the integrity of their biological communities (Jordan et al. 1993; Castelle et al. 1994; Sweeney et al. 2004; Walter and Merritts 2008). Wider forested riparian corridors support a higher abundance of macroinvertebrates and process more carbon, nitrogen, and pesticides than narrower reaches. Attributes of riparian corridors, as measured by ecological indicators, can serve as estimates of condition (e.g., King et al. 2005; Brooks et al. 2009).

The question of "How wide should a riparian corridor be?" is often asked, but seldom answered satisfactorily, because complex ecological processes and societal decision-making are involved. If one uses the idiom of "wider is better," and focuses on a width that encompasses an array of ecological concerns, then an appropriate range of widths can be determined. This is, of course, highly dependent upon the conservation and management objectives for a given area. Given the moderate to high levels of development (e.g., towns, highways, railroads) outside of many riparian corridors in the MAR, an appropriate objective might be "as wide as possible given the surrounding land use constraints." A review of the literature indicates that a naturalistic riparian corridor >300 m (including the river channel and both sides) will provide both interior conditions and dispersal pathways. A width of this dimension is recommended because edge effects (e.g., changing microclimates, increased predation and parasitism, etc.) can penetrate interior habitats of any type by 30–100 m (Environmental Law Institute 2003). Also, if the landscape corridor in question serves as both habitat for resident species and a pathway for dispersing and migratory species, then it needs to be sufficiently wide to maintain suitable interior conditions, whether these are forests and/or wetlands. The maximum expected complement of bird species within a community has been found when forested corridors 100–300 m wide were present (Bierregaard et al. 1992; Croonquist and Brooks 1991, 1993; O'Connell et al. 2003).

Along rivers and floodplains, natural hydrologic and hydraulic processes can create an abundance of meanders both along the stream channel and appearing as historic oxbow lakes and wetlands within the floodplain. Wide-ranging species that are not wetland dependent (coyote (*Canis latrans*), gray fox (*Urocyon cinereoargenteus*), bald eagle (*Haliaeetus leucocephalus*)), some requiring primarily interior forested habitats (fisher (*Martes pennanti*), bobcat (*Lynx rufus*), black bear (*Ursus americanus*)), still occur in the MAR. Carnivores that are dependent on natural riparian banks also can be present (river otter (*Lontra canadensis*), mink (*Mustela vison*), northern water shrew (*Sorex palustris*), belted kingfisher (*Ceryle alcyon*), northern water snake (*Nerodia sipedon*)). Other species found in the region are dependent on large mature trees, such as bat species roosting under bark, flying squirrels using tree cavities, and wood ducks (*Aix sponsa*) nesting in tree cavities. Vernal pool obligates such as wood frog (*Rana sylvatica*), spotted and Jefferson salamanders (*Ambystoma maculatum*, *A. jeffersonianum*), and facultative species like wood turtle (*Glyptemys insculpta*) and star-nosed mole (*Condylura cristata*) use these isolated wetlands throughout the region, and are especially abundant where clusters of vernal pools are found in proximity to riverine systems. Species that typically use emergent wetlands occur, including bird species of concern (bitterns; rails; sedge and marsh wrens, *Cistothorus platensis*, *C. palustris*), as well as frogs, turtles, meadow voles (*Microtus pennsylvanicus*), and house-building muskrats (*Ondatra zibethicus*). In the Appalachians, high species richness occurs among most truly aquatic taxa, including stream and wetland insects and crustaceans, mussels, dragonflies and damselflies, fish, and aquatic vascular plants (e.g., Abell et al. 2000). Collectively, this diverse array of stream, riparian, wetland and upland habitats supports a large number of species of concern and species that require relatively connected and continuous forested landscapes along riparian corridors.

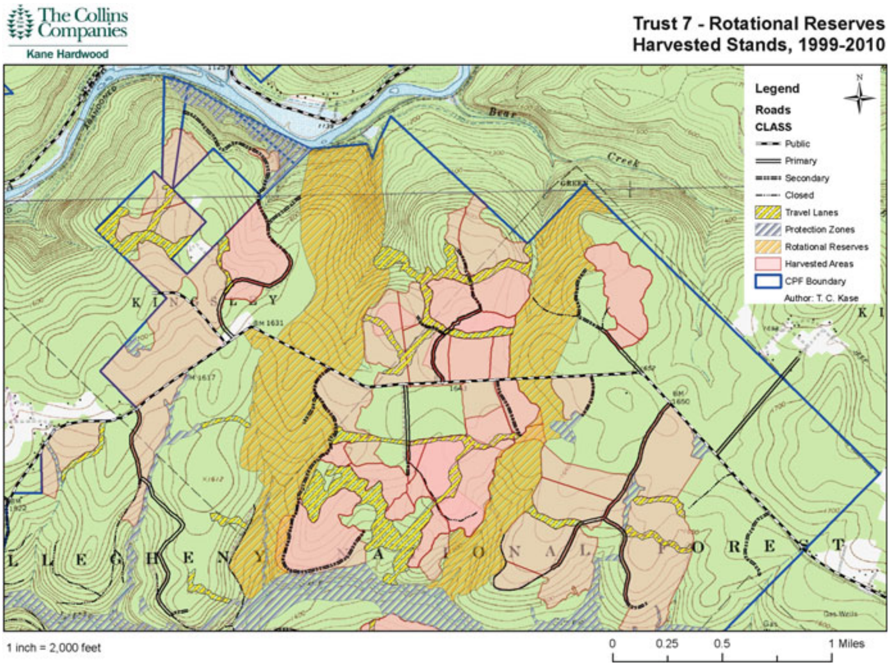

Fig. 14.3 Landscape corridors implemented on private forestry lands of Collins Pine Company northcentral Pennsylvania to meet sustainability guidelines

As stated previously, minimum dispersal distances from vernal pools for amphibians reportedly range from 100 to 300 m (Semlitsch and Bodie 2003; deMaynadier and Houlahan 2008). Reptiles and small mammals will use comparable corridors, but large mammals, particularly carnivores, may require wider and more continuous connectivity (Hargis et al. 1999; Gomper 2002; Hubbard and Serfass 2005; Sinclair et al. 2005). This was the intent of conserving a corridor in northcentral Pennsylvania. Brooks and others (unpublished) designed and implemented two, 300-m wide landscape corridors on a 4,000-ha parcel owned by Collins Pine Company, Kane, PA. The intent was to maintain a closed canopy forest in these corridors while sustainable forest harvesting occurred in the surrounding terrain. The corridors connected permanent reserves along major river floodplains, traversing headwater streams on either side of an upland hill (Fig. 14.3). After a decade, the corridors remain intact, and the bird community, as assessed using the Bird Community Index (O'Connell et al. 1998), has maintained the characteristics of bird communities found in mature, core forests.

Thus, an overall recommendation for a diverse faunal community is to protect, conserve, and restore the forested riparian corridor to a width of >300 m wherever possible. Encroachments, gaps, and early successional portions will occur within such a corridor, but the number and size of these should not be increased, and should be reduced using appropriate restoration strategies. Brooks et al. (2006, 2009) have used observations of stressors occurring in the buffer around wetlands, streams, and riparian areas as a means to assess the condition of these resources, and as a way of identifying potential restoration targets for maintaining adequate buffer widths.

When considering how various stressors influence aquatic landscapes, it is instructive to consider deviations from reference standard conditions that support the highest levels of biological integrity. In the eastern United States, the best attainable conditions for aquatic systems are usually derived from a landscape dominated by mature forests, which produce characteristic inputs of organic matter, shade over wetlands and narrow stream corridors, and habitat for an expected set of species. In the floodplains of larger rivers, microtopographic heterogeneity arises from the interplay of hydrologic forces, vegetative structure, and underlying soil characteristics. The resultant mosaic of wet and dry patches found in natural floodplains and along the interfaces between aquatic and terrestrial systems supports a diversity of biological communities adapted to wetting and drying cycles. These physical and biological complexities interact with and upon the materials present through biogeochemical processes to produce the ecological functions and services recognized from these systems.

That said does not mean that only forested landscapes should prevail in the MAR; they do not now, and are unlikely to in the future, given the millions of people that live in the region. Reference domains can exist for all major types of land use; forested, natural herbaceous, mixed, agricultural, and urban (Brooks et al. 2005). One set of goals could be to strive for the best possible spatial arrangement of habitat types that included natural or semi-natural vegetation in large, unfragmented blocks, patches, or corridors. To protect the structure and functions of the region's aquatic and terrestrial ecosystems, this *green infrastructure* needs to exist within the realm of the *built infrastructure* we have created since European settlement began in the early 1600s.

14.4 Planning for Habitat Connectivity

There are numerous legal instruments and planning tools available to enhance the conservation and management of aquatic ecosystems. In Chap. 13, policies, laws, and regulations for protecting wetlands and other waters were reviewed. Beyond these legal mandates, such as regulations, zoning authority, and mitigation requirements, there are a series of voluntary options that can be implemented by resource managers and individual landowners. Permanent conservation of wetlands, waters, and surrounding lands can be accomplished by outright fee simple purchase or through gifts of parcels to conservation agencies or organizations that agree to maintain conservation values in perpetuity. Alternatively, easements, which provide reduced tax burdens to participants, have a similar outcome, but the landowner maintains ownership and an agreed upon bundle of rights (e.g., life interest, management authority, etc.). The Land Trust Alliance has compiled a set of options on a state-by-state basis (2011, http://www.landtrustalliance.org/policy/tax-matters/campaigns/state-tax-incentives). An example from Downeast Maine demonstrates how these approaches can be applied. The Frenchman Bay Conservancy and its partners are seeking to create a landscape corridor for wildlife spanning the Schoodic

Peninsula from north to south. The goal is to use land purchases and easements to construct corridors that is beneficial for both terrestrial and semiaquatic carnivores. Research projects by Church (2011) and Smith (2012) have assisted the Conservancy in locating suitable parcels by transforming habitat models into proposed landscape corridors that link existing and potential reserves (Fig. 14.4).

On a less permanent basis, restoration and creation of wetlands or other ecosystem components is widely embraced and practiced by a wide spectrum of organizations and individuals. A review of restoration options for wetlands is provided in Chap. 12. Funds to implement these practices are available through a variety of federal, state, local, and private grants, incentives, and cost-sharing programs (e.g., Environmental Law Institute 2003, see Chap. 13).

Landscape-level conservation, restoration, and management goals are being actively pursued throughout the MAR by:

- States
  - Maryland's GreenPrint Program, http://www.greenprint.maryland.gov/
  - Virginia Outdoor Foundation, http://www.virginiaoutdoorsfoundation.org/
  - Delaware's Green Infrastructure Program, http://www.dnrec.delaware.gov/GI/Pages/GIPlanning.aspx
- Regional land trusts
  - Pennsylvania's Smart Conservation Program, www.natlands.org/services/for-municipalities/smartconservation/
- Regional partnerships
  - Chesapeake's Bay Bank, http://www.thebaybank.org/
- A variety of national, international, and conservation programs
  - Watershed assistance grants programs
  - National Fish and Wildlife Foundation, http://www.nfwf.org/
  - Natural Resources Conservation Service's Wetland Reserve Program, http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/programs/easements/wetlands
  - Ducks Unlimited, http://www.ducks.org/
  - The Nature Conservancy, http://www.nature.org/
  - Ramsar Convention on Wetlands of International Importance, http://www.ramsar.org/

This list is only a small representation of the array of programs available in the MAR. So, how do we focus these efforts is a scientifically valid approach, to optimize their effectiveness in conserving and managing aquatic ecosystems?

Connectivity along any natural corridor can be achieved in two ways: (1) by managing the entire landscape associated with the corridor to facilitate movements, and (2) by maintaining specific habitats that assist species in their movements through less than optimal or inhospitable patches. In the latter case, the connectivity can be achieved by either maintaining continuous connections among suitable habitats, or having "stepping stones" (small patches) in proximity to each other, such that movements among patches are feasible (Bennett 2003). The required spatial arrangement necessarily varies among species based on their size, site fidelity, and dispersal or migratory distances. Vernal pool amphibians are much more sedentary

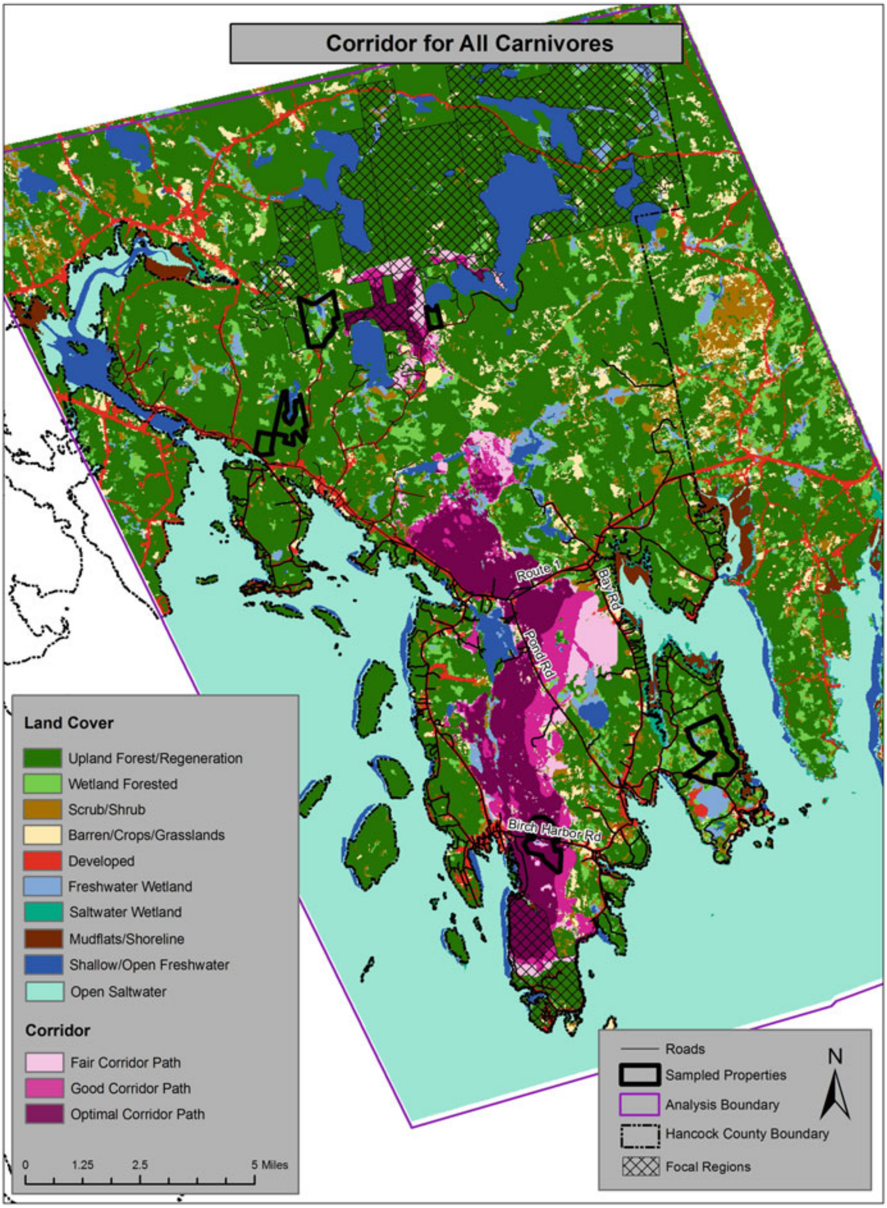

Fig. 14.4 Proposed landscape corridors in Downeast Maine designed to facilitate movements of terrestrial and semiaquatic carnivores using land acquisition and easement techniques (map produced with GIS analysis by A. Church (2011))

throughout their life cycle than migratory birds, but each benefits from having intact naturalistic habitat mosaics within their daily, seasonal, or lifetime home ranges. When landscape connectivity is lost or degraded, then populations of species that require these features are either (1) reduced in number or (2) subjected to increased isolation from other populations, thereby increasing the risk of local extinctions (Bennett 2003).

Most human-caused disturbances set back ecological succession to early stages. That is, for varying lengths of time, mature forests and large trees along riparian corridors will be absent, soil formation may be retarded, and the composition of floral and faunal communities will be different. Although natural processes also retard succession (e.g., severe floods, fire, disease and insect epidemics), in the MAR these typically create a quilt-like mosaic of recovering habitat patches. Organisms persist in neighboring refugia and then recolonize the recovering habitats at an appropriate time. In contrast, human-induced land use changes are more likely to result in larger (e.g., residential subdivisions, farm fields, commercial development) and more permanent alterations (e.g., any impervious surfaces, hard engineering structures, soil compaction from use of heavy equipment, substitution of annual (crop) and perennial (lawns) vegetation for woody species), and introduce ornamental, invasive, and/or exotic species. If the remaining biodiversity of the region is to be conserved, then land use policies and practices should be evaluated carefully as to their true environmental benefits and costs.

As humans continue to transform the landscape of the MAR and elsewhere, forest cover will be generally reduced, replaced by agricultural, suburban, and urban land uses linked through transportation and utility corridors. The spatial extent and pattern of these changes determines the degree of alteration and degradation observed in aquatic landscapes. Additionally, point sources of urban stormwater, agricultural runoff, and other pollutants can severely degrade these systems. Degrees of change can be detected through monitoring and assessment if selected attributes are used as indicators or vital signs. If the desired spatial mix and connectivity of natural habitats and human-influenced land uses can be determined, then land use policies and management practices can be focused on achieving those goals. The common thread to consider when planning land use policies and practices for wetlands, streams, rivers, and lakes is to treat aquatic landscapes holistically rather than as a set of separate, disconnected components.

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