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CHARACTERIZING HYDROLOGIC SETTINGS AND HYDROLOGIC REGIMES OF HEADWATER RIPARIAN WETLANDS IN THE RIDGE AND VALLEY OF PENNSYLVANIA

A Dissertation in

Geography

by

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ABSTRACT

Riverine systems are highly complex and dynamic systems. They have been described as having a four-dimensional nature of fluxes of materials, energy, and biota in a riverine system through: lateral connectivity between the channel and the riparian area or floodplain; longitudinal connectivity between channel segments upstream and downstream; vertical connectivity between the channel or floodplain and the aquifer; and temporal dynamics of the system. Floodplains can be seen as the 'beads on a string' of the riparian corridor where maximum three-dimensional hydrologic exchange allows for the fluxes of energy, materials, and biota within and between heterogeneous habitats. Building on Landscape Ecology and Fluvial Geomorphology a River Ecosystem Synthesis (RES) Framework looks to understand rivers as dynamic shifting mosaics of geomorphic patches or Functional Process Zones. Wetlands are a key part of this shifting mosaic, but have been under characterized and underrepresented in the RES, primarily due to a lack of cross-pollination between wetland and stream ecology. This research intended to help nudge the discussion of wetlands towards the RES Framework and help include wetlands in the integrated approach being pursued by stream ecologists under the RES. Additionally, there is a general need to better understand the hydrologic setting and dynamics of headwater wetlands.

Drawing upon the opportunities to inform and be informed by the stream ecology literature by looking at wetlands in a 4-dimensional, dynamic, River Ecosystem Synthesis approach, this study sought to: better understand the hydrologic setting and the seasonal dynamics of the hydrologic regime for headwater wetlands in the Susquehanna River Basin. The objectives were achieved through a synthesis at the basin scale (Chapter 2)

and detailed analyses in a single sub-watershed (Chapter 3) and a subset of wetlands with long-term water level records (Chapter 4). In Chapter 2, this work used the framework of the four-dimensional nature of the RES to generate a revised approach to scaling for the study of wetland services in the Susquehanna River Basin. One aspect of the revised scaling hierarchy was the use of a reach-scale. Chapter 3 used the understanding that in river ecosystems wetlands and, in turn high bio-complexity, occurs where there are laterally and longitudinally unconstrained reaches. This research identified what topographic characteristics defined unconstrained reaches in this physiographic setting based on the known occurrence of wetlands in the study area. Chapter 4 explored the vertical and temporal dimensions of wetlands in this physiographic setting by exploring the relationships between water level, wetland type, and seasonal fluctuations across years with a range of drought and deluge conditions using time series analysis.

Chapter 2 is a synthesis of the efforts of the ecological portion of a study of climate change and ecosystem services provided by freshwater wetlands of the Susquehanna River Basin study to characterize and map a hierarchical landscape classification for use in the study. The scaling hierarchy analysis not only identified a gap in spatial scale of data between disciplines, but it identified the reach as a scale to bridge that gap. Building upon several existing classification schemes, a revised hierarchical landscape classification was generated: Basin, Physiographic Province, Sub-watershed, Channel Reach, and Habitat (macro- and micro-). This work not only proposed the use of a reach scale, rare in wetland studies but very common in stream studies, but articulated a process-based macro- and micro-habitat classification.

Chapter 3 improved the spatial prediction of headwater riparian wetlands through identification of reach settings unconstrained latitudinally and longitudinally that allow for this three dimensional exchange of water. Known locations of mapped, non-open water National Wetlands Inventory (NWI) wetlands, field-identified non-NWI wetlands, and non-wetland locations (n=40, 30, and 35, respectively) were used to build a predictive partition tree. Predictive variables were DEM-derived topographic indices for the stream reaches: valley width, mean stream slope, and contributing area. The partition tree resulted in a 5-node tree (overall R^2=0.61). These classes ranged from very high likelihood of wetland occurrence to very low likelihood of wetland occurrence or least constrained to most constrained. This classification is a useful approach to characterizing wetland and non-wetland reach settings, especially in screening out the least likely wetland-supporting or most constrained reaches within a watershed.

Chapter 4 used a suite of time series analyses to explore the hydrographs of five headwater wetlands in terms of their dynamics and response to climatic drivers. Cross correlations between daily differences in water levels and precipitation showed significant correlations for most wetlands under dry and wet conditions on the same day time lag. Of the wetlands evaluated, all experienced a summer drawdown in water level except for the wettest sites in the wettest years. Further, the timing of the beginning of summer drawdown varied greatly for the period of record for the three studied wetlands (slope=61 days; headwater floodplain=58 days; and riparian depression=91 days) with the slope wetland drawing down earlier on average than headwater floodplain or slope wetlands (average day of the year 132, 156, 152 respectively). The moving averages of the water levels generally followed the trends of the downstream stream baseflow, except

for the wettest site in the wettest year. Though the hydrologic data are only available as a highly discontinuous record over a 10-year period, more continuous records when analyzed as case studies with time series analyses can give insight into the dynamics and responses of hydrologic behavior of headwater wetlands to climatic drivers.

This work is one step in a process to integrate scientific understanding and management of wetlands, floodplains, and streams as one inter-connected system. The River Ecosystem Synthesis Framework encourages this multi-system approach to understanding the 4-dimiensional nature of river ecosystems through the integration of principles and approaches of fluvial geomorphology and landscape ecology into stream ecology. However, the nature and functioning of wetlands in the river ecosystem needs to be incorporated into and advanced by this framework. The work in this study helps to move this integration forward by exploring the four-dimensional nature of headwater wetlands in the river ecosystem, which can contribute to this critical, but limited literature at the nexus between streams and wetlands.

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DEDICATION

To Nancy Brown: scholar, friend, curmudgeon, and lover of life. Her extreme curiosity, dogged work ethic, razor sharp wit and sarcasm, low tolerance for academic shenanigans, and big heart made her an incredible companion for the grad school experience. I was lucky to know her. She was exceptional and is deeply missed.

Chapter 1

CONCEPTUAL FRAMEWORK

‘The sustainable conservation of diversity in riverine wetlands implies knowledge of the basic geomorphological and ecological processes that inter play at the landscape scale.’ - Bornette et al. 1998

BACKGROUND

Riverine systems are highly complex and dynamic systems. Ward (1989) eloquently described this as the four-dimensional nature of lotic systems by describing the fluxes of materials, energy, and biota in a riverine system through three spatial dimensions, plus time. Thus, the four dimensions are: *lateral* connectivity between the channel and the riparian area or floodplain; *longitudinal* connectivity between channel segments upstream and downstream; *vertical* connectivity between the channel or floodplain and the aquifer; and *temporal* dynamics of the system ranging from short term or ‘behavioral response’ to long term ‘evolutionary change’.

Floodplains can be seen as the ‘beads on a string’ of the riparian corridor where maximum three-dimensional hydrologic exchange allows for the fluxes of energy, materials, and biota within and between heterogeneous habitats (Figure 1) (Stanford and Ward 1993, Ward et al. 2002). These beads are comprised of aquatic habitats (both mainstem and slack waters) and the lower and upper aquatic-terrestrial zones driven by below bank or flow pulse flows and overbank or flood pulse events respectively (Junk et al. 1989).

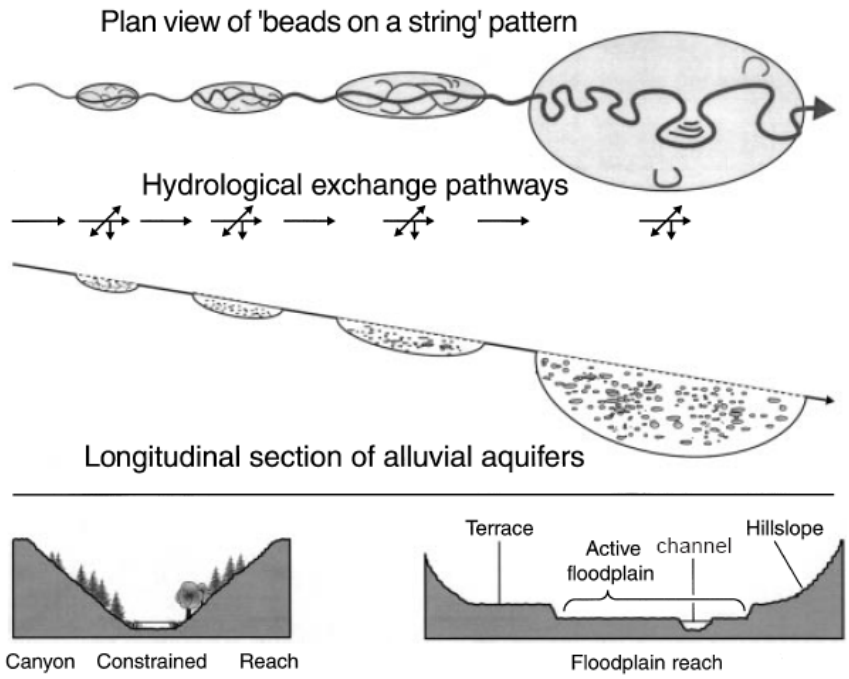

Figure 1. Diagram representing an idealized 'bead on a string' configuration of a riverine network with alternating constrained and unconstrained (floodplain) reaches (from Ward et al. 2002).

Thorp et al. (2006) lay out a framework for studying and understanding biocomplexity of riverine systems as the River Ecosystem Synthesis Framework. They defined biocomplexity as "structural and functional phenomena arising from dynamic interactions amongst biotic elements of the ecosystem and between these components and the physiochemical environment" (Thorp et al. 2006: 124). This work built upon the view of river network as a primarily unidirectional gradient from headwater to mouth as presented in the River Continuum Concept (Vannote et al. 1980) and incorporated the importance of the lateral connection of the stream with the flood plain (Junk et al. 1989). Further, they proposed a combined approach to the understanding of riverine systems by combining the ecological aspects of fluvial geomorphology with the patch dynamics models used in terrestrial landscape studies. They conceptualize a riverine system as a

dynamic shifting mosaic (Stanford et al. 2005) of geomorphic patches or Functional Process Zones. They state that linking the appropriate spatio-temporal scales with ecologically-relevant processes and identifying spatiotemporal patterns and hierarchies are important avenues for research in riverine systems.

Landscape ecology can inform the study of riparian systems in several key ways: through understanding the hierarchical nature of scaling in ecosystems, through the focus on process in ecosystems, and through the exploration of patch dynamics (Ward 1989). Riparian systems have often been thought of as simply corridors in the landscape and not as complex and diverse landscapes in their own right (Poole 2002, Thorp et al. 2006). There is increasing dialogue about incorporating approaches from landscape ecology in the study of riverine systems (Ward et al. 1999), with a subfield of 'Fluvial Landscape Ecology' proposed as an avenue of study for integrating approaches from geography, hydrology, and ecology (Figure 2) (Poole 2002).

The growing dialogue about a 'River Ecosystem Synthesis Framework' or a 'Fluvial Landscape Ecology', however, is primarily in the stream ecology literature. It notably does include some discussion of floodplains, but only marginally addresses the role of wetlands in a riverine system. Further, as headwater systems comprise 60-75% of the length of streams in the United States (Leopold et al. 1964) and are thought to set the biogeochemical state for downstream systems (Brinson 1993) there needs to be an increased understanding of the role of headwater systems in a landscape perspective on riverine systems (Wantzen et al. 2008).

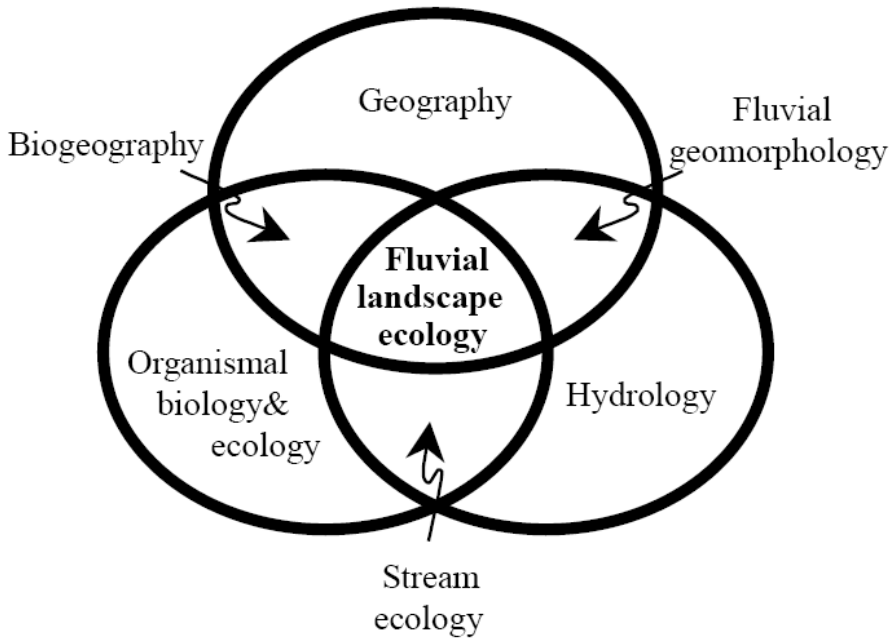

Figure 2. Diagram showing the foundational fields for a proposed sub-discipline 'fluvial landscape ecology' (from Poole 2002).

Wetland literature has long recognized that landscape position or geomorphic setting is intrinsic in the hydrology of a wetland (Brinson 1993) and that the hydrology of a wetland is the single most important driver of the maintenance of the types of and processes performed by wetlands in a landscape (Mitsch and Gosselink 2000). Hydrology influences the biogeochemical, biodiversity, and flood retention functions that a wetland performs (Brinson 1993). However, wetland hydrology is poorly characterized in general (National Research Council 1995) and particularly in headwater systems and in terms of dynamics and response to climatic drivers (Cole et al. 1997, Cole and Brooks 2000). Further understanding and characterizing the landscape setting or 'landscape template' (Bedford 1996) of a wetland is critically important to understanding wetland dynamics and for planning protection, restoration, and mitigation efforts. Characterizing

the landscape setting of a wetland can be used to infer the water source to a wetland and, in turn, its resilience to changes in hydrology driven by climate change (Winter 2000).

So, there is a general need to better understand the hydrologic setting and dynamics of headwater wetlands. Further, there are opportunities to draw upon and inform the stream ecology literature by looking at wetlands in a 4-dimensional, dynamic, River Ecosystem Synthesis approach.

SCOPE OF WORK

RESEARCH OBJECTIVES AND QUESTIONS

The general goals of this dissertation were to better understand the hydrologic setting and the seasonal dynamics of the hydrologic regime for headwater wetlands in the Susquehanna River Basin. The objectives were achieved through a synthesis at the basin scale and detailed analyses in a single sub-watershed and for wetlands that have long-term water level records as outlined below. The research questions for each chapter are presented in the Table 1.

More specifically, the goal of Chapter 2 was to articulate an ecologically-relevant spatial hierarchy for the study of wetland ecosystem services in the Susquehanna River Basin. Ecosystem services represent the benefits that are derived by humans from the functions or processes performed by an ecosystem (Costanza et al. 1997). For example, the long- and short-term storage of water in a wetland provides flood protection for individuals living downstream (Brinson 1993). Within a River Ecosystem Synthesis Framework, Chapter 2 looked to identify a “nested, discontinuous hierarchies of patch

mosaics" (Thorp et al. 2006) suitable for the study of wetland ecosystem services at a basin-scale based on longitudinal and lateral dimensions of a riverine ecosystem.

Additionally, under the River Ecosystem Synthesis Framework (Thorp et al. 2006), this research also sought to help to identify and understand headwater riparian wetlands as critical Functional Process Zones through modeling and mapping longitudinal and lateral reach-level hydrogeomorphic factors associated with riparian wetland occurrence (Chapter 3). Analytically, this research used techniques primarily drawn from stream ecology and fluvial geomorphology for unconstrained reaches where riparian wetlands occur.

Finally, this research sought to explore the temporal and vertical dimensions of headwater riparian wetlands by characterizing hydrologic regimes of water depths among different wetland types over time (Chapter 4). A range of time series analyses were used to achieve the objective of exploring the dynamics in water level fluctuations within headwater riparian wetlands.

Table 1. Objectives and research questions for the dissertation chapters.

Chapter	Objective	Research Question
2	Articulate an ecologically-relevant spatial hierarchy for the study of wetland services in the Susquehanna River Basin	Does the existing hierarchy allow for integration across multi-disciplinary efforts and data sources?
		Are there spatial scales of study that will facilitate 'scaling up' of ecosystem studies to a small-watershed or basin-wide scale?
		Can process-based tools from geomorphology help to quantify size class breaks in contributing area to the study ecosytems?
3	Identify and map reach-level characteristics associated with riparian wetland occurrence	How accurately do modeled geomorphic variables derived from digital elevation models predict quantitative and qualitative reach-level variables?
		Can these topographic characteristics be mapped to help identify 'wetland supportive environments' or unconstrained reaches?
		Can known occurrences of wetlands be predicted from reach-level topographic variables?
4	Explore the temporal dynamics of the water levels in headwater riparian wetlands	How well can changes in water level in head water wetlands be predicted by off-site precipitation data?
		Does the timing of the summer draw down differ between wetlands of different HGM classes and how does this vary between years of extreme wetness and dryness?

OUTLINE OF CHAPTERS

The chapters of this dissertation are situated within the four-dimensional framework of the dynamics of riverine systems (Figure 3) (Ward 1989). Chapter 2 is integrative and looks to lay out an appropriate spatio-temporal scale for studying wetlands in the Susquehanna River Basin taking into account the four-dimensional nature of the system. Chapter 3 focuses primarily on the latitudinal connection between the stream and floodplain looking to identify where the reaches are sufficiently unconstrained to support riparian wetlands. Further, once these areas are indentified they were used to

map the longitudinal distribution of wetland-supportive reaches in the watershed. Chapter 4 focuses on the vertical and temporal dynamics of the riverine system by analyzing well data for dynamics in water level fluctuations. Finally, Chapter 5 summarizes this work, putting the findings into perspective for management and future research.

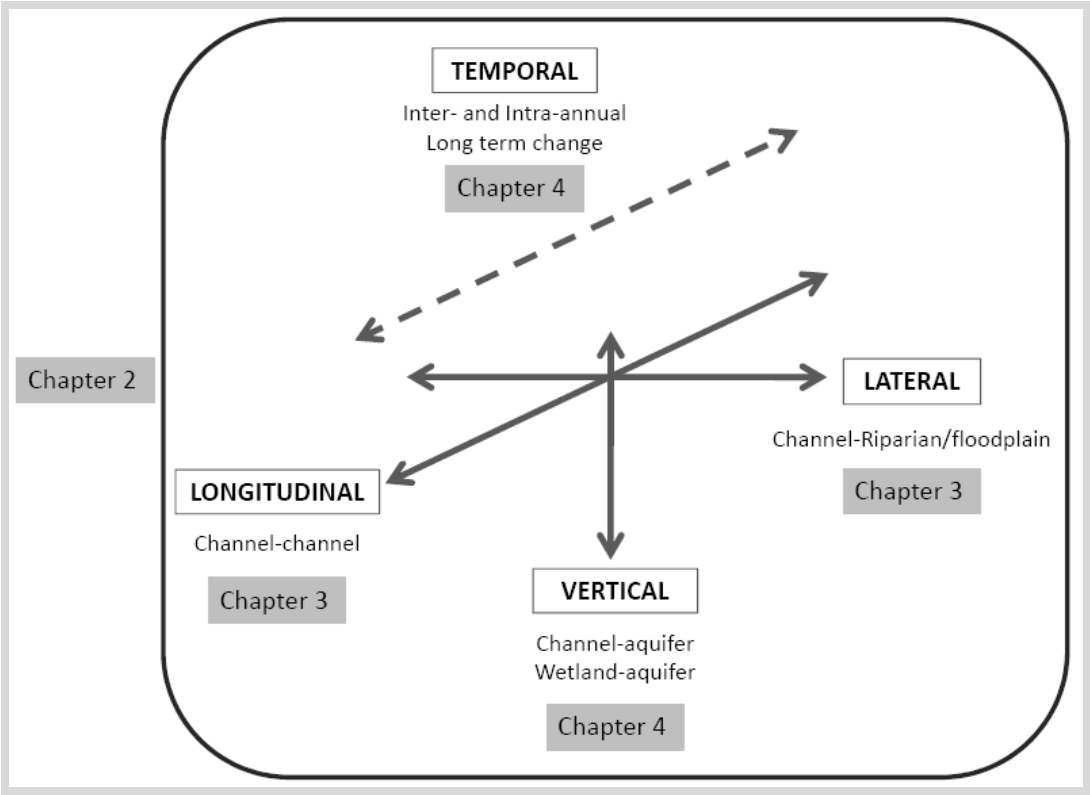

Figure 3. Situation of the dissertation chapters in the conceptual framework of the 4-dimensional nature of riverine systems (Sensu Ward 1989).

CONTRIBUTIONS

This work contributes to the literature on: the landscape ecology of headwater riparian systems, the integration between fluvial geomorphology, stream ecology, and wetland studies; fundamental wetland hydrology; the resilience of ecosystems; and the management of integrated stream-wetland-riparian headwater systems.

As stated above, the literature on the landscape ecology of riparian systems draws on fluvial geomorphology and landscape ecology. Specifically, landscape ecology is used in the study of the distribution of ecosystems as patches in a landscape, the flux of biota and materials between these patches, and the general focus on pattern and process in ecosystem studies. The work in this dissertation contributes to this literature on the fundamental level of helping to understand the "where" of headwater riparian wetlands in order to then allow for the study of them as functional patches providing ecosystem services within a landscape. Additionally, this work contributes to the sparse literature that specifically focuses on landscape ecology of headwater systems.

The characterization of landscape settings for headwater wetlands in this dissertation is improved by incorporating aspects of fluvial geomorphology, specifically the longitudinal and latitudinal directions of the River Ecosystem Synthesis Framework, in understanding of the occurrence of riverine wetland systems. This attempt at integration helps to bridge some of the disciplinary gap between stream and wetland ecology and will contribute to that gap in the literature.

The work of this dissertation, both in the understanding and characterization of the hydrologic setting (longitudinal and lateral) and hydrologic dynamics (temporal and vertical) of headwater riparian systems contributes to the literature of wetland hydrology. There is a noted lack in understanding of both the hydrologic setting and regimes of headwater wetlands, and this study draws both a data-rich watershed and several data-rich wetlands to further the hydrologic understanding of these systems.

Additionally, this work contributes to the understanding of the bio-physical aspects of ecosystem resilience to perturbation by exploring the relationships between

wetland water level dynamics and the drivers of anthropogenic and climate changes, such as land use and precipitation patterns.

Finally, beyond the contribution to the academic and intellectual literature, this study contributes as applied research to the study and management of headwater riparian systems. Specifically, the spatially-explicit hierarchy will be helpful for organizing monitoring and assessment efforts. Further, adding to the knowledge base about where riparian wetlands are likely to occur can help in the siting of hydrologically appropriate restoration and mitigation efforts. And, maybe most importantly, currently wetlands and streams are primarily studied and managed separately, though there is growing understanding that in many regards riparian wetlands and streams are functionally and hydrologically one integrated system. The techniques, approaches, and findings of this research can help to nudge the management of these systems towards an integrated River Ecosystem Synthesis Framework.

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Chapter 2

HIERARCHICAL LANDSCAPE SETTING OF HEADWATER FLOODPLAIN WETLANDS OF THE SUSQUEHANNA RIVER BASIN

ABSTRACT

There is a growing recognition in aquatic ecology and landscape ecology that riparian systems do not solely respond to simple downstream gradients, but instead are driven by complex interactions between their landscape settings at multiple scales throughout the drainage network. Identifying an appropriate hierarchical landscape classification can aid in integrating across disciplines and scaling up findings from plot-level studies. Understanding the distribution of ecosystems can aid in understanding how these systems function and provide services across multiple scales and maintain resilience in the face of human-induced stressors. This chapter is a synthesis of the efforts of the ecological portion of a study of climate change and ecosystem services provided by freshwater wetlands of the Susquehanna River Basin to characterize and map a hierarchical landscape classification for use in the study. Specifically, the study sought to: identify an ecologically relevant over-arching, hierarchical landscape framework; refine or develop ecologically relevant classifications within the levels of the framework; and map or predict distributions of the elements of the classification where possible. Several existing classification schemes were explored to propose the following overarching hierarchical landscape classification: Basin, Physiographic Province, Sub-watershed, Channel Reach, and Habitat. Within the watershed level, relevant size classes were generated based on geomorphic relationship between stream slope and contributing area. In the reach level, a method was developed to identify constrained versus unconstrained

reaches relative to headwater riparian occurrence. Additionally, a conceptual model of reach types was developed that include: stream-, floodplain-, wetland-dominated or mixed. Some aspects of this work is ongoing, but this chapter presents the over-arching framework and the work done within the scales to date.

INTRODUCTION

There is a growing recognition in aquatic and landscape ecology that riparian systems do not solely respond to simple downstream gradients, but instead are driven by complex interactions between their landscape settings at multiple scales throughout the drainage network (Flores et al. 2006). Riparian ecosystems are positioned between and are part of both terrestrial and aquatic ecosystems and their hydrologic regime and disturbance regimes are driven by dynamics of the hillslope and larger groundwater hydrology, as well as the fluvial system. In turn, when riparian ecosystems (wetlands, floodplains, and streams) have a high degree of connectivity, they act as one integrated riparian system (Naiman et al. 2005, Brooks et al. 2009). However, there is little communication between the sub-disciplines of stream ecology and wetland ecology and these ecosystems are generally studied as separate entities.

Further, regional, multidisciplinary studies of riparian ecosystems look at many processes and patterns at various spatial and temporal scales due to the varying scales upon which these processes act. Also, there is the practical factor of investigators needing to choose a resolution that delivers appropriate data for each analysis (Benda et al. 2002). An appropriate hierarchical landscape classification can aid in integrating

across disciplines and spatially scaling up findings from plot-level studies (Frissel et al. 1986).

Thorp et al. (2006) laid out a framework for studying and understanding biocomplexity of riverine systems as the River Ecosystem Synthesis Framework. They defined biocomplexity as “structural and functional phenomena arising from dynamic interactions amongst biotic elements of the ecosystem and between these components and the physiochemical environment” (Thorp et al. 2006: 124). This work built upon the view of river network as a primarily unidirectional gradient from headwater to mouth as presented in the River Continuum Concept (Vannote et al. 1980) and incorporated the importance of the lateral connection of the stream with the flood plain (Junk et al. 1989). Further, they proposed a combined approach to the understanding of riverine systems by combining the ecological aspects of fluvial geomorphology with the patch dynamics models used in terrestrial landscape studies. They conceptualize a riverine system as a dynamic shifting mosaic (Stanford et al. 2005) of geomorphic patches or Functional Process Zones. They state that linking the appropriate spatio-temporal scales with ecologically-relevant processes and identifying spatiotemporal patterns and hierarchies are important avenues for research in riverine systems.

RESILIENCE AND LANDSCAPE DIVERSITY

Understanding the distribution of ecosystems across multiple scales can help investigators to characterize their resilience to disturbance. The earth’s ecosystems are increasingly dominated by the direct and indirect actions of humans (Vitousek et al. 1997), while in turn, there is a growing understanding of the dependence of human

systems on ecosystems to provide critical services and functions (Costanza et al. 1997). In light of this improved understanding, that humans and ecosystems are highly intertwined, there is a need to try and understand what the effects of human actions are on ecosystems. Can management decisions be made regarding human use of the environment that reduce impacts to ecosystems? Are there certain thresholds of disturbance, above which ecosystems no longer perform functions necessary to support human systems? As a key strategy to maintaining ecosystem functions and services in the face of multiple stressors, it is important to understand and manage for the maintenance of the resilience of ecosystems (Fischlin et al. 2007). Ecological resilience is the magnitude of stress or perturbation that a system can withstand before it changes into an alternative stable state (Gunderson 2000a). At this "flip-in-state" there is often a corresponding change in the processes that structure the system.

Observations of natural, disturbed, and managed systems indicate that change in the systems is not always gradual: there are both long-term processes and episodic events that shape ecosystems; rare and catastrophic events can shape a system; critical processes function at different rates; and many aspects of ecosystems are not continuous but instead are "lumpy" (discontinuous) in space and time (Holling and Gunderson 2002).

Further, ecosystem complexity both within and across scales is a major factor in maintaining ecosystem resiliency. Complexity with functional redundancy allows for functional diversity where multiple species can perform the same functions in an ecosystem or multiple ecosystems perform the same functions in a landscape (Folke et al. 2004). And, the resilience of a landscape is not necessarily dependent on one element, but instead "by the functions those elements provide, and their distribution within and

across scales with systems with a diversity of function within scales and redundancy across scales are the most resilient" (Allen et al. 2005). Understanding the distribution of ecosystems can aid in understanding how these systems function and provide services across multiple scales (Thorp et al. 2008).

CLIMATE CHANGE

In an attempt to further explore these questions, a larger effort to is underway to understand the potential linear and non-linear responses of the ecological functions and services of freshwater wetlands in the Susquehanna River Basin to climate change. This study is looking at the interaction of climate change and land use change on freshwater wetlands, using hydrology as the main driver of change. The goal is to build conceptual and empirical relationships between measured and modeled wetland hydrology and the ecological services they provide, identify thresholds in those relationships, and then use those relationships and break points to forecast the provision of services under climate and land use change scenarios.

SCOPE OF THIS CHAPTER

This chapter is a synthesis of the efforts of the ecological portion of the climate change and ecosystem services study to characterize and map a hierarchical landscape classification for use in the study. Specifically, this chapter sought to:

- Identify an ecologically relevant over-arching, hierarchical landscape framework;
- Refine or develop ecologically relevant classifications within the levels of the framework; and
- Map the elements of the classification where possible.

More specifically, this research sought to address a series of related research questions:

- Does the existing hierarchy allow for integration across multi-disciplinary efforts and data sources?
- Are there spatial scales of study that will facilitate 'scaling up' of ecosystem studies to a small-watershed or basin-wide scale?
- Can process-based tools from geomorphology help to quantify size class breaks in contributing area to the study ecosytems?

Some aspects of this work is ongoing, but this chapter presents the over-arching framework and the work done within the scales to date.

APPROACH AND RESULTS

STUDY AREA

The study area for ecosystem services project is the Susquehanna River Basin in the Mid-Atlantic Region of the United States, which is the largest tributary to the Chesapeake Bay (Figure 1) (SRBC, 1998). The basin crosses three states (New York, Pennsylvania, and Maryland) and is approximately 70,000 square kilometers. The basin also falls within several physiographic provinces predominantly within the Ridge and Valley, the glaciated- and unglaciated-Appalachian Plateau, and the Piedmont. Wetlands comprise a relatively small percentage of the basin by area, but provide services disproportionate to their extent in the landscape (Wardrop et al. 2007).

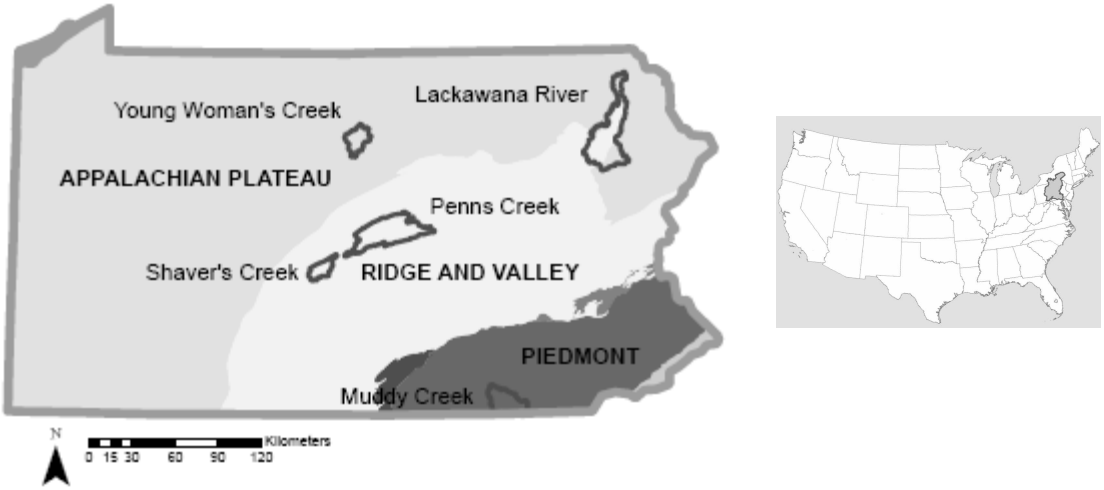


Figure 1. Susquehanna River Basin location map and physiographic provinces, showing intensively studies watersheds. Inset shows the boundaries of the Susquehanna River Basin.

HEADWATER FLOODPLAIN ECOSYSTEMS

The wetlands focused on for this study are predominantly headwater floodplain wetlands with some riparian depression and toe of slope wetlands that occur within the floodplain of a headwater stream (Cole et al. 2008). Notably, this study did not look at open water, isolated, fringing, or stratigraphic slope wetlands (Cole et al. 1997, Cole et al. 2008).

As previously mentioned, hydrological complexity (and connectivity) is not distributed evenly or randomly within a stream network. The study focused primarily on headwater riparian systems (roughly 0 – 3rd order systems) in part because they constitute the vast majority of the stream network in the Susquehanna Basin. Further, wetlands associated with these streams make up the majority of the resource, usually > 90%, and

perform a range of functions (Wardrop et al. 2007). Additionally, headwater floodplains are not as well studied as larger floodplain systems.

OVERVIEW OF HIERARCHICAL LANDSCAPE CLASSIFICATION

The proposed classification built upon several existing classification schemes to propose the following overarching hierarchical landscape classification: Basin, Physiographic Province, Sub-watershed, Channel Reach, and Macro- and Micro-habitat (Thorp et al. 2008, Naiman et al. 2005, Montgomery 1999, Petts and Amoros 1996, Frissell et al. 1986). The basin scale was fixed by the scope of work of the project. Below is a description of the efforts to identify ecologically relevant characterizations and classifications within each level of the landscape classification and to describe mapping and prediction efforts where applicable.

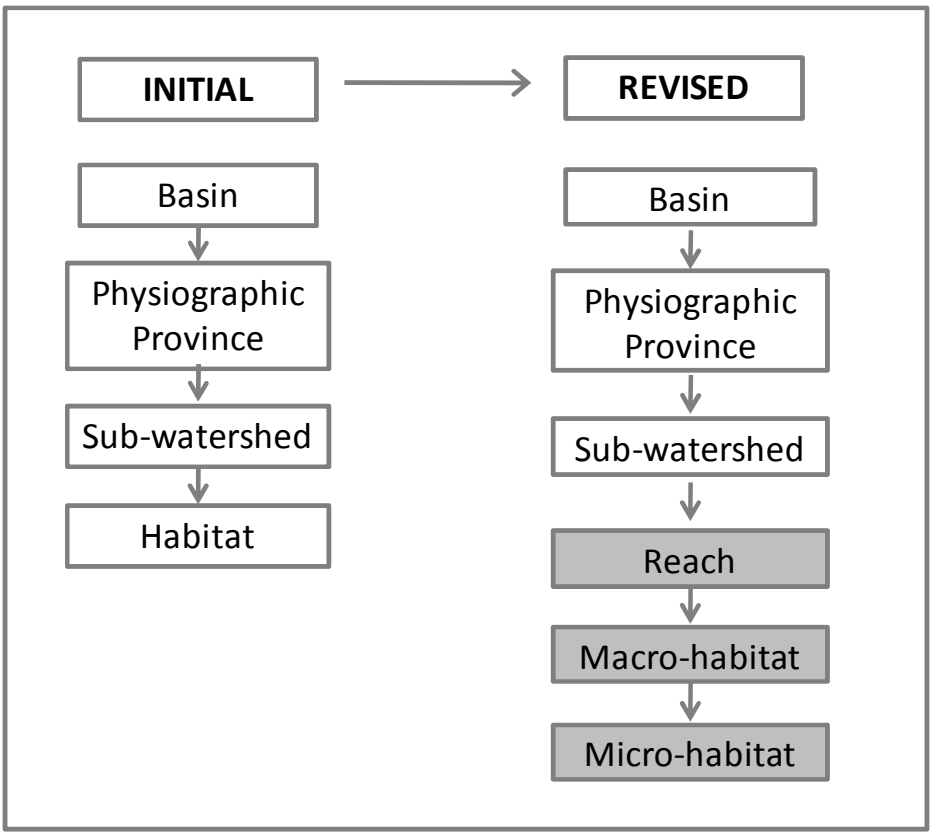


Figure 2. Initial and revised hierarchy of landscape settings for describing ecologically-relevant components for headwater ecosystems.

Physiographic Province

There are several ecoregional and physiographic landscape classifications (e.g., Omernick 1987, Bailey 1996). Physiographic Provinces are based on primarily on landform and geology. Unlike other landscape classification, Physiographic Provinces do not incorporate land use, which is critically important for this study as land use change is one of the dominant drivers of interest. Land use, characterized as "reference domains", has been used successfully to stratify regional riparian studies (Wardrop et al. 2005, Brooks et al. 2009). Additionally, the conceptual models developed as the framework for

the integrated hydrologic model used in the study were developed by Physiographic Province and Section (Duffy and Evans 2000).

The dominant physiographic provinces in the basin are: the Ridge and Valley Physiographic Province, the Appalachian Plateau, and the Piedmont. These regions vary in their topography, geology, soils, climate, and land use patterns (Wardop et al. 2005). The study stratified the sub-watersheds by Physiographic Province. Though major wetland types occur in all provinces across the region (Cole et al. 2008), the hydrology and patterns of human disturbance vary greatly within the Mid-Atlantic Region, primarily constrained by physiographic difference and historically-based patterns of land use (Griscom et al. 2007).

Watershed

Within the small watershed scale, the characteristics of floodplain ecosystems change with the size of the stream in which the riparian area is associated. For example, a headwater stream may be more likely to be intermittent or have lower stream power than a larger order stream. In turn, wetlands, when combined with their associated riparian components, can be categorized as headwater systems or complexes (Brooks et al. 2004). Based on previous studies and field experience, there were at least three classes of floodplain systems associated with: small headwater, moderate headwater, and mainstem streams. Extreme headwaters fall outside of these classes as they are very small, often ephemeral streams that rarely have associated floodplain habitat. Small and moderate headwater systems have discontinuous floodplains, whereas the mainstem systems can have continuous floodplains traversing many kilometers. However, identifying

appropriate break points in contributing area between these classes can be observationally and analytically difficult.

To explore potential area size class breaks, geomorphic relationships in the study watersheds were evaluated. The watersheds with more intensive investigations were five sub-basins within the Susquehanna basin: Shaver's Creek, Muddy Creek, Young Woman's Creek, Penns Creek, and Lackawana Creek (Figure 2). These watersheds were selected based on a number of criteria including: dominant study area Physiographic Province, mix of land uses represented between watersheds, and availability of existing hydrologic data (wetland water levels and stream gage data). All of the study watersheds are true headwater watersheds, though in the Susquehanna there are many pass-through watersheds with headwater streams draining directly into larger mainstem reaches (Hughes et al. 1981). All except for Penns Creek, the study watersheds correspond with USGS HUC 11 designations; Penns Creek is combined with Elk Creek in the headwaters and was truncated at the stream gage.

Synthetic streams were generated for each watershed from 10-m digital elevation models (DEMs) (http://seamless.usgs.gov/products/3arc.php) (Gesch 2007). Plots of channel density (m/m^2) against contributing area (m^2) were generated for each watershed based on the synthetic streams, and inflection points were identified to use as channel initiation thresholds (Miller 2003, Clarke et al. 2008). These were then revised based on plotting the synthetic streams on top of a hillshade and looking for feathering, where stream channels form in parallel lines in the landscape because they are outside of an actual topographic channel. The channel initiation thresholds identified through these two approaches resulted in a range of thresholds from 0.02-0.1 km^2. So, I assumed that

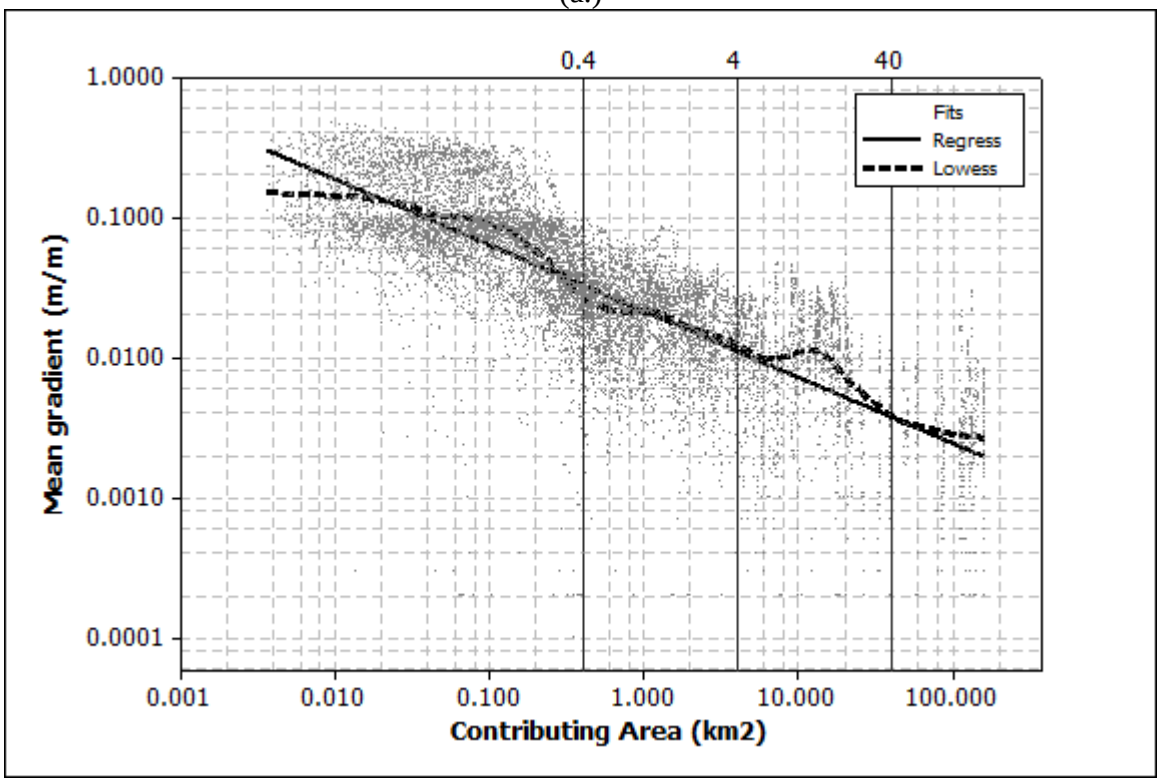
synthetic channels with a contributing area less than these thresholds was not an actual channel, but more likely a small swale. Further, this range of values constituted the smallest contributing area of the extreme headwater class.

To explore other potential area size class breaks, the GIS-generated slopes of stream segments were plotted against the contributing area for the reach on log-log axes. Identifying inflections in this distribution has been proposed as a way to look for shifts in the process domains that dominate the stream at differing stream sizes (Ijjasz-Vasquez and Bras 1995, Montgomery et al. 1996, and Tucker and Bras 1998).

The slope-area plots were generated for all of five of the study watersheds, showing similar inflection points. Shown are the data for the stream reaches of Shaver's Creek, which significantly fit ($p<0.001$) a negative sloping trend ($R^2=0.62$), with segments with larger contributing areas having lower slopes (Figure 3). A local regression smoother was also fitted to the data to show more localized trends in the data (degree of smoothing= 0.1, steps=2) with the degree of smoothing indicating the proportion of the data around any given point that was fitted with the regression and the 2 steps were used to reduce the influence of outliers. The most distinct shift in the data is seen around 0.4 km^2 where there is a marked decrease in average channel slope. Previous studies suggest that this first shift (farthest upstream) indicates a shift from hillslope to fluvial processes, and is fairly close to the values found in a California study (0.06 and 0.07 km^2) (Whipple and Tucker 1999). The next notable shift appears to occur around 4 km^2 with another drop in average slope, and finally a break around 40 km^2. The final split has much less data, because there are fewer stream segments with larger contributing areas in the watershed.

These groupings (Group 1: <0.4 km^2, Group 2: 0.4 to <4 km^2, Group 3: 4 to <40 km^2, and Group 4: > or = 40km^2) show decreasing mean stream gradient and decreasing proportion of the watershed with approximately half of the stream length in the extreme headwater class (Group 1) (Figure 4).

(a.)



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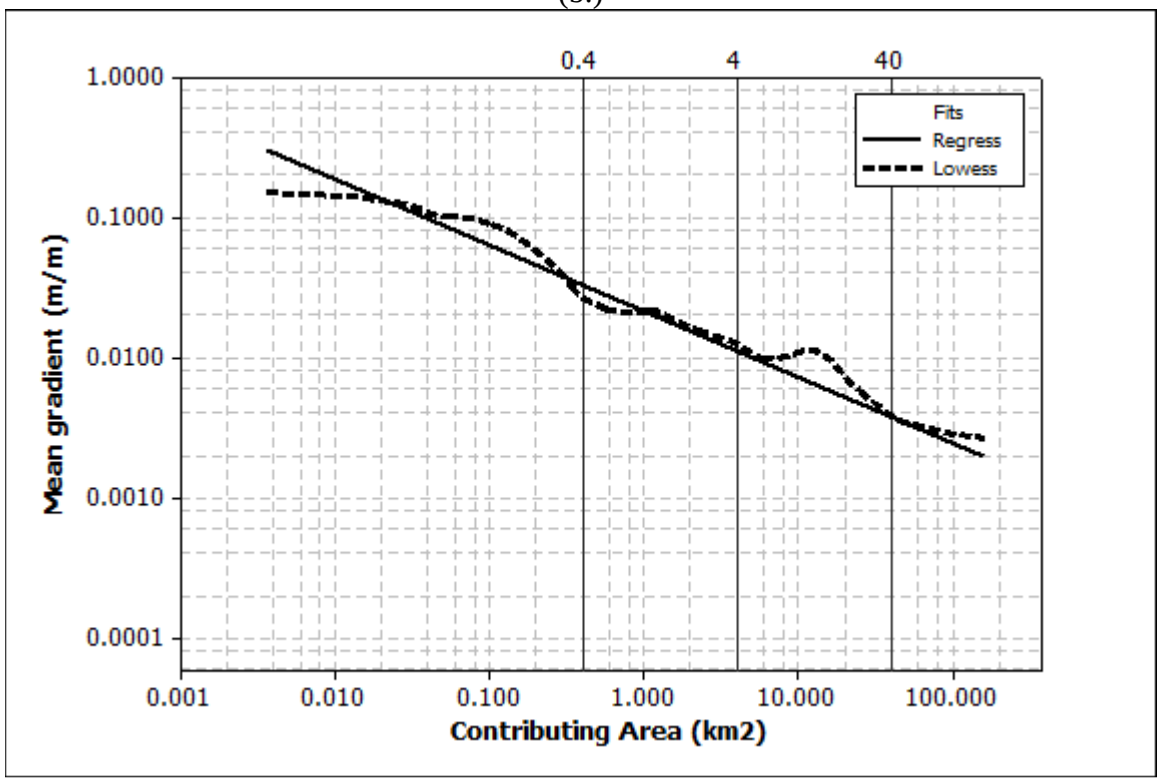


Figure 3. The contributing area versus the mean stream gradient for all reaches in the Shaver's Creek watershed, Pennsylvania, plotted on log-log scale and fits of linear regression and Lowess smoothers with (a) and without (b) the data points for each reach.

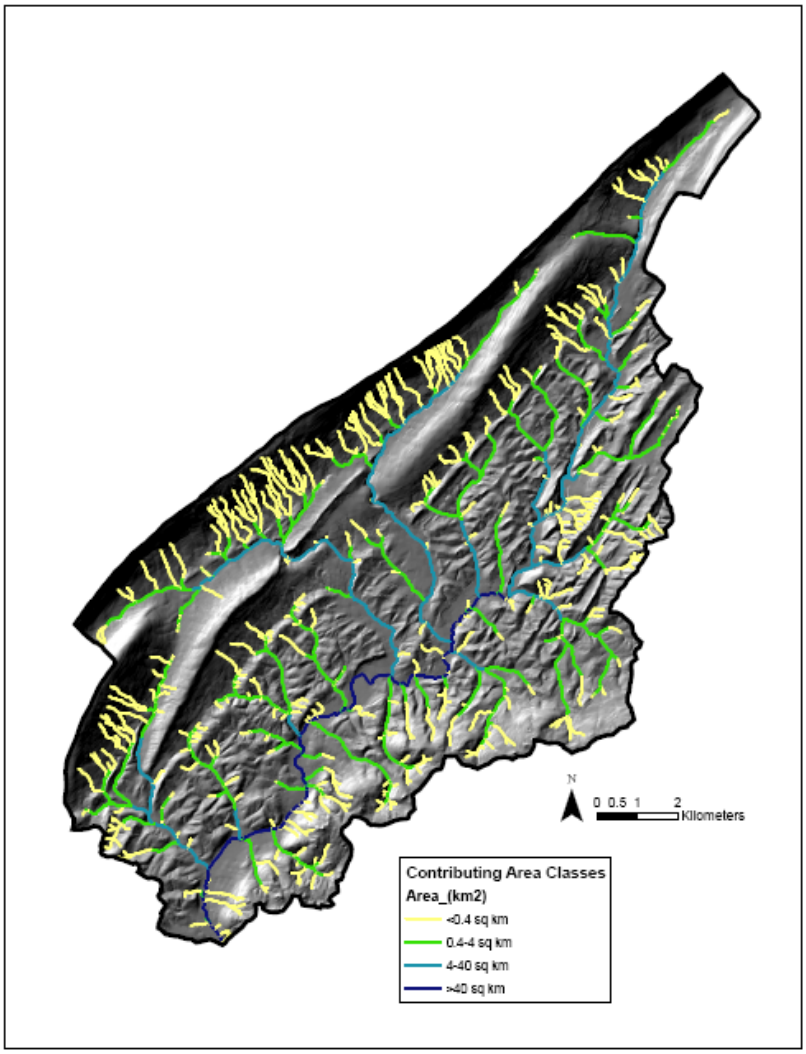


Figure 4. Map of Shaver's Creek watershed, Pennsylvania, showing synthetic stream locations for four contributing area size classes.

Reach Scale

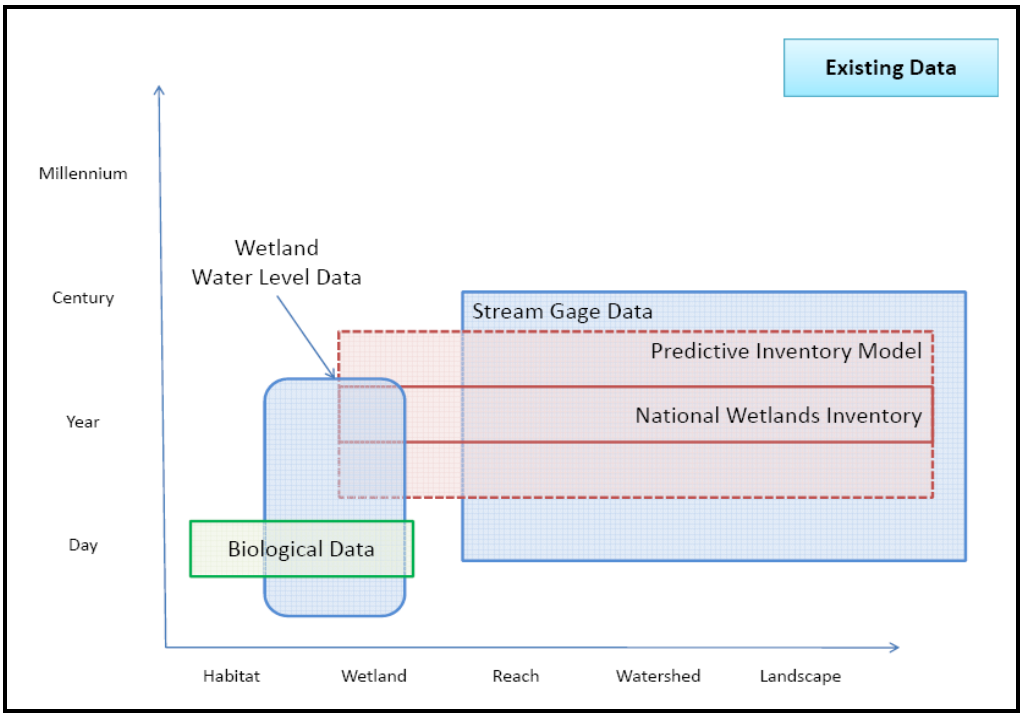
It has been suggested that "in every respect the valley rules the stream" through controlling the energy and its magnitude associated with overbank flows and the delivery of materials (Hynes 1975). Reaches that have floodplain wetland habitats typically are unconstrained latitudinal and longitudinally for the reasons Hynes lays out and because they are often locations for regional, toe-of-slope ground water discharges (Winter 2000, Winter 2001).

The reach scale was identified as an important scale of analysis when the study design was reevaluated following a process of diagramming disciplinary knowledge and datasets (Benda at al. 2002). Datasets and knowledge structures were diagrammed for the disciplines involved in the study (hydrology, ecology, meteorology, and sociology) to explicitly express their spatial and temporal resolution. The existing data relating to the ecological and hydrological portions of the study (Figure 5.a) showed a spatial gap in the data between the *in situ* wetland ecological data collected at the wetland and habitat levels (site level and plot level) and the hydrologic model data. The hydrologic model is a highly scalable, physics-based, coupled groundwater, surface water and soil water model (Qu and Duffy 2006). This model can scale down to an individual wetland, but this requires increasing the number of elements of model within and surrounding a wetland, and the overall number of elements of the model is limiting due to the large number of calculations per element (Figure 5.b). Therefore, when running a model on a watershed basis, the domain of the model cannot be decomposed to capture all of the wetlands in the watershed. So, as a matter of practicality, except for a few case studies, the model provides output at a larger spatial scale than at an individual wetland.

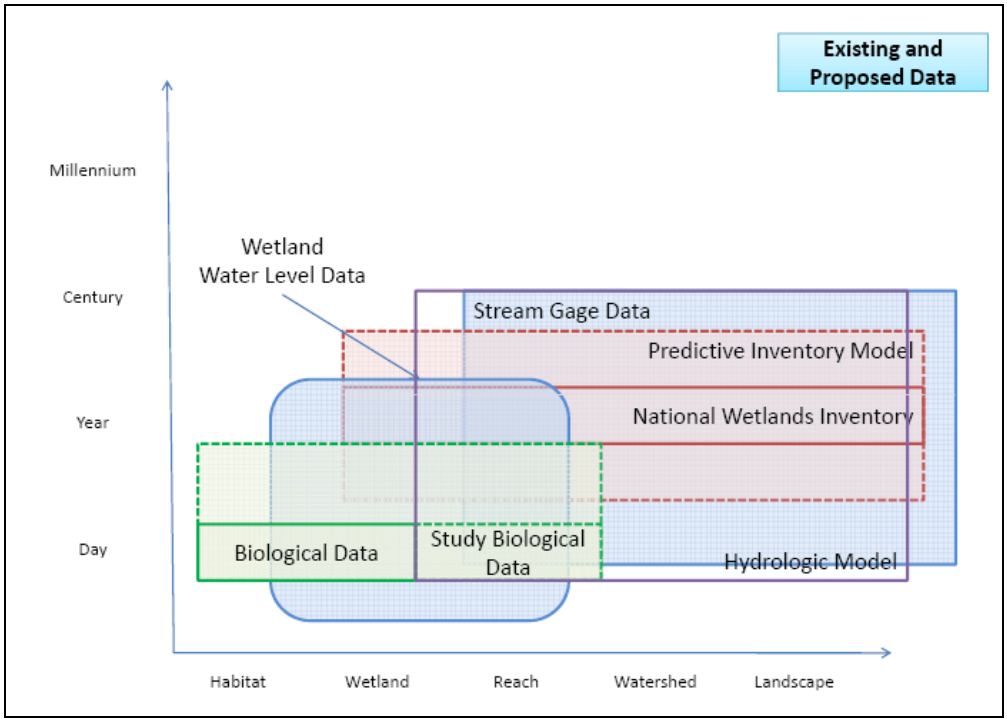
Unconstrained reaches allow for hydrologic complexity in that the sources of water to the reach come from three directions: groundwater, surface water, and overbank flooding (Stanford et al. 2005). Unconstrained reaches can be areas of periodic deposition as flows slow when they change gradient and, in turn, the power of the stream decreases. Additionally, where there are unconstrained reaches, the fluvial processes driving the structure of the reach change. The processes change due to debris flow, scour, and some flooding to a reach driven by channel migration, avulsion, and flooding (Montgomery

1999). The types of habitat change with allowing for off-channel habitats and a higher likelihood of hyporheic habitat in unconstrained reaches.

Thus, the most appropriate reach type for characterizing hydrological complexity was considered to be the partially confined and unconfined channel reaches in a watershed that contain stream-wetland complexes. These reach types are typically arrayed along a hydrogeomorphic gradient of wetland dominated to floodplain dominated types, depending on the relative contributions of groundwater and surface water sources.



(a)



(b)

Figure 5. Spatial and temporal scales associated with (a) existing and (b) existing and proposed data for the study.

Unconfined reaches were identified through topographic analysis of the sub-watersheds. Using known locations of confined and unconfined reaches (in this case those that are sufficiently unconfined to have a wetland present), a predictive model was generated that put the reaches into classes based on topographic variables (valley width, longitudinal stream gradient, and contributing area) (see Chapter 3). Then maps were generated of the distribution of the reach classes based on the degree of confinement (Figure 6).

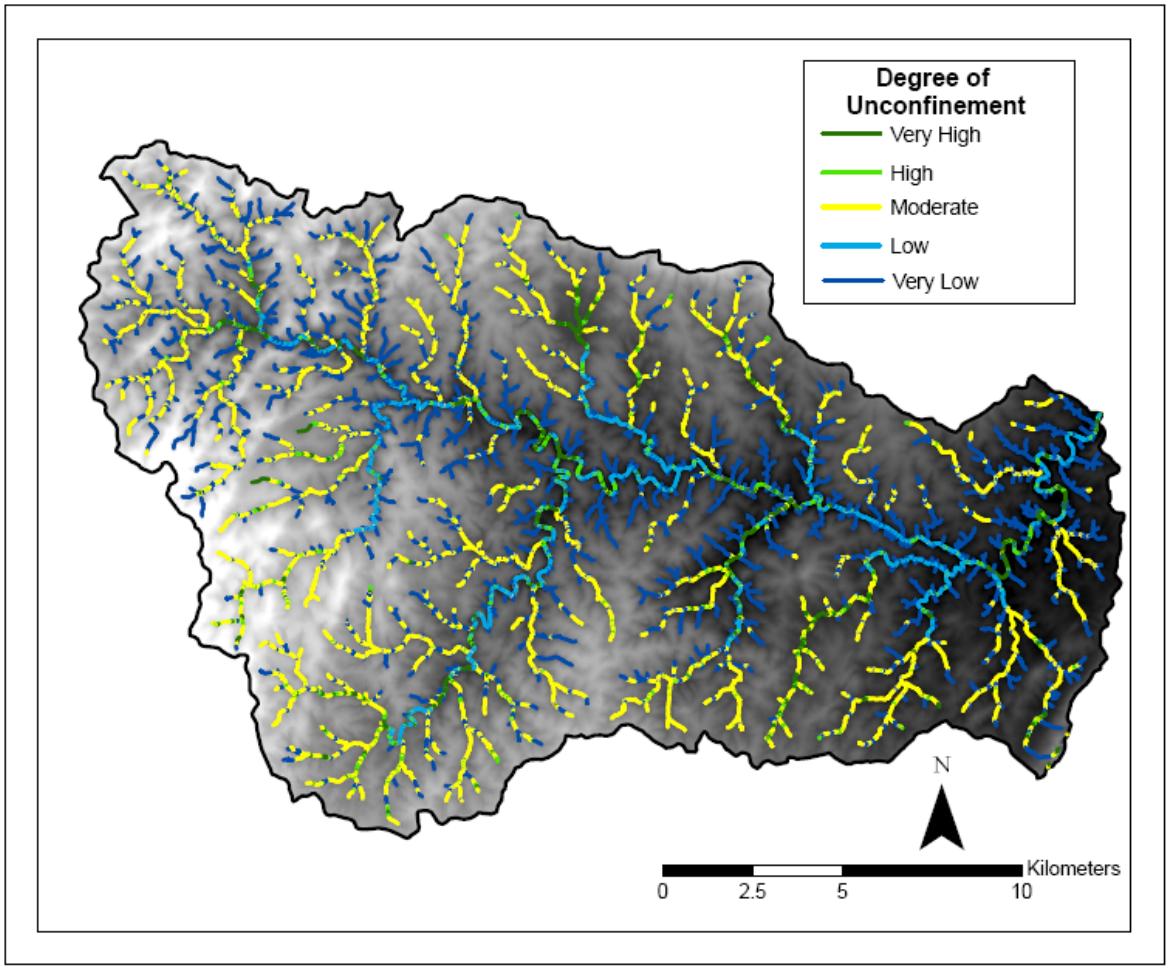


Figure 6. Stream reaches mapped by classes of the degree of unconfinement (very high is the most unconfined and very low is the most confined) for Muddy Creek watershed, Pennsylvania.

Habitat

The floodplain habitats occur at both macro- and micro-scales. The macro-habitat scale can be simply seen as the coarse scale range of different ecosystem types: stream, floodplain, and wetland. Micro-habitats are the sub-habitats within each of these ecosystem types.

Reach Types and Macro Habitat: Reaches have a range of associated macro-habitats: stream, wetland, and floodplain. One end of the spectrum are stream-dominated reaches. These primarily only have stream habitat including pools, riffles, and runs. There is usually very little flow-pulse floodplain habitat or wetland habitat. There may be small seeps adjacent to the stream formed from topographic breaks in slope or bedrock contacts or fissures.

Wetland-dominated reaches have wetlands in the majority of the floodplain. This arrangement includes having wetlands up to the edge of the stream. These are often very wet or saturated headwater floodplain wetlands. Typically, the stream habitat associated with these reaches is relatively slow moving, often tannic water, flowing through organic soils. These reaches may be associated with old beaver activity.

Floodplain-dominated reaches have little or no wetland-associated with the reach. However, there are extensive floodplain habitats including a range of side channels (parapotomal and pleisiapotamal) indicating fairly frequent flow pulses. Floodplain habitat in this study is habitat associated with the flow pulse or within the bankfull width flow events.

Mixed reaches show have a mixture of wetland and floodplain habitats within the same reach. The wetlands may or may not be adjacent to the stream. They are often along

the toe of slope and do not extend to the edge of the stream. These types of reaches seem to occur where the stream valley is sufficiently wide, but is slightly more vertically steep (or vertically constrained) than the wetland dominated reaches.

Four-dimensional hydrological connectivity at the reach scale determines the proportions of wetland types within a riverine system and the degree of interaction between them. Stream-wetland complexes occur most often in partially confined and unconfined channel reaches. Their differing hydrogeomorphic characteristics result in a hierarchical patchwork of process zones, each with distinct habitats and biological communities. By evaluating the biology, habitat, and hydrological character of individual units within a reach, the functions provided by each element can be determined and potentially scaled up to determine their distributions and redundancy across scales. The primary, and often covarying, drivers of the different reach types are:

- degree of lateral confinement of the stream valley,
- degree of longitudinal confinement of the stream valley,
- amount and timing of groundwater discharge, and
- amount of stream power experienced in the system.

Micro Habitat: The complexity of micro-habitat within the riparian area ecosystems (floodplains, streams, and wetlands) is driven by the flow regime of the stream, the discharge of groundwater, and the occurrence of structuring elements (such as downed trees or boulders) (Newson and Newson 2001, Stanford et al. 2005). Because of the dynamic nature of these systems, the individual elements may not be stable in space and time. For example, a side channel may dry down in the dry season and a debris jam may be moved through the system by a large flow event.

A processed-based classification scheme was developed to use in the inventory of the micro-habitats associated with riparian ecosystems (Figure 7). The classification

attempts to capture the diversity of micro-habitats in the stream and floodplain (including wetlands), which include elements that are influenced by the flow pulse (or within bank events) and the flood pulse (over bank events). These flows occur at different magnitude and frequencies, and the element occurrences of these certain associated habitats can act as indicators of past flow events.

These macro- and micro-habitats perform a range of functions within the riparian area, and when mapped either through prediction or field identification, they can help to understand the distribution of floodplain services provided in a region.

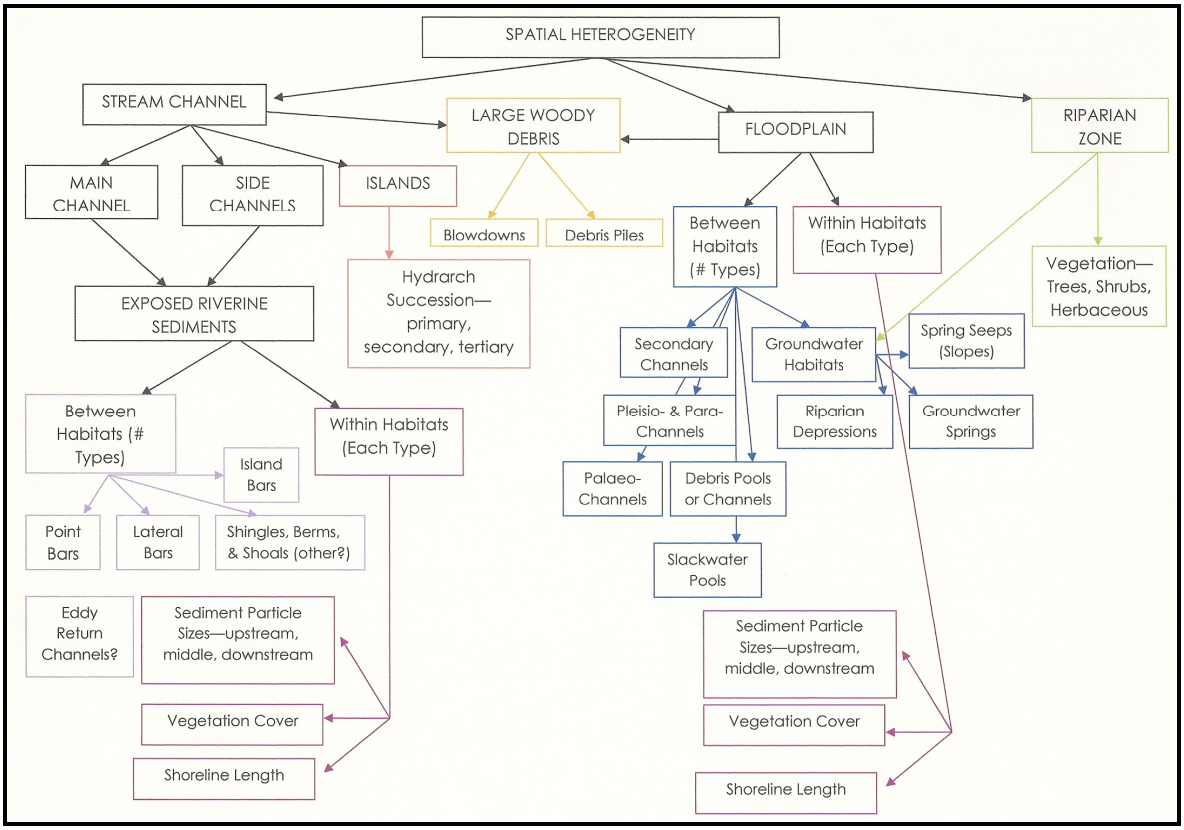


Figure 7. Process-based classification for floodplain micro-habitats (modified from S. Yetter).

DISCUSSION

This chapter synthesizes the efforts to date of the ecological portion of a climate change and ecosystem services study to identify an ecologically relevant, over-arching, hierarchical landscape framework; refine or develop ecologically relevant classifications within the levels of the framework; and map the elements of the classification where possible.

In Chapter 2, the overall objective was to articulate an ecologically-relevant spatial hierarchy for the study of wetland services in the Susquehanna River Basin that incorporated the four dimensions of the River Ecosystem Synthesis Framework. One of the key research questions was, would the existing, proposed hierarchy for the study allow for integration across disciplines and data sources in the study? By following the scaling exercise suggested for interdisciplinary studies of environmental issues (Benda et al. 2002), there was a gap identified in spatial scale between the hydrologic model output and the scale at which *in situ* wetland data, both hydrologic and biologic, had been and were proposed to be collected. Wetland data had been collected at the site- and plot-levels, while hydrologic output could be consistently generated in small-watersheds by stream segment- or reach-scales for this study.

In general, a tiered-approach to landscape and habitat classification for stream systems is fairly common, but fewer wetlands studies use this approach (Dahl et al. 2007). Particularly, the use of the reach-scale is not common in wetland studies, but is a very useful for mapping wetland probabilities and scaling up findings from site-level to small watershed or basin scales. Using similar spatial scales to stream studies, in this case

not only allowed us to better integrate across disciplines in this study but to improve the potential for communication of results between wetland and stream ecology.

Also in this study, there were issues with how the data could be scaled up from intensively studied sites to the small-watershed and basin-scales. One major issue in scaling up from site-level data was that there is a problem with inventory with an underrepresentation of the area of wetlands in the existing wetland inventory (Wardrop et al. 2007). This meant that using model output with the National Wetland Inventory wetlands alone would greatly under-represent findings. Also, some of the rapid assessment fieldwork for the study locations were to be generated following a stratified, random sampling approach. Using the NWI to stratify the sampling would leave out much of the wetland resource, particularly the floodplain habitats this sampling was attempting to characterize. Borrowing primarily stream ecology literature, the reach was proposed as a base unit for analysis for intensive site-level data collection as an intermediate scale between the sub-watershed and the habitat scales. Refining a hierarchical landscape setting framework has allowed us to conceptualize scaling the results to a basin-scale and to better integrate across sub-disciplines. For example, the collection of biological data occurs at the habitat- and reach-scales and will be related to hydrologic data collected or modeled at the reach scales (Figure 8).

Additionally, the habitats of the study were further classified at the macro- and micro-habitat sales. At the macro-scale the wetlands in the study were classified as mainstem or headwater floodplain wetlands using an approach from fluvial geomorphology. Traditionally, classification efforts break mainstem and headwater floodplain wetlands using Strahler stream order, with wetlands associated with 1st

through 3rd order as headwater floodplains and 4th order and larger as mainstem wetlands. However, the contributing area varies greatly within a stream order in the Susquehanna River Basin and at a landscape scale streams are not often consistently mapped at the same resolution (Clarke et al. 2008). This study borrowed the slope-area plot approach from fluvial geomorphology to attempt to form this break on process-based criteria instead of on relatively coarse predictors such as stream order. All stream reaches in the watershed were mapped as synthetic streams and slope and contributing area were calculated for each segment. The slope versus the area for each study reach was plotted and non-linearities or discontinuities were identified in the plot. These discontinuities were guides for the breaks in size classes in study area reaches based on contributing area: 0.4 to <4 km^2 are small headwaters, 4 to <40 km^2 are headwaters, and > or = 40 km^2 are mainstems. These were not only helpful in identifying the break between headwater and mainstem sites, but it allowed for the identification of an extreme headwater class (<0.4 km^2). This class of stream was unlikely to have floodplains, because they do not have sufficient stream power and in the Ridge and Valley they were typically fairly steep where there was little lateral or longitudinal connection from the stream to the riparian area. At the micro-habitat scale, a process-based classification for stream, wetland, and floodplain habitats was developed for use in a rapid assessment. Again, few studies focus on wetlands, streams, and floodplains collectively, whereas this classification allowed for the assessment across macro-habitat types within one reach. Also, the presence or absence of such habitats acts as indicators for processes that occur in that setting.

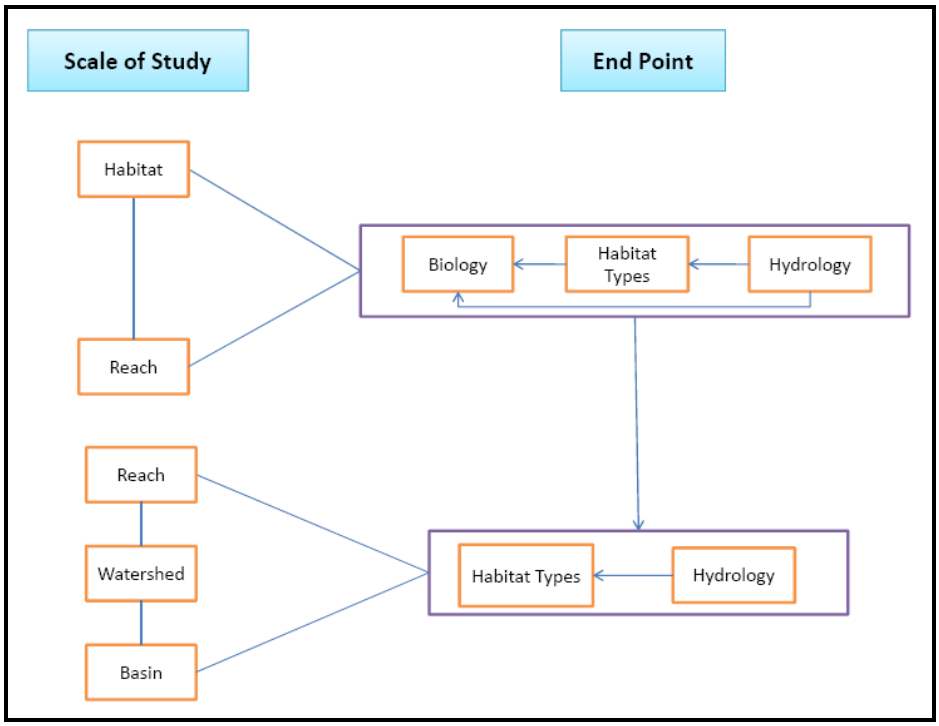


Figure 8. Spatial scale habitats compared to end points for the study of headwater ecosystems.

FUTURE DIRECTIONS

There is on-going field-level validation of the reach characterization and the occurrence and mapping of micro- and macro-habitats. These findings can be iteratively used to refine and improve the classes within the hierarchical framework. Specifically, the spatial prediction of reach types could be improved through increasing field studies in different physiographic provinces. Also the reach classification will be linked with output from an integrated hydrologic model to predict the occurrence of habitat types classified and mapped through field study.

The mapping of the hierarchical classifications could also be improved through the use of different and/or higher resolution spatial data, particularly in terms of topographic data, geologic data, and soils. LiDAR-derived DEMS are soon to be

available for the Commonwealth of Pennsylvania (Delano and Braun 2006). These data could be useful in identifying reach breaks within the riparian corridor or possibly could give more accurate valley width and valley and stream slope measurements. Geologic data is available for the state, however, fractures and fissures, one important predictor of wetland occurrence, are not consistently mapped in the state (Sloto 1994). There are also soils datasets that may be useful in stratifying the landscape for wetland studies, namely the county-level SSURGO datasets that are available for nearly all counties in Pennsylvania. Also, the Soil Climate Atlas data could be a helpful addition to characterizing the setting of a wetland as it uses historic climatic data, soil properties, and topography to model elements of the soil water balance (Waltman et al. 1997). Soils information could be used to classify the riparian soils as colluvial, alluvial, or mixed (Benda et al. 2005, Mourier et al. 2008). This study found that riparian soils material shifted from mixed alluvial and colluvial in the headwaters to more dominant fluvial soils along larger order streams, showing the shift from mixed hillslope/fluvial influence to a more fluvial influence within the system.

Potential for mapping habitats using remotely sensed data such as near infrared (IR) or fine horizontal resolution DEMs. Also, network analysis could provide a means to quantify the connectivity of floodplains ecosystems in a watershed and how they function in concert (Benda et al. 2004a). Stream studies have found that physical complexity increases at stream junctions (Benda et al. 2004b, Bigelow 2007), and it could be useful to pursue in terms of floodplain complexity.

Borrowing primarily from stream studies, this hierarchical landscape approach has helped us to understand riparian wetlands in a larger landscape context and has

pushed us towards integrating across stream ecology and wetland ecology disciplinary boundaries.

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Chapter 3

REACH-LEVEL TOPOGRAPHIC CHARACTERISTICS ASSOCIATED WITH RIPARIAN WETLAND OCCURRENCE: A CASE STUDY IN SHAVERS CREEK WATERSHED, PENNSYLVANIA, USA

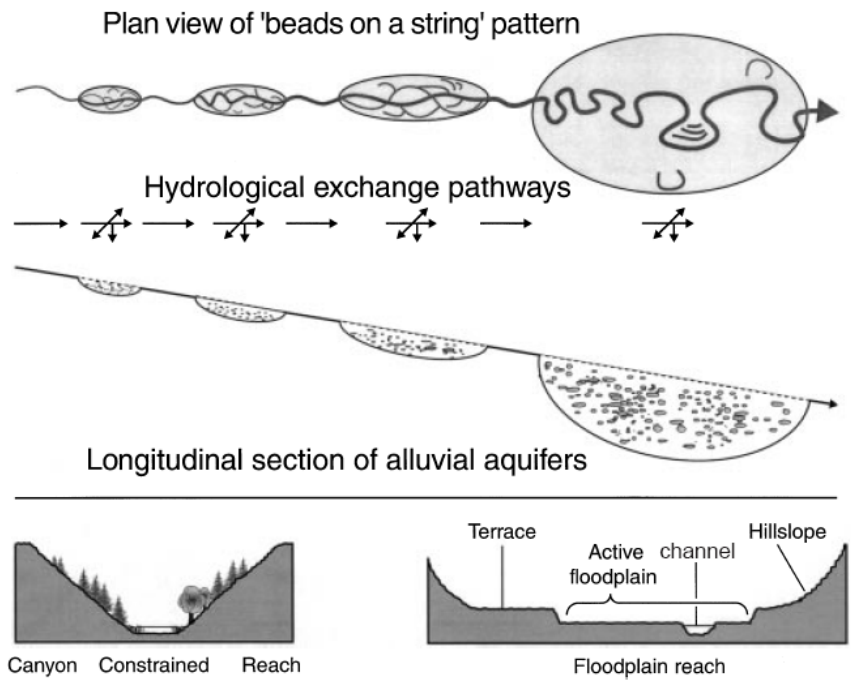
ABSTRACT

Hydrologically and physically complex floodplain systems occur where there is vertical, horizontal, and longitudinal hydrologic connectivity between the floodplain and the stream and groundwater systems. Reaches that have floodplain and wetland habitats typically are unconstrained latitudinally and longitudinally where overbank flooding can occur, and where there is regional, groundwater discharge at the toe-of-slopes. Building on this knowledge, the goal of this study was to improve the spatial prediction of headwater riparian wetlands through identification of suitable reach settings that allow for this three dimensional exchange of water. Known locations of mapped, non-open water National Wetlands Inventory (NWI) wetlands, field-identified non-NWI wetlands, and non-wetland locations (n=40, 30, and 35, respectively) were used to build a predictive partition tree. Predictive variables were DEM-derived topographic indices for the stream reaches: valley width, mean stream slope, and contributing area. The partition tree resulted in a 5-node tree (overall R^2=0.61). These classes ranged from very high likelihood of wetland occurrence to very low likelihood of wetland occurrence or least constrained to most constrained. This classification is a useful approach to characterizing wetland and non-wetland reach settings, especially in screening out the least likely wetland-supporting or most constrained reaches within a watershed.

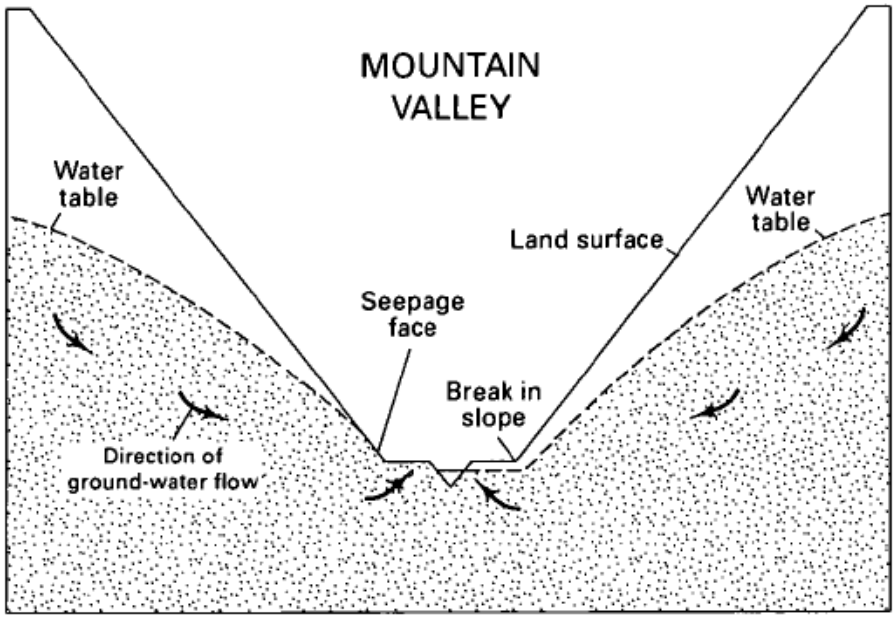
INTRODUCTION

JUSTIFICATION/LITERATURE REVIEW

Hydrologically and physically complex floodplain systems occur where there is vertical, horizontal, and longitudinal hydrologic connectivity between the floodplain and the stream and groundwater systems (Naiman and DeCamps 1997, Ward et al. 2002, Stanford et al. 2005) (Figure 1a). Though there is growing recognition that floodplain wetlands are driven by complex interactions between their landscape setting at multiple scales throughout the drainage network (Flores et al. 2006), they only occur where the local valley configuration allows for this three dimensional exchange of water. Hynes suggests that "in every respect the valley rules the stream" by controlling the magnitude, timing, and energy associated with overbank flows and with the delivery of materials from upstream (Hynes 1975). Reaches that have floodplain wetland habitat typically are unconstrained latitudinally and longitudinally where overbank flooding can occur, and where there is regional, ground water discharge at the toes of slopes (Winter 2000, Winter 2001) (Figure 1b). Building on this knowledge, the goal of this study was to improve the spatial prediction of headwater riparian wetlands through identification of suitable reach settings that allow for this three dimensional exchange of water. This geographic prediction of wetland occurrence can aid in the geographic scaling up of wetland field studies and planning for restoration and preservation of these systems.



(a)



(b)

Figure 1. Three-dimensional hydrologic exchange pathways that occur in a floodplain (an unconstrained) versus constrained reach (from Ward et al. 2002) (b) a Generalized Hydrologic Landscape for mountainous terrain showing the groundwater discharge at the break or toe of slope (from Winter 2001).

Hydrogeomorphic Setting at Multiple Scales

Characterizing the hydrogeomorphic setting of a wetland is important, but fairly difficult to do (Brinson 1993, Bedford 1996). Identification of "wetland-supportive environments" is of primary importance in wetland and floodplain studies (Adamus et al. 2002). It allows one to understand what potential a place holds for maintaining specific riparian community types and levels of habitat diversity. This helps identify appropriate reference conditions for monitoring and assessment. In the face of human disturbance, sensitive habitats may disappear, but it is difficult to know if a habitat has disappeared or if it was unlikely to have occurred in that setting. Further, understanding the geomorphic setting of a floodplain system is one tool for identifying appropriate locations for wetland or floodplain restoration. Far too frequently, restoration sites are selected due to access or ownership constraints, placing mitigation sites in physiographically inappropriate sites (Bedford 1996, Cole and Brooks 2000, Gebo 2009). Additionally, understanding geomorphic settings is useful for modeling purposes. In headwater systems, wetlands in particular are cryptic and difficult to detect through traditional mapping efforts due to their small size and screening by forest canopies (Creed et al. 2003). One study found the area of wetlands was underrepresented on the National Wetlands Inventory by 45% in the Mid-Atlantic Region (Wardop et al. 2007), due precisely to these attributes.

There is a growing understanding that riparian systems act in a larger network (Benda et al. 2004) and that their functions are controlled by connectivity at landscape or watershed scales. For example, even in undisturbed systems the site-level diversity macroinvertebrate diversity may be low (Alpha), but the between-site diversity (Beta) within the watershed may be very high (Yetter and Brooks unpublished). Finally,

considering wetland setting at a reach scale helps to scale up research findings to the level of a watershed or landscape (Benda et al. 2002). Also, understanding habitat occurrence at a reach scale can aid in efforts to understand, monitor, and manage a range of habitats in concert instead of focusing primarily on stream habitats at the exclusion of wetlands or vice versa.

Efforts to characterize wetland settings are challenging for several reasons. Some of the difficulties in characterizing the hydrogeomorphic setting of wetlands derives from the processes driving wetland habitat formation, which occurs at multiple scales (Adamus 2002, Dahl et al. 2004). Further, some of these drivers are stochastic processes that are difficult to predict in space and time and, therefore, difficult to map with confidence. For example, the driver of wetland occurrence associated with a headwater stream may operate at a fairly local scale, such as the occurrence of a clay confining layer or downed wood for which there is no regionally available datasets. Therefore, it is beneficial to think about wetland occurrence and the understanding of hydrogeomorphic settings as a series of probabilities. For example, the regional setting may provide a high likelihood of wetland occurrence, but the actual occurrence of a wetland may be driven by the precise location of stochastically-driven processes such as faults or fractures in bedrock. Again, this makes the characterization of wetland reaches more useful as an exercise in identifying levels of probability of occurrence (high or low) for varying types of wetlands. Usually, the extremes of hydrogeologic setting are easier to identify than moderate settings. For example, an extremely steep reach along a stream's longitudinal axis (>15% gradient) is likely to have only limited contact with adjacent wetlands and therefore have a low probability of floodplain wetland occurrence, while a moderately

steep reach (1-3% slope) may or may not have floodplain wetlands depending on other factors.

Beyond the issues with characterizing the settings of riparian headwater wetlands, there are additional challenges in attempting to comprehensively map these settings within a study area. This is an important endeavor for several reasons. The mapping challenges include: the lack of spatial data at the spatial resolution necessary to capture the process, the limited extent of available datasets, and the variability in mapping accuracy of spatial datasets. The limited spatial resolution of data can be seen in the resolution of spatially referenced county-level soils data. The characteristics of soils are important to processes acting at multiple scales, including those at the site level. For example, the occurrence of a confining layer, which is often delimited in a SSURGO soils layer as an inclusion in a fairly large polygon. An example of the limited extent of spatial datasets and the variability of data accuracy is the mapping of geologic faults and fractures. In Pennsylvania, these are extremely important drivers of wetland hydrogeologic setting, however, they are mapped in the best detail for only some regions of the state, notably Chester County (Sloto 1994, Kalinowsky et al. 1996) and in areas adjacent to universities with strong geosciences programs (PTGS 2004). Fortunately, topography is consistently mapped regionally at 30-m and 10-m horizontal resolutions (Gesch 2007), and with 1-m soon to be available for the entire state (Delano and Braun 2007).

Previous efforts have been fairly successful at enhancing the ability of current wetland inventory (NWI) to predict the occurrence in area of wetlands in the Ridge and Valley physiographic province (Wardrop 2007). In an effort to expand the sampling

frame for a wetland condition study beyond those found in the NWI, the study used spatial representations of several key physiographic variables to predict wetland occurrence. These variables included: stream proximity, NWI-wetland adjacency, breaks in slope, and the occurrence of geologic contacts (McLaughlin 1999). The first two indicators build on existing mapped habitat and proved to be successful indicators. The second two variables act as predictors of where ground water is most likely to be discharging in the landscape. Breaks in slope or toeslopes are areas where groundwater flow lines curve towards the surface. Lithologic contacts of differing hydraulic conductivity (K) are also likely locations for groundwater to discharge at the surface. Finally, fractures or fissures, though rarely consistently mapped, can provide a strong indicator of groundwater discharge.

Potential locations of floodplain habitats containing wetlands can be found by using contour lines on topographic maps to identify locations where streams flow from steep, constrained reaches into longitudinally and latitudinally unconstrained reaches. This method has proven to be highly effective at finding potential sampling sites with high floodplain complexity (Yetter and Brooks unpublished). Using topographic data, characterizing the constraint of reaches can be used to develop a procedure for identifying reaches with the high potential for having floodplain habitats. Further, floodplain initiation in headwaters can be defined as the transitional point from a confined valley setting to a partially confined valley setting (discontinuous floodplains) and a unconfined valley setting (continuous floodplain) (Jain et al. 2008). Conceptually, these unconstrained reaches allow for hydrologic complexity because sources of water come in three directions: groundwater, surface water, and overbank flooding (Naiman

and DeCamps 1997, Ward et al. 2002, Stanford et al. 2005). Unconstrained reaches can be areas of periodic deposition as flows slow when they change gradient and in turn the power of the stream decreases. Additionally, where there are unconstrained reaches, the fluvial processes driving the structure of the reach change. The processes change from debris flow, scour, and some flooding to a reach driven by channel migration, avulsion, and flooding (Montgomery 1999). In turn, the types of habitat change by allowing for off channel habitats and higher likelihood of hyporheic habitats in unconstrained reaches.

Headwater Riparian Wetlands

The Ridge and Valley is an unglaciated physiographic province, where wetlands comprise a fairly small portion of the landscape estimated at only 2-5% (Wardrop et al. 2007). In the unglaciated portions of the state, wetlands are typically associated with streams (Brooks and Tiner 1989) and first, second, and third order streams comprise about 75% of the state's total stream length. In the Upper Juniata watershed, it is estimated that wetlands associated with headwater streams comprise 73% of the total wetland area in the watershed (Wardrop et al. 2007a).

In smaller systems there is more connection with the hillslope; in larger systems the power of the stream can override local setting (Formann 1995). Morphology of smaller systems are more strongly influenced by stochastic events (trees falls, mass wasting, beaver activity, roads, large storm events, fractures, former mill ponds, and tile drainage) (Gomi et al. 2002, Gooderham et al. 2007).

The threshold between headwater and mainstem segments of a river is not always clear cut, and many different cutoffs have been used. The published regional HGM

wetland classifications have made the split between mainstem and head water wetlands based on the order of the associated stream: 1st or 2nd order (Cole et al. 1997) or 1st, 2nd, and 3rd order streams (Cole et al. 2008). In this case, the order they use is the Strahler order (Strahler 1952). However, the actual size of the stream associated with a wetland can be radically different within a stream order. Also, streams may not be mapped at a consistent scale or level of accuracy from region to region (Clarke et al. 2008).

The wetlands focused on for this study are predominantly headwater floodplain wetlands with some riparian depressions and topographic slope wetlands that occur within the floodplain of a headwater stream (Cole et al. 2008). Notably, this study did not look at open water, isolated, fringing, or stratigraphic slope wetlands (Cole et al. 1997, Cole et al. 2008).

Reach Characterization and Classification

To map the occurrence of reaches likely to support riparian wetlands, relevant metrics must be generated in a GIS. Longitudinal slope can be measured at a variety of spatial scales. Latitudinal constraint can be captured by measuring the width of the valley floor. Though this is a relatively straight forward concept, it is somewhat difficult to measure in a GIS. Stream power (or specific stream power) has been shown to be an important predictive variable for stream-associated habitats and can be calculated from DEMs of contributing area and channel gradient.

Unconstrained Reaches: Unconstrained reaches have the potential to allow for 3-dimensional hydrologic exchange and an unconstrained reach is defined by both the

degree of longitudinal slope and lateral constraint. One study in the Umatilla River Basin characterized floodplain settings with a range of GIS-generated variables including upstream channel gradient and historical and present floodplain width (Adamus et al. 2002). Additionally, a study of mountain drainages identified spatially overlapping process domains regarding the material fluxes in a watershed from source, transport, and response with variation in stream gradient alone (Montgomery and Buffington 1997).

Stream Power: The fundamental gradients of slope and discharge of a stream have been shown to correspond with changes in specific stream power, shear strength, and bedload of a stream (Dunne and Leopold 1979, Bledsoe et al. 2008). One comprehensive review of stream classification suggested that specific stream power was the best descriptor of the overall hydraulic and sedimentation regimes of a stream (Bledsoe et al. 2008). One study developed a genetic classification of stream reaches based on stream power with three main categories (high- or medium-energy, non-cohesive, and low-energy cohesive), which were then split into thirteen subcategories (Nanson and Croke 1992).

Fewer studies have used specific stream power in relation to wetland geomorphic settings, with a few notable exceptions. Stream power was used to classify reaches of the Arkansas River for restoration purposes (O'Neill et al. 1997). They used the specific stream power of a 10-year event as an indicator of the level of disturbance experienced in the channel and floodplain, and found that in high energy reaches ($\omega \geq 8$ W/m^2) the valley floors were dominated by the stream channel and in lower energy settings ($\omega \leq 3$ W/m^2) had larger alluvial valleys or meandering channels. These classes were identified through visual analysis of specific stream power plotted against distance for three stream

reaches. Another study on the Chickahominy River found that where stream power was low, there were larger concentrations of scrub-shrub emergent wetlands and the highest rates of sediment deposition (Hupp et al. 1993).

LIMITATIONS

There are some clear limitations to relying solely on topographic predictors of wetland occurrence. First, as mentioned before, there are other drivers of wetland occurrence than topography including soils, geology (changes in transmissivity, fractures and fissures, and karst formations), the regional topographic setting or landforms. And, as mentioned earlier, the potential landscape drivers of headwater riparian wetland occurrence function at multiple spatial scales. This study is focused on only one of those scales, the meso- or reach-scale. Also, other human disturbances can impact the actual occurrence from landscape-level changes in land use to site-level alterations including filling and drainage. Finally, the specific topographic predictors of wetland may vary with other physiographic settings.

GOALS

Given the limitations described above, the overarching intent of this study is to use topographic data to help identify potential reaches that could support headwater riparian wetlands at the landscape scale for entire watersheds.

More specific goals of this study are to:

- Map topographic metrics from prior studies as potential predictors of riparian wetland and floodplain complexity,

- Test these topographic metrics to determine is they differentiate between wetland associated and non-wetland stream reaches, and
- Map the probability of riparian wetland associated reaches based on these predictions.

Similarly the specific research questions addressed by this study ask:

- How accurately do modeled geomorphic variables derived from digital elevation models predict quantitative?

- Can these topographic characteristics be mapped to help indentify 'wetland supportive environments' or unconstrained reaches?

- Can known occurrences of wetlands be predicted from reach-level topographic variables?

METHODS

STUDY AREA

The Shaver's Creek watershed is a sub-basin of the Upper Juniata watershed within Huntingdon County, Pennsylvania, which contributes to the Susquehanna River and eventually to the Chesapeake Bay (Figure 2). The watershed is approximately 163 km^2 (63 $miles^2$) and at the 1:24,000 USGS topographic scale, it is a 5th order stream at its outlet. Currently, it is a rural basin with 71% forested-, 28% agricultural-, and less than 1% urban-land uses inferred from a three-class land cover classification of the 2001 National Land Cover Dataset (Brooks et al. 2004). Most of the agriculture occurs in the valleys, while the ridges are predominantly forested. In recent history, the majority of the basin's forest was cleared for use as charcoal in the iron industry at the turn of the 20th

century (Schein and Miller 1995). By the late 1930s, the land use pattern in the watershed followed the same general trend as seen today with forested ridges and agricultural valleys, with the notable exception that the valley was more intensely farmed with less forest buffering streams and wooded lots interspersed between farm fields (PADCNR 2010). The basin is predominantly sandstone (Tuscarora and Clinton Groups) in the upper portion of the watershed and primarily shale (Wills Creek, Bloomsburg, and Mifflin formations) along the mainstem of Shaver's Creek, with a small portion of the southeastern portion of the watershed underlain by carbonate rock (Keyser Formation) (PTGS 2001) (Figure 3b). The soils of the watershed are predominantly Edon-Weikert Complex on moderately steep to extremely steep slopes and Hazelton-Dekalb on moderately steep to steep slopes, making up 26% and 20% of the watershed respectively (SCS 1978). Approximately 7% of the watershed is covered by soil map units where the dominant component (>85%) is hydric (SCS 1978) (Figure 3c).

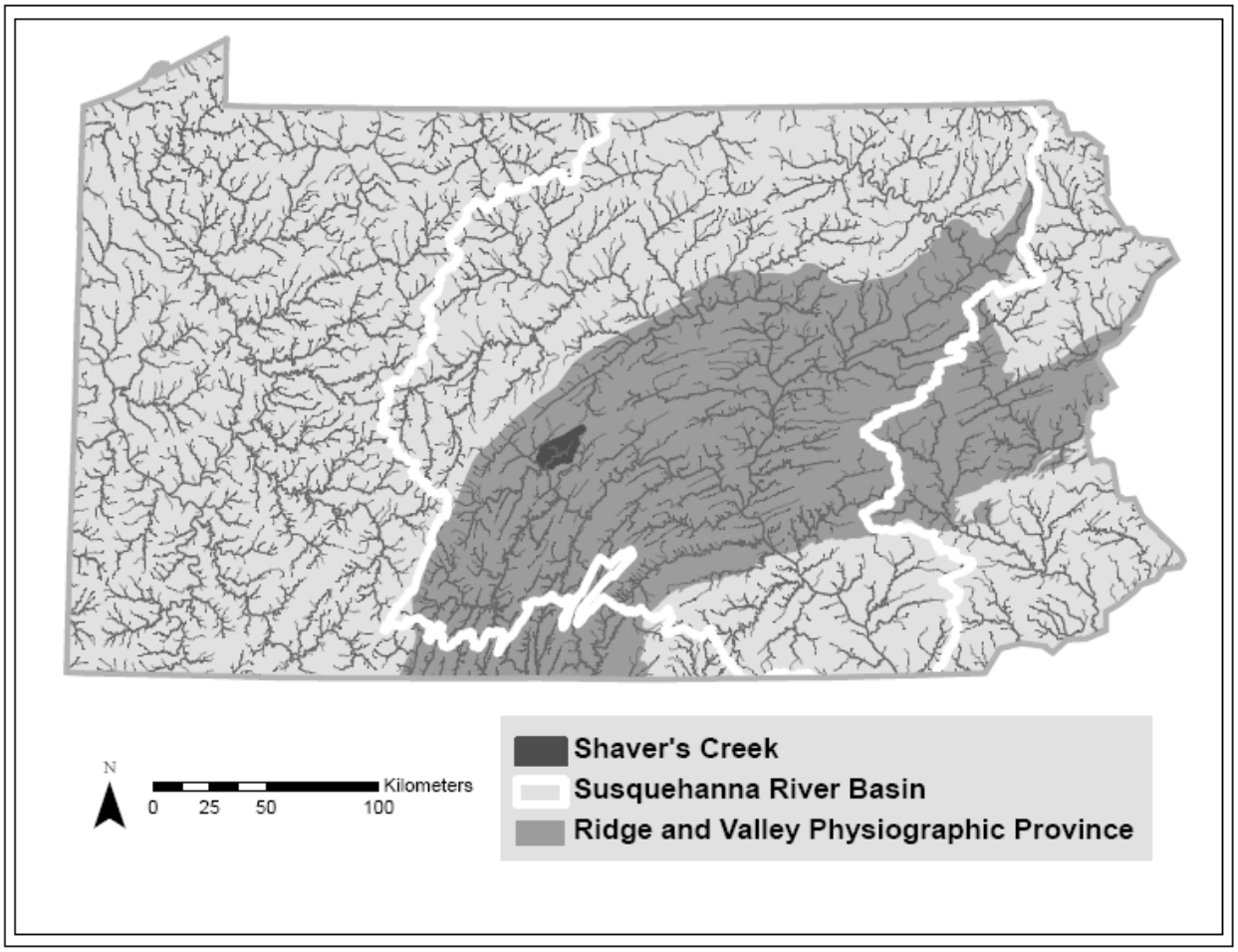
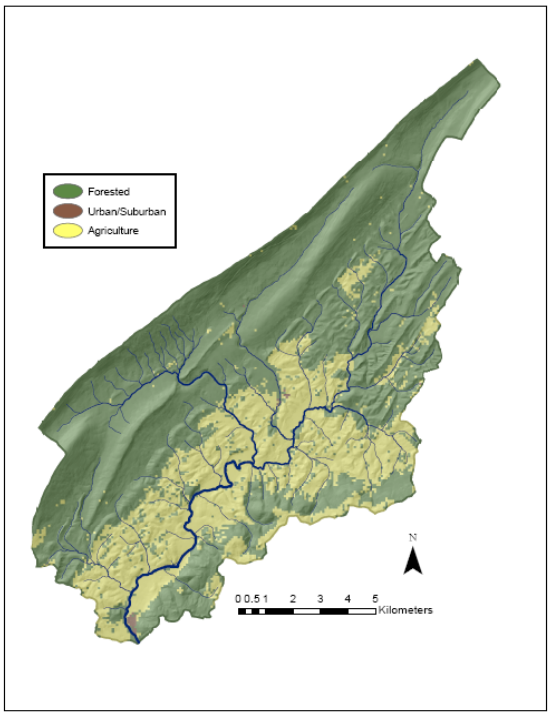


Figure 2. Location map showing Shaver's Creek and Susquehanna River watershed boundaries, the Ridge and Valley Physiographic Province boundaries, and streams (3rd order or greater).



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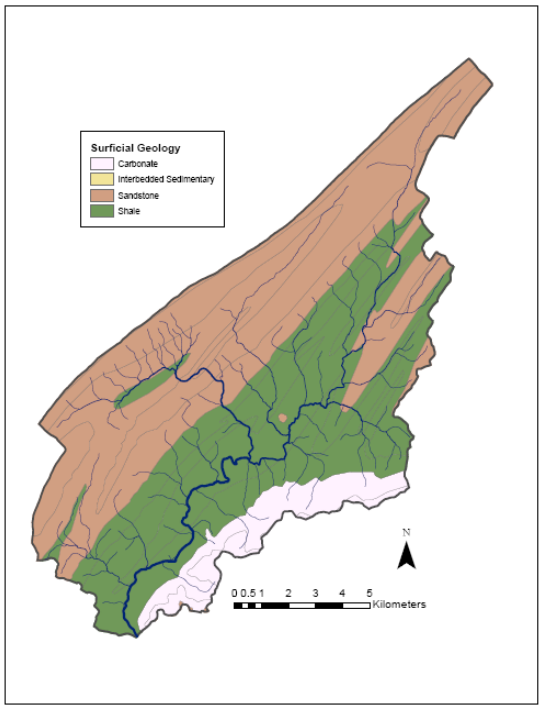

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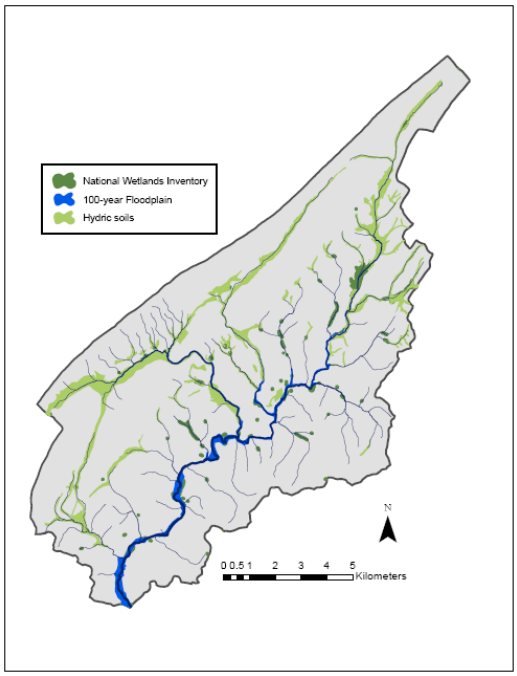


(c.)

Figure 3. Maps showing the distribution of (a) land cover (forested, urban/suburban, and agricultural), (b) major surficial geology types, and (c) the National Wetlands Inventory (boundaries exaggerated so they can be seen at a basin-scale), hydric soils, and 100-year floodplain for Shaver's Creek watershed.

SPATIAL DATA

The elevation data used in the analysis are digital elevation models (DEMs) from the National Elevation Dataset (NED) 1/3 arc second, which translates to approximately a 10 m horizontal resolution (http://seamless.usgs.gov/products/3arc.php) (Gesch 2007). These data were used because it had the finest horizontal resolution available at the time. Previous work has shown 10-m DEMs to accurately generate channel gradient for the purposes of predicting aquatic habitat (Neeson et al. 2008).

The stated vertical accuracy is +/- 7 m, however, this is an estimate for the data for the continental United States and the actual vertical resolution is based on the individual DEMs. Further, for portions of Pennsylvania the source data is based on 1 arc second (30 m horizontal resolution) data and interpolated into the 10 m DEMs. The DEMs are based on 7.5 minute 1:24,000 quadrangle data. All of Shaver's Creek has 10-m resolution source data.

The National Wetlands Inventory data was obtained through Riparia at the Pennsylvania State University, as earlier efforts to compile a statewide coverage also collapsed and coded Cowardin wetland types in the dataset (Brooks et al. 2004).

Additional spatial data used in validation and verification were accessed through the Pennsylvania Spatial Data Access (PASDA) and include: 100-year Floodplain, streams, and land use (Table 1).

Table 1. Characteristics of and sources for spatial data.

Type	Source	Date	Cell Size	Source Data
Land Use	Pennsylvania Gap Analysis Project	1998	30 m	LandSAT Thematic Mapper imagery (1993-1995)
Streams	Pennsylvania DEP	1996	N/A	1:24,000 7.5 minute USGS topographic maps
100-year Floodplain	Pennsylvania DEP	1999	N/A	1:24,000 Federal Emergency Management Agency Flood Insurance Rate Maps
Digital Elevation Models	U.S. Geologic Survey	varies	10 m	1:24,000 7.5 minute USGS topographic maps
Statewide National Wetlands Inventory	Penn State University	2004	N/A	National Wetlands Inventory maps, USFWS (varying dates)
Surficial Geology	PA Department of Conservation and Natural Resource	2001	25 m	mainly 1:250,000 Geologic Map of Pennsylvania (1980)
SSURGO Soils	USDA Conservation Service	1996-2002	N/A	Huntingdon County soil survey map (1978)

GENERATING STREAM SEGMENT CHARACTERISTICS

Synthetic streams are a powerful tool in characterizing the stream network within a basin. They are generated from a DEM through a series of steps including: the generation of a filled DEM, a slope grid, a flow direction grid, and a flow accumulation grid (Wilson and Gallant 2000). Synthetic streams were generated for this analysis using Netrace and Bldgrds. These are stand-alone, Fortran-based programs for the analysis of DEMs (Miller 2003). The topographic stream metrics, including valley width, valley slope, and contributing area (which is used in the calculation of specific stream power) were also calculated with these programs.

VALIDATION DATA

In order to test the ability of the mapped reach characteristics to predict the presence or absence of wetlands, known locations of wetland and non-wetland associated reaches were identified and mapped in the watershed as described below.

Wetland Reaches

The wetland-associated reaches came from two generalized sources: NWI and non-NWI wetlands. The NWI includes a range of wetland types, and since this analysis focuses on riparian wetlands, certain wetland types were filtered out. The types not included in the analysis were: all riverine, all lacustrine, and palustrine unconsolidated bottom (which are typically farm ponds in this region).

Riparian wetlands that do not occur in the NWI (non-NWI wetlands) were identified through a number of sources including: previously studied wetlands (Brooks 2004, Wardrop et al. 2007), wetlands identified through the Pennsylvania State University's School of Forest Resources managers for the Stone Valley Experimental Forest (SVEF) (Harding 2009), and those identified through field investigations for this study. Field investigations involved driving and walking large portions of the watershed to identify study sites for a companion research project.

The aerial extent of each non-NWI wetland (except for those identified through SVEF) was manually interpreted using a combination of site sketches, site visits, topographic maps, and hillshades generated with DEMs.

Non-wetland reaches

Known occurrences of reaches that do not have wetlands associated with them were identified through field visits for this study. There were many stream reaches in the watershed that did not have associated wetlands, but the history of human disturbance in the watershed (particularly tile drainage) make it difficult to assess whether or not a reach historically had wetlands associated or not (Adamus et al. 2002). Therefore, only stream reaches that did not appear to have a clear human disturbance driver for not having a wetland were used in the analysis.

STATISTICAL ANALYSIS

Descriptive Statistics

Synthetic streams were generated and the stream segments were assigned topographic characteristics. Their characteristics were described and explored through empirical cumulative distribution functions, which were generated by reach type for each variable and cross correlation plots and matrices (Minitab 2007). Cumulative distribution functions were used, because they are useful descriptive statistics that allow for a complete display of the data without imposed categories or filtering.

Predictive Statistics

The ability of stream segment topographic characteristics to predict wetland occurrence was quantified through a classification or partition tree approach. A partition tree generates groups of entities by making binary splits in predictor variables, resulting in groups with less and less variability in the response measure (Breiman et al. 1984). A classification tree was selected as an analytical tool because it is nonparametric. When

given multiple predictor variables it only uses predictor variables that add the most to the model, it can use categorical response variables, and it does not try to fit a single linear model to the response data (Feldesman 2002, King et al. 2005b). Classification trees were generated using "Partition" in JMP 8.0 (SAS Institute Inc., 1989-2008). The split criterion was maximum significance and the missing value rule was random. The minimum split criterion was 25 cases in a leaf node. Because the response variable was categorical the goodness of fit of the model was measured by both R^2 and the likelihood-ratio chi-square (G^2) (Hill and Lewicki 2007).

RESULTS

STREAM SEGMENT CHARACTERISTICS

The modeled stream variables used for mapping potential wetland occurrence were identified based on a review of the literature and included: valley width, longitudinal stream slope, stream power, and specific stream power.

Stream Initiation Threshold

In any synthetic stream generation process, a threshold for channel initiation must be determined. Meaning, how large an area draining to a single point indicates the beginning of a channel? In this case, a single value channel initiation threshold was used, and it was determined through an equation generated by Montgomery and Dietrich (1992):

$$a_{cr}S^{\alpha} = C$$

Where a_{cr} = critical specific contributing area (the upslope contributing surface area that contributes to a specified length of elevation contour in units of A/l or l^{1}), S = slope, α =

a constant between 1 and 2, and C is a constant. This method for determining channel initiation is generally used in flatter terrain (Miller 2003). C can be determined running an unconstrained flow accumulation model on the DEM and then plotting channel density generated against the drainage area on a log-log scale. Where there is an inflection in the curve this is an indicator that flow is occurring outside of defined channels or "feathering" is occurring (Clarke et al. 2008). In this case the inflection point occurred near 3,000 m^2 (0.003 km^2) (Figure 4), which was used as the channel initiation threshold. This value generated streams with a drainage density of approximately 3 km/km^2. The drainage density in the basin based on 1:24,000 USGS maps is 3.01 km/km^2 (190 km/ 63 km^2). The DLG typically underestimates stream length, so this is a conservative estimate of the stream length in the basin. Further, this threshold did not cause significant feathering in the synthetic streams that it generated.

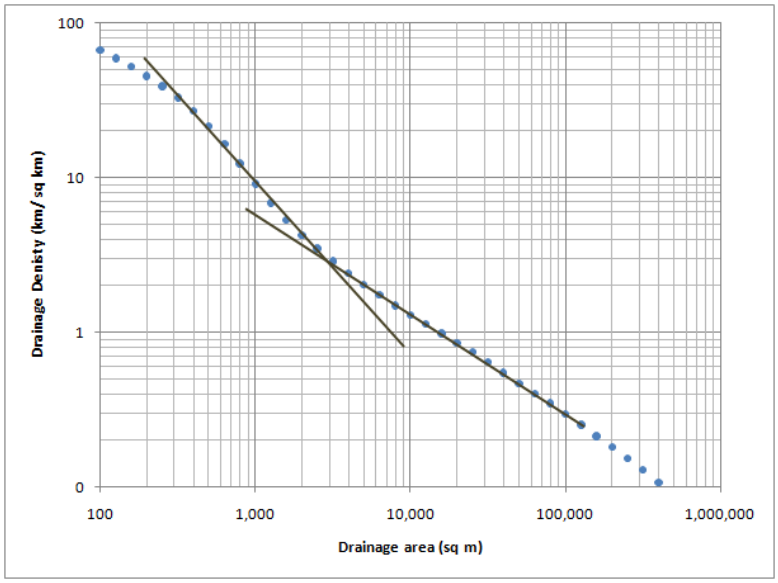


Figure 4. Log-log plot of drainage density (km/km^2) generated against the drainage area (m^2) threshold used.

Stream Segment Length

There are two key approaches to defining a reach break: length-based or geomorphically-based approaches. A length-based approach may simply mean using a fixed-length reach size or weighted-length by contributing area or stream order. One common geomorphic approach is to break reaches at stream junctions, either breaking reaches at every stream junction or only when a higher order, a fixed order stream size, or a stream with a specified contributing area is crossed. Further, geomorphic breaks in stream reaches could be defined by change in a specific geomorphic parameter that is often tied with the form and functioning of a stream, such as the longitudinal slope of the stream. Another example of using a geomorphic criterion is the use of specific stream power to identify reach breaks (Kondolf et al. 2003). Breaks can also be defined by the geomorphic similarity between reaches, based on single or multiple physiographic characteristics.

Beyond the selection of approach, the actual outcome of the reach characterization is influenced by the scale, both in resolution and extent, of one's data. In terms of resolution, using breaks with a stream order criteria is dependent on the resolution of the stream network you are using. Additionally, using a longitudinal slope break as the criteria would be dependent on the resolution of the DEMs used in the analysis; a coarse dataset would not capture slope changes over a few meters, which may be geomorphically important changes. In terms of extent, using geomorphic similarity would depend on the range of conditions encountered in the study area, where spanning a larger area might capture more variation in the same number of classes. Further, the extent over which geomorphic or physiographic variables are calculated may have a

significant effect on the location of reach breaks. For example, lateral confinement of a reach can be measured at multiple scales where a reach may appear unconfined at a relatively coarse scale (750 m), but confined at a closer scale (250 m). Finally, the choice of statistics used to determine the cutoffs between classes can vary greatly including crisp classes based on professional judgment or literature-based values or can involve ranges of class membership as in a fuzzy classification approach.

The approach used in this study was a fixed-length approach, using a fairly small (30 m) reach length. This size was chosen because it was close to but slightly larger than the resolution of the DEMs (10 m). Also, this approach was taken so that the least amount of a priori scaling was imposed on the calculated metrics and resulting analysis.

Contributing Area

As mentioned above, the contributing area for each cell was generated from a flow direction grid to calculate the number of cells contributing to the surface flow of a single cell. Flow direction was calculated using the D∞ flow direction algorithm which uses the slope of a cell to identify up to two downstream cells of any given cell and proportion how much of the flow from that cell will enter those two cells (Tarboton 1997). This algorithm uses a slightly different approach in steep versus flat terrain: a steepest decent approach is used in steep areas and a zonal approach in flat areas which forces flows from a flat area to the lowest adjacent point (Clarke et al. 2008). The number of accumulated cells (flow accumulation) was then multiplied by the area of each cell. The contributing area for each reach or stream segment was assigned based on the contributing area at the downstream end of the reach.

Active Channel Width

Active channel width was estimated based on a power function between contributing area and bankfull width (Leopold et al. 1964, Dunne and Leopold 1978, Reinfelds et al. 2004) for the stream data points in the Susquehanna River Basin in the U.S. Environmental Protection Agency's (USEPA) Wadeable Streams Assessment (WSA) (USEPA 2000, Paulsen 2008). Sites were used in the analysis if they: had single site visits and had non-zero values in the active channel width and depth measurements and the contributing area columns.

$$BFW = \alpha A_d^{\beta}$$

Where BFW is the width of the channel at bankfull (m), A_d is the drainage area to the point (km^2), and α and β are constants. For this data set $\alpha = 2.095$ and $\beta = 0.44$. This function has an adjusted $R^2 = 0.87$ with $p < 0.001$. These results are similar to other studies in the overlapping regions including the Ridge and Valley ($\alpha = 1.96$ and $\beta = 0.40$) (Mohamoud and Parmar 2006), the Mid-Atlantic ($\alpha = 2.57$ and $\beta = 0.38$) (Foustini et al. 2009), and the Northern Appalachian WSA Aggregate Ecoregion ($\alpha = 2.55$ and $\beta = 0.39$) (Foustini et al. 2009). The Foustini et al. results are not surprisingly close as they used different subsets of the same dataset as this analysis.

Active Channel Height

The attempt to generate a power function between active channel height measured at EMAP sites and contributing area resulted in very low R square value (R^2=0.18), even when controlling for contributing area and watershed-level land use disturbance. Therefore, in the absence of published regional relationships, the constants in the power

function were taken from the equation generated for the Oregon coastal streams (Clarke et al. 2008).

Valley Floor Width

The valley floor is determined as an area adjacent to the stream that is both: within a specified height of the channel elevation and having a specified gradient steeper than the channel gradient. The elevation range is specified by a given number of bank-full depths, which are estimated through the method described above. The algorithm makes this evaluation for each hillside (non-stream pixel) by comparing its slope and elevation values with the slope and elevation values of the reach it drains into. If the pixel falls within the designated range, it is considered part of the valley floor. The elevation criteria for defining the valley floor was given as 1.8 times the estimated bankfull depth for the stream channel (vh) based on the estimated value for a 50-year flood (Dunne and Leopold 1978). In another study in coastal Oregon, vh was assigned at 2.5 (Miller 2003). The slope criterion was set at within 15% slope greater than the slope of the channel. This slope class was taken from the Natural Resource Conservation Service topographic classes used in soil surveys (USDA 1951) which include: 3-8% slope as rolling or gently sloping, 8-15% as sloping, and > 15% as moderately steep or steep.

The width of the valley floor is then measured at each pixel within the stream channel. A transect across the valley floor is measured at a fixed height (vh) from the stream. Since the orientation of the valley relative to the direction of flow of the stream is not fixed (due to meanders, junctions, etc.) transects are measured across a range of angles, and the smallest transect is considered the alley width (Miller 2003, Clarke et al. 2008). The estimated channel width serves as a minimum for the valley width.

The measures of valley width for the stream segments derived from DEMs using NetTrace were checked for randomly selected reaches (n=106). The validation estimate of valley width was derived from a manual measurement of valley width from DEMs and topographic base maps. The reaches were subjectively classified as to whether they appeared to be constrained, moderately constrained, or unconstrained. The DEM-derived valley width and the manual valley estimates were highly correlated ($r=0.823$, $p<0.001$). Further, the three classes of constraint showed differences in valley width ($F=13.09$, $p<0.001$) with the unconstrained reaches significantly wider than the moderately constrained and constrained reaches (mean values of 65.2 m, 31.2 m, and 20.0 m respectively) (Figure 5). The moderately constrained reaches were only slightly and not significantly wider than the constrained reaches.

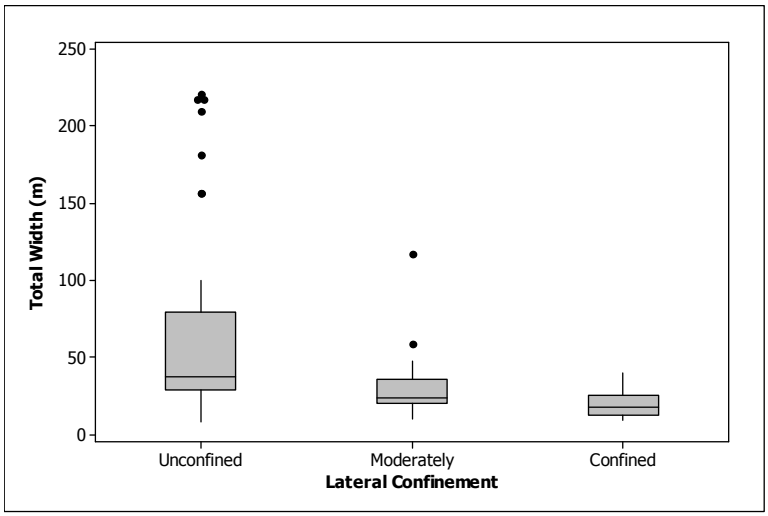


Figure5. Box and whisker plots of total valley width (m) derived from DEMs by class of lateral confinement (unconfined, moderately confined, and confined).

The valley width metric was checked, again, by randomly comparing it with the width of the mapped 100-year floodplain (n=40). For this watershed, the 100-year

floodplain is mapped only along the larger order streams, predominantly along the lower mainstem of Shaver's Creek. However, this check does give an indication of how well valley width is estimated in the larger stream reaches. With a comparison of 40 randomly generated locations, there was a significant correlation between the two measures of valley width ($r=0.809$, $p<0.001$). Because DEM-derived stream channels are single-line features and not double-line, where a tributary joins a wide mainstem channel there is a portion of the tributary that is within the actual estimated mainstream channel. Therefore, when these streams were encountered in the validation they were not included, and can be coded out of the final results of a stream network by buffering the single-line mainstem stream by the estimated channel width and marking tributaries that fall within that buffer as purely artifacts.

Longitudinal Slope

The slope of the reach was calculated by fitting a second-order polynomial to the elevations within a moving window 200-m long along the synthetic stream channel.

The measurements for channel slope were checked against hand-measured slopes calculated from contour lines on the USGS topographic quads (1:24,000). At randomly selected reaches ($n=50$) the longitudinal stream slope was calculated by measuring the distance along the stream between two contour lines and then dividing this length by the change in elevation (20 ft or 6.096 m). The results showed a significant correlation between the modeled stream gradient and those measured from the topographic maps ($r=0.857$, $p<0.001$).

Mean Annual Discharge

A regional relationship for mean annual discharge in Central Pennsylvania was developed with the equation (Brush 1961) from a series of basins which range in contributing area from 20 km^2 to 162,300 km^2:

$$Q_{2.3} = c\ A_d^n$$

Where $Q_{2.3}$ is the mean annual flood (m^3/s), A_d is the drainage area to the point (km^2), and c and n are constants. A regional relationship for mean annual discharge in Pennsylvania was developed based on a series of 18 stations which range in contributing area from 20 km^2 to 162,300 km^2 with the constants c = 0.50 and n = 0.80 (r=0.823) (Brush 1961, Knighton 1999). The 2.3 recurrence interval is often assumed to be the bankfull discharge (Dunne and Leopold 1978), however, in this case the recurrence interval was chosen from a plot of discharge against recurrence intervals for Standing Stone Creek in Central Pennsylvania, which showed the 2.3 year recurrence interval most closely predicted the discharge at bankfull (Brush 1961).

Stream Power and Specific Stream Power

Stream power is a commonly used parameter in stream classification and characterization approaches (Nanson and Croake 1992, Bledsoe et al. 2008, review in Kondolf et al. 2003) and is a measure of the energy of a section of stream defined as:

$$\Omega = \gamma QS$$

Where Ω is the stream power (watts/m or W/m), γ is the specific weight of water (9807 N/m^2), Q is discharge (m^3/s), and S is the energy slope (m/m). Energy slope is often represented simply by the bed slope of the stream reach (Jain et al. 2006). Specific stream

power is the stream power per unit area of the stream bed and is often used in geomorphologic studies because it accounts for variation in channel width and the variation of work performed over the actually area of the stream bed (Reinsfeld et al. 2004) Stream power can be estimated where:

$$\omega = \Omega / W$$

Where ω is the specific stream power (W/m^2), Ω is the total stream power as outlined above (W/m), and W is the width of the active channel (m).

VALIDATION

To identify the reach characteristics associated with the known locations of wetland and non-wetland occurrence, the stream reach shape file was intersected with a polygon layer of the wetland and non-wetland areas. The reaches that were completely contained by the known location polygons were then classified as wetland (NWI or non-NWI) and non-wetland associated reaches.

Therefore, only known locations of wetlands that intersected the modeled stream network were used in the analysis. Evaluation of the wetlands not intersecting the stream predominantly appeared to be isolated wetlands based on their wetland type (palustrine emergent) and/or landscape position. A few wetlands did not intersect the modeled streams because they either occurred in the extreme headwater of a stream, were a small sub-polygon of a larger wetland, occurred in a flat area where the synthetic stream did not closely match the location of the actual stream, or they were a very small distance from the stream (<10 m).

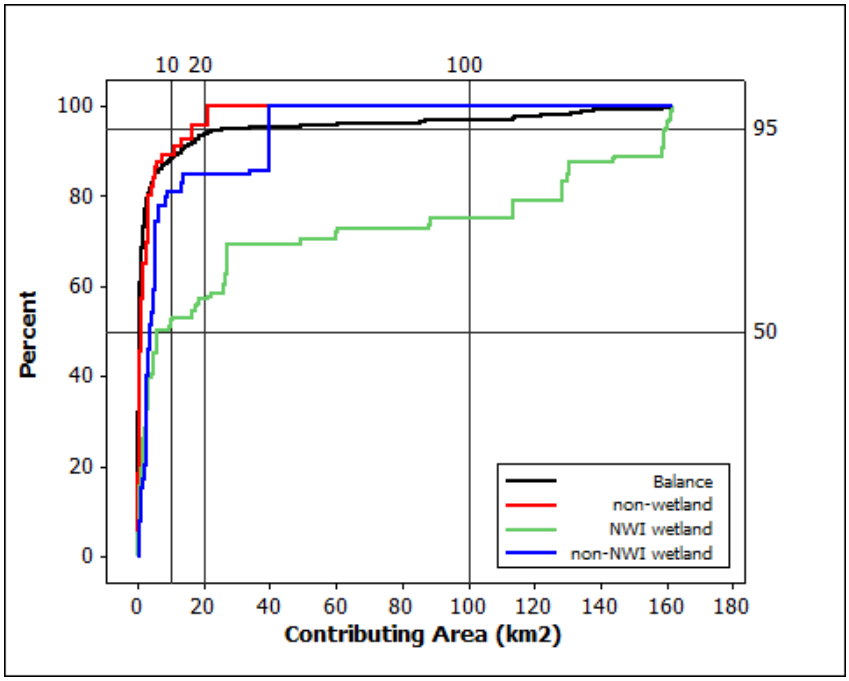
There were 40 NWI wetlands and 30 non-NWI wetlands used. They ranged in size from 0.0008-0.1259 km^2 (0.09-12.6 ac). The NWI wetlands were on average larger than the non-NWI wetlands (1.9 versus 1.1 ha respectively), however the median sizes were fairly close (0.8 versus 0.6 ha respectively). The total area of riparian wetlands identified through this study is 1.06 km^2, which though most likely not comprehensive inventory of riparian wetlands, accounts for only 0.65% of the watershed by area.

The contributing areas ranged from 0.05 to 161.89 km^2 (Table 2 and Figure 6a). The mean contributing area to the wetlands was much higher for NWI wetlands than non-NWI wetlands or non-wetland (374.7, 88.88, and 3.17 km^2 respectively), however, the median values were less than 10 km^2 for each type (Table 2 and Figure 6a). It should be noted that the contributing area for the entire Shaver's Creek watershed is 163 km^2.

There were 35 non-wetland reaches used in the study with contributing areas of mean of 3.17 km^2 and median of 0.87 km^2.

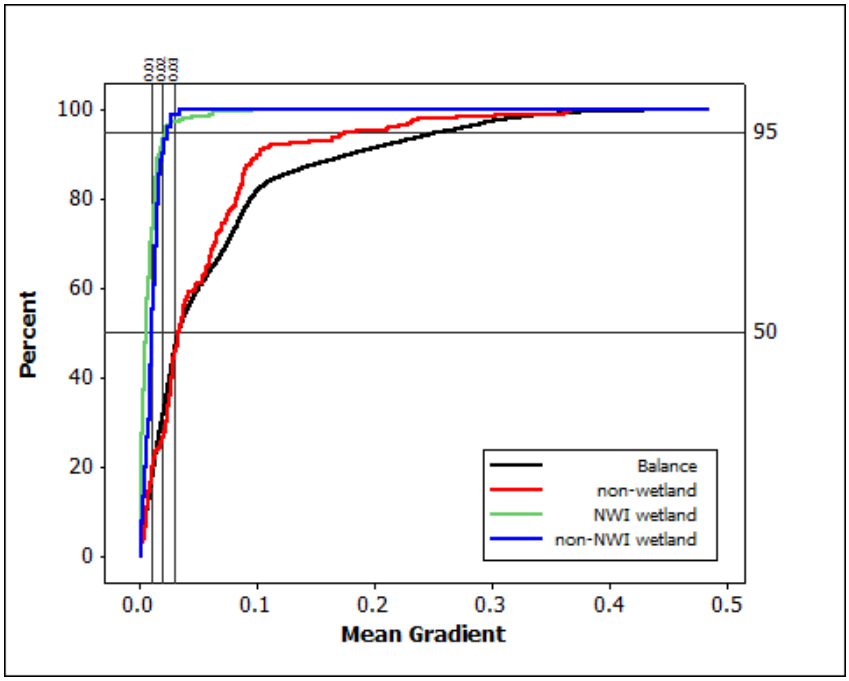
Table 2. Statistics for the contributing areas of know locations of wetlands and non-wetland locations and the area of the wetlands (NWI and non-NWI).

		NWI wetlands	non-NWI wetlands	non-wetland	Total
	Count	40	30	35	**105**
Wetland Area (km^2)	Sum	0.7439	0.3134	-	**1.0573**
	Avg	0.0186	0.0104	-	
	Med	0.0085	0.0066	-	
	Min	0.0008	0.0012	-	
	Max	0.1259	0.0495	-	
Wetland Area (acres)	Sum	183.8	77.5	-	**261.3**
	Avg	4.6	2.6	-	
	Med	2.1	1.6	-	
	Min	0.2	0.3	-	
	Max	31.1	12.2	-	
Contributing Area (km^2)	Avg	37.47	8.88	3.17	
	Med	8.70	2.46	0.87	
	Min	0.05	0.25	0.03	
	Max	161.89	119.53	21.04	

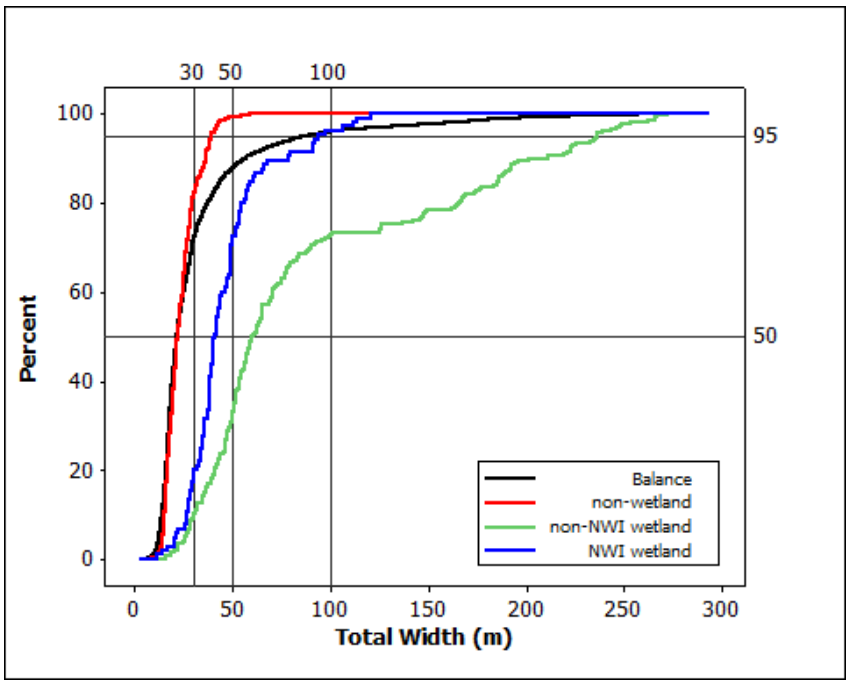


(a.)

(b.)


0.01
0.02
0.03
100
80
60
40
20
0
95
50
Percent
0.0
0.1
0.2
0.3
0.4
0.5
Mean Gradient
Balance
non-wetland
NWI wetland
non-NWI wetland

(c.)


30
50
100
100
80
60
40
20
0
95
50
Percent
0
50
100
150
200
250
300
Total Width (m)
Balance
non-wetland
non-NWI wetland
NWI wetland

(d.)

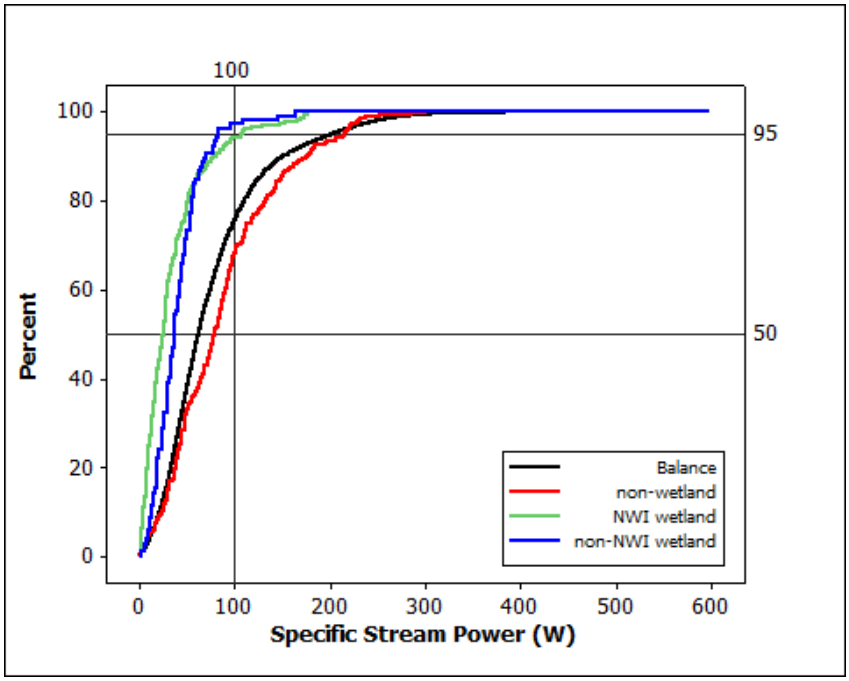


Figure 6. The empirical cumulative distribution function for the (a) contributing area (km^2), (b) mean gradient, (c) total valley width (m), and (d) specific stream power of the reaches associated with known locations of wetlands (NWI and non-NWI) and non-wetland areas and the balance of the watershed.

Comparisons of Stream Characteristics

Comparing the reaches associated with the wetland (NWI and non-NWI wetlands) and non-wetland reaches shows some marked differences in the measured stream variables.

Stream Gradient Wetland reaches (both NWI and non-NWI) are associated with much lower gradient stream reaches than the non-wetland reaches identified in this study (Figure 6b). Notably the NWI and non-NWI wetlands had a similar distribution of stream slope characteristics with approximately 95% of the wetland-associated reaches <2% gradient and 50%, and no wetland-associated reaches with a slope >8%. However, 95% of the non-wetland reaches were <20% gradient, 50% >3%, and less than approximately 15% with a gradient <1%.

Valley Width: The valley width was also lower for both of the wetland types than the non-wetland reaches (Figure 6c). Notably the NWI wetlands were generally associated with wider reaches than the non-wetlands. Very few non-wetland reaches had a width of > 40 m, while greater than half of the non-NWI and more than 80% of the NWI wetlands had a valley width >40 m.

Specific Stream Power: The distribution of specific stream power (W/m^2) was similar for the wetland and non-wetland associated stream segments (Figure 6d). They were both lower than the specific stream power for non-wetland associated segments, with nearly all wetland-associated stream segments and only 70% of all non-wetland associated stream segments having a specific stream power less than 100 W/m^2.

Cross-Correlation: The mapped stream characteristics were all significantly correlated ($p<0.001$) with each other (Table 3 and Figure 7). Further, mean gradient or the log of mean gradient had an absolute value of the Pearson's correlation coefficient > 0.45 (Table 3). The cross distribution plots, when colored as wetland or non-wetland associated stream segments, showed a fairly clear separation by type when looking at valley width and mean gradient.

Table 3. Cross correlation matrix Pearson's correlation coefficient (r) of the stream reach characteristics: contributing area (km^2), mean gradient (m/m), log mean gradient (m/m), valley width (m), and specific stream power (W). All correlations were statistically significant with a $p<0.001$.

	Contributing Area	Mean Gradient	Log Mean Gradient	Valley Width
Mean Gradient	-0.223			
Log Mean Gradient	-0.467	0.767		
Valley Width	0.783	-0.349	-0.661	
Specific Stream Power	-0.108	0.778	0.685	-0.266

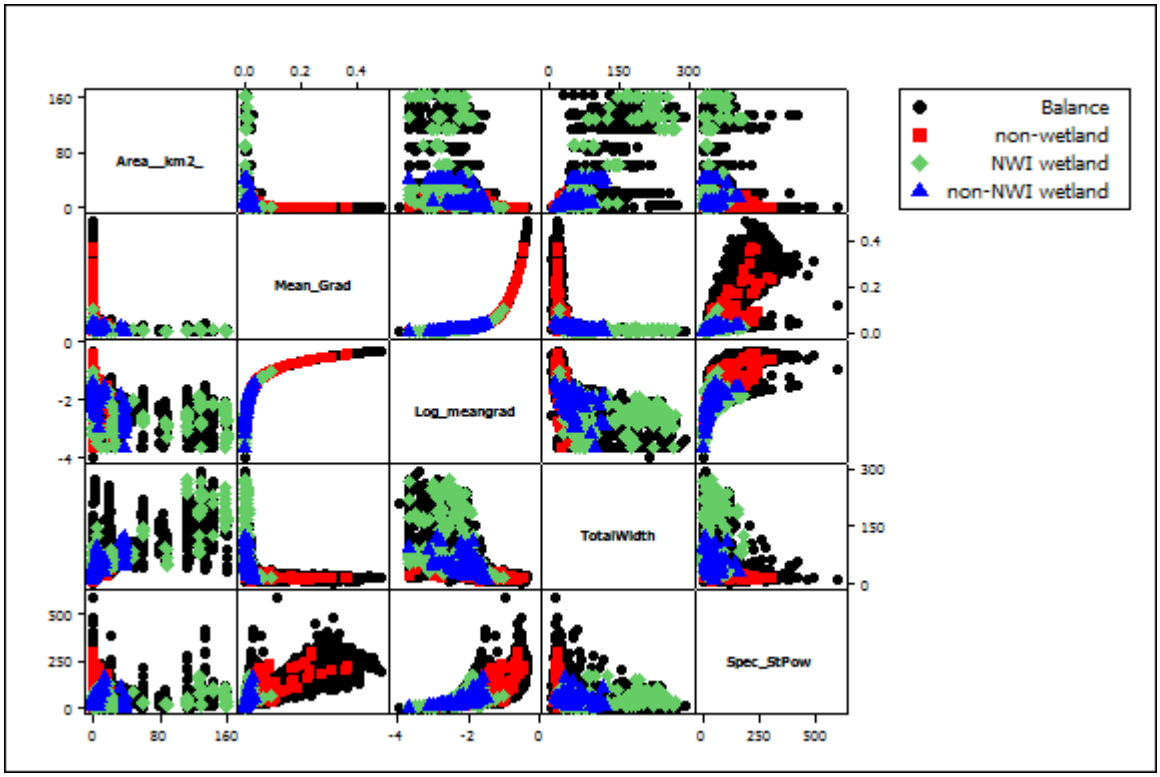


Figure 7. Cross-distribution matrix for the stream reach characteristics: contributing area (km^2), mean gradient (m/m), log mean gradient (m/m), valley width (m), and specific stream power (W). Individual segments are coded as to whether they are associated with: NWI wetlands, non-NWI wetlands, non-wetland, or are the balance of the segments.

Partition of Stream Segments

A partition or classification tree was used to predict the association of stream segments with or without wetlands. The predictive variables used were: mean gradient, valley width, specific stream power, and contributing area. Contributing area was added not because there was a logical conceptual model for the relationship between riparian wetland occurrence and a change in the variable, but instead to help to control for differences in the contributing area for the NWI wetland, non-NWI wetland, and non-wetland associated stream segments. If contributing area is a strong predictor of riparian wetland occurrence in this dataset it would be used in the partitioning of the tree. Further, some of the predictor variables used might not have a simple relationship with the

occurrence of riparian wetland or unconstrained settings, but be mitigated by their location in the watershed.

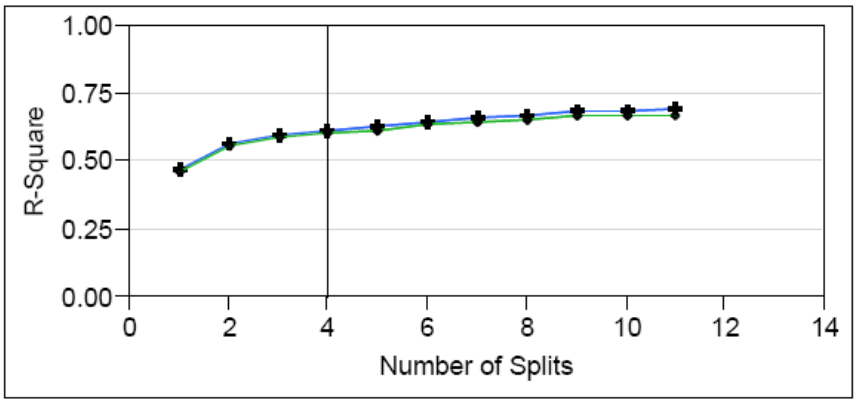
Trees were run with a k-fold cross validation with k=10, therefore, the data were separated into 10 sets, which were iteratively used as a validation set while the remaining nine sets were used to generate the tree. The tree was run out to 11 splits, when the model R^2 and the k-fold cross validated R^2 began to diverge and the candidate splits that improved the significance had too few cases (Figure 8a). This tree had an overall R^2=0.697, G^2= 295 and k-folded R^2=0.675, G^2=316. The tree was then pruned based on several criteria. First, the largest increases in R^2 came from the first few splits, with a relative leveling off on the split history curve (Figure 8a) and the cost of cross validation (or the difference between overall and k-folded R^2) was very small (Hill and Lewicki 2007). Also, the next split in the tree was to a node that was already 94% stream reaches. And, for the purpose of mapping, a more parsimonious tree with fewer splits would be more helpful in identifying mappable characteristics. Another test of the overall effectiveness of the partition tree is a receiver operating characteristic (ROC), which plots the sensitivity (true positives) on the y-axis against false positives (specificity) on the x-axis. The higher the curve lifts off of the diagonal, the better the fit of the model. This is measured by the area under the curve, where 0.5 would be a curve following the diagonal with the model having no predictive power, and an area of 1 under the curve would be a perfectly fit model. In this case the curves for the prediction of wetland and non-wetland associated segments is quite high (areas of 0.9479 for both types) (Figure 8b).

The resulting tree had four splits and five terminal nodes and had an overall R^2=0.615, G^2= 375 and k-folded R^2=0.600, G^2=390. The first, and most significant split,

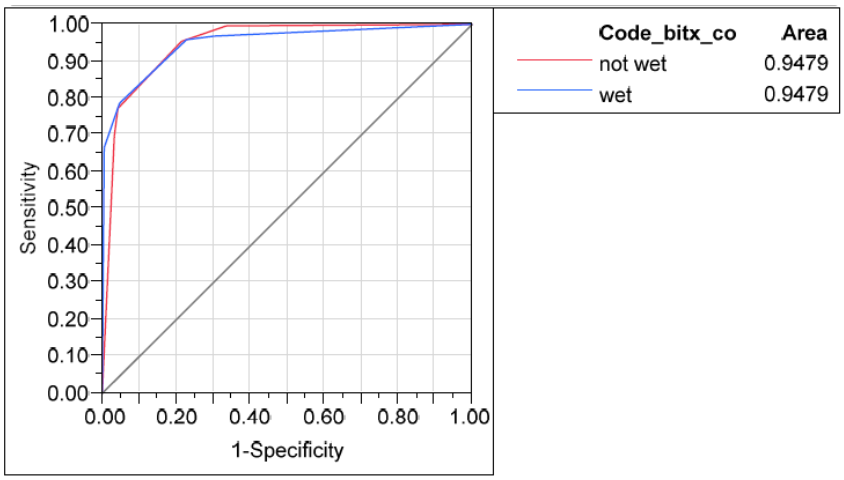
was from a break in valley width at 39 m. Those segments greater than or equal to 39 m were disproportionately more likely to be associated with wetlands than those less than 39 m (Figures 9 and 10). The branch of segments with valley widths > 39 m were further split at valley widths of 48.2 m with those wider valleys having even more likelihood of wetland occurrence based on the data. For those segments with a valley width < 39 m, the next split is on stream gradient with those segments having a slope greater than 0.023 much less likely to have wetlands associated with the stream. Further, the lower gradient group was then split on contributing area of the segment, with those having a contributing area < 10.9 m having a higher likelihood of wetland occurrence.

The final groupings of stream segments created five groups based on the likelihood of wetland occurrence (p=proportion of wetland-associated segments) (Figure 10):

- Very high (p=0.99)- highly laterally unconstrained
- High (p=0.80)- laterally unconstrained
- Moderate (p=0.54)- smaller streams, less laterally or longitudinally constrained for size
- Low (p=0.12)- larger streams, constrained longitudinally and somewhat laterally
- Very low (p=0.06)- highly constrained both laterally and longitudinally



(a.)



(b.)

Figure 8. (a) The split history curve showing of overall R^2 (upper curve) and k-fold R^2 (lower curve) against the potential splits in the tree. Also the (b) partition tree receiver operating characteristic (ROC) plot of the sensitivity (true positives) on the y-axis against false positives (specificity) on the x-axis.

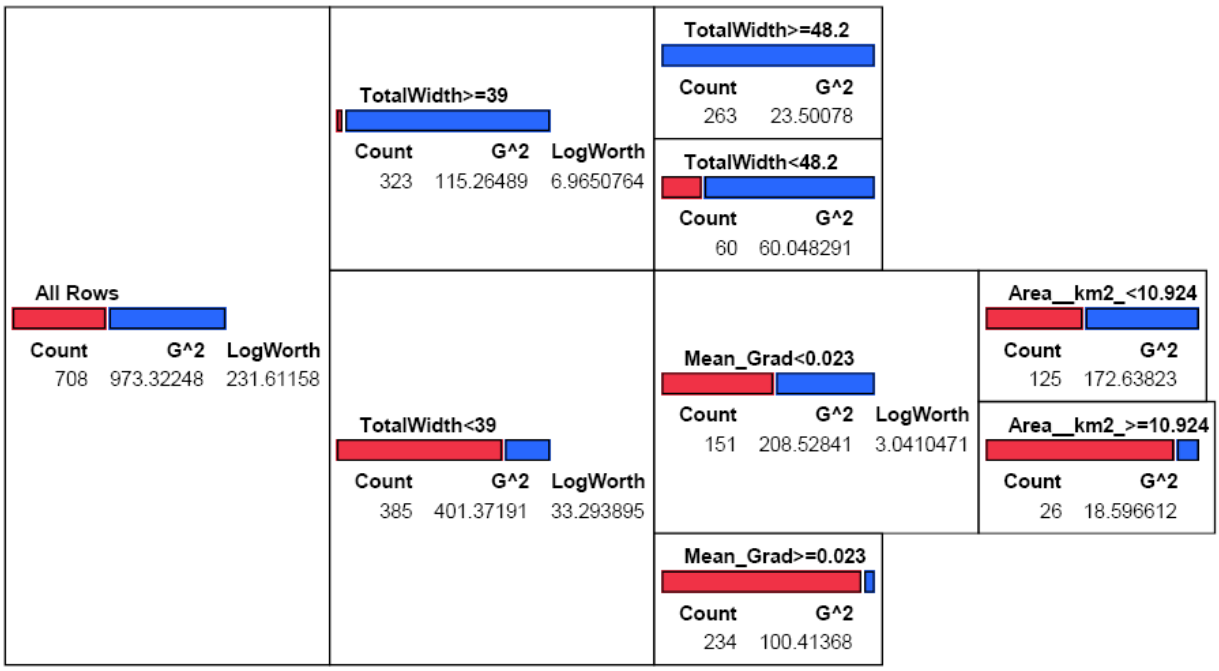


Figure 9. Partition tree diagram for the prediction of wetland (blue) and non-wetland (red) associated reaches based on total width, mean gradient, and contributing area.

Response Probabilities

Leaf	not wet	wet
TotalWidth>=39&TotalWidth>=48.2	0.0076	0.9924
TotalWidth>=39&TotalWidth<48.2	0.2000	0.8000
TotalWidth<39&Mean_Grad<0.023&Area__km2_<10.924	0.4640	0.5360
TotalWidth<39&Mean_Grad<0.023&Area__km2_>=10.924	0.8846	0.1154
TotalWidth<39&Mean_Grad>=0.023	0.9444	0.0556

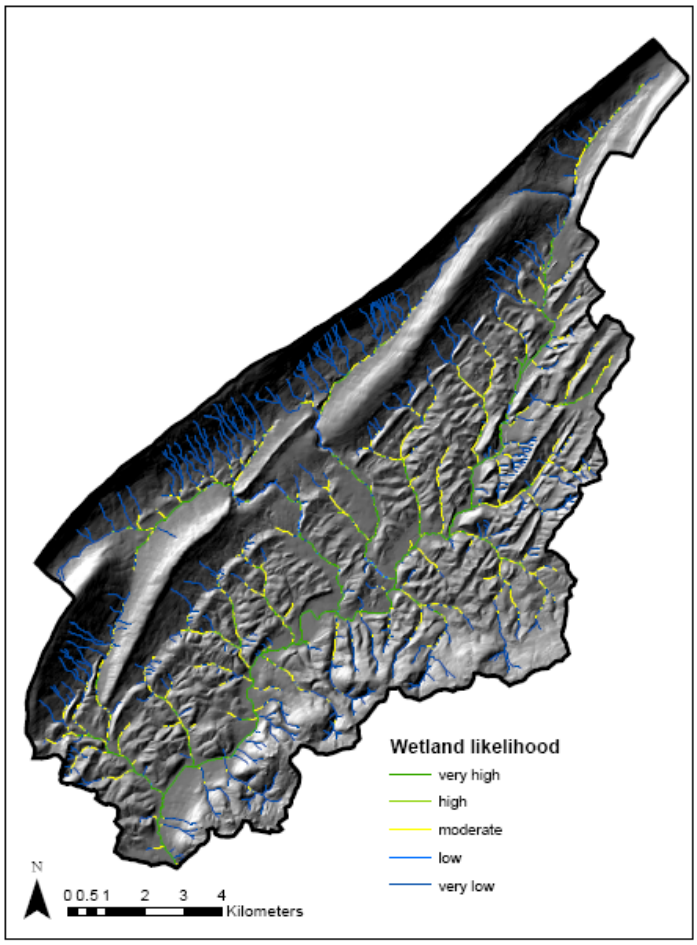
Response Counts

Leaf	not wet	wet
TotalWidth>=39&TotalWidth>=48.2	2	261
TotalWidth>=39&TotalWidth<48.2	12	48
TotalWidth<39&Mean_Grad<0.023&Area__km2_<10.924	58	67
TotalWidth<39&Mean_Grad<0.023&Area__km2_>=10.924	23	3
TotalWidth<39&Mean_Grad>=0.023	221	13

Figure 10. Leaf diagram for the partition tree showing response probabilities and counts for terminal nodes.

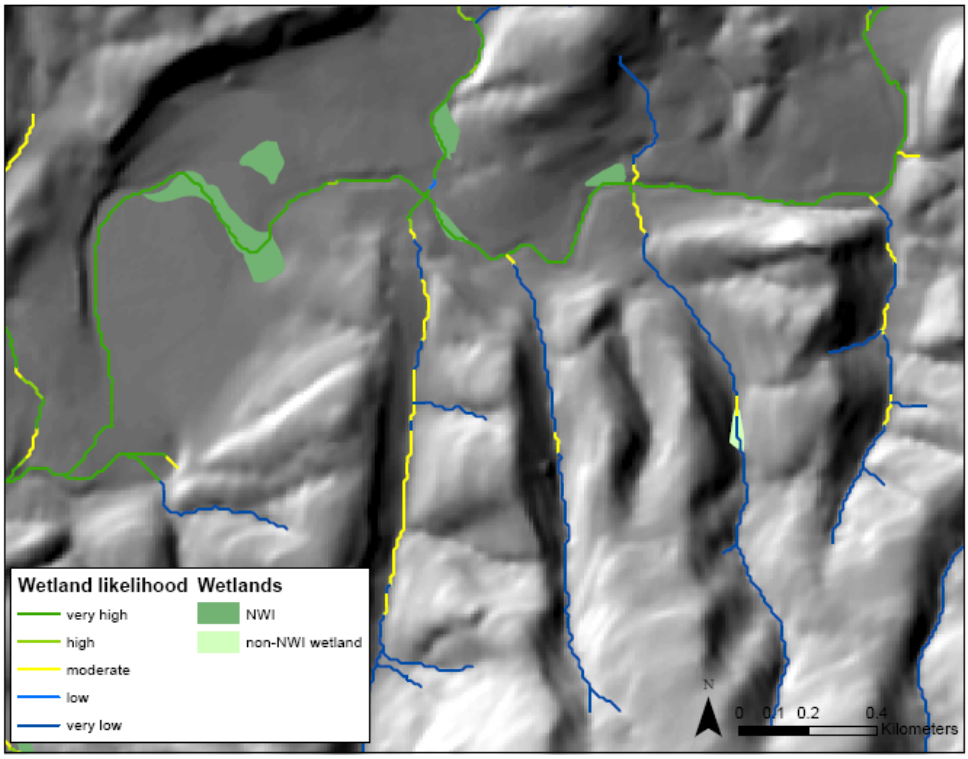
MAPPING PROBABILITY OF RIPARIAN WETLAND OCCURRENCE

The groupings generated by the partition tree can be used to map the probability of riparian wetland occurrence association with stream segments. This was done for the Shaver's Creek watershed (Figure 11a). More than half of the stream length in the watershed is in the low likelihood category with noticeably small average contributing areas (Table 4). The very high likelihood reaches make up 14% of the total length and have quite noticeably large contributing areas, while the high likelihood segments make up only 6% of the stream length and have much lower contributing areas than the very high class. Further, the low likelihood class has a much larger average contributing area than the moderate class, which is not surprising as the partition break was on contributing area, but the low class only makes up 1% of the stream length while the moderate class makes up 18%.


Wetland likelihood
very high
high
moderate
low
very low
N
0 0.5 1 2 3 4
Kilometers

(a.)

(b.)



(c.)

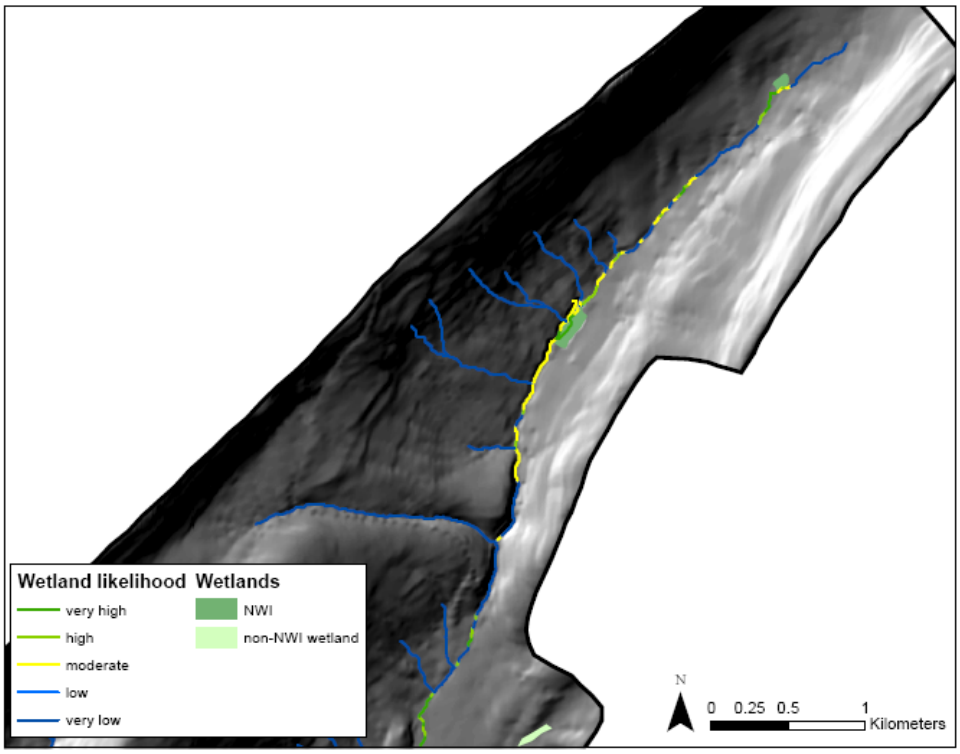


Figure 11. Maps of predicted wetland likelihood for (a) all of Shaver's Creek, (b) the Shedd Property, and (c) upper Shaver's Creek.

Table 4. Stream length (km), proportion of total stream length, and average contributing area (km^2) for stream segments in the Shaver's Creek watershed based on predicted likelihood of wetland occurrence.

Likelihood of Wetland Occurrence	Stream Length (km)	Proportion of Total Stream Length	Average Contributing Area (km^2)
Very high	45.2	0.14	46.75
High	19.7	0.06	7.16
Moderate	57.4	0.18	1.75
Low	3.6	0.01	17.73
Very low	198.4	0.61	0.59
Total:	324.2		

DISCUSSION

The overall goal was to enhance the ability to predict floodplain wetland occurrence using the knowledge that floodplains occur where there is lateral and longitudinal connectivity between the stream and riparian area, and then identify and map reach-level characteristics associated with riparian wetland occurrence.

TOPOGRAPHIC VARIABLES

One of the key research questions of this work was how accurately do modeled geomorphic variables derived from digital elevation models predict quantitative and qualitative reach-level variables? To address this question, an approach was identified to generate synthetic streams for the study watershed and model basic topographic variables from digital elevation models. Conceptually, valley width is one of the most important variables in identifying unconstrained reaches where wetlands might occur (Ward et al. 2002), but it is one of the hardest topographic variables to quantify. There are many programs and routines that generate synthetic stream and topographic indices (Tarboton

1997, Wilson and Gallant 2000), however, identifying a tool that could calculate valley width proved to be difficult. A series of freeware, Fortran-based programs (BLDGRDS and NETTRACE) had been used and validated in a study of streams in the Coast Range of Oregon to calculate valley and stream metrics including valley width for use in a multi-disciplinary aquatic ecology study (Clarke et al. 2008). Once mapped, this chapter addressed how accurately these modeled geomorphic variables derived from digital elevation models predicted reach-level variables. Notably, modeled valley width closely predicted manually measured valley widths ($r=0.823$, $p<0.001$) and 100-year floodplain widths ($r=0.809$, $p<0.001$). Also, modeled valley slope closely predicted valley slope measured from 1:24,000 USGS topographic maps ($r=0.857$, $p<0.001$). Manually measuring reach characteristics introduces its own set of errors: for example observer judgment in determining the limits of the valley and very widely separated iso-topo lines in low relief settings giving slope for a much larger stream segment than the target reach. But many studies blindly use GIS-measured topographic variables without any validation or verification (Clarke et al. 2008), so some positive corroboration that fairly coarse-level DEMs highly correlate with independent measures of reach characteristics is important.

MAPPING WETLAND SUPPORTIVE ENVIRONMENTS

Additionally, this work sought to explore if topographic characteristics can be mapped to help indentify 'wetland supportive environments' or unconstrained reaches? The most obvious predictions of wetland occurrence are that floodplain wetlands are less likely to occur in steep, constrained extreme headwaters. As mentioned earlier, this predictive mapping does not indicate that no wetlands occur in steep, constrained reaches, conversely stratigraphic slope wetlands are likely to be associated with steep stream

reaches (Cole et al. 2008). And, of course, there are entire wetland types, notably isolated wetlands, not associated with streams whose occurrence should be predicted through other means (Julian et al. 2009).

Also, the modeling of synthetic streams using DEMs, as described above, requires a selection of methodology and threshold selection, which invariably results in mapping some stream occurrences where there are none and in under representing headwater streams in other locations. One known example of under-representation in this watershed is a small un-named tributary to Shaver's Creek in the northwest portion of the watershed where an extreme headwater wetland was identified in the field and the synthetic streams were not generated that far up in the watershed.

Further, the larger, less horizontally constrained portions of the watershed are much more likely to have floodplain wetlands associated with the streams. This supports the notion that the downstream trend in floodplains are from discontinuous floodplains to continuous floodplains associated with larger streams (Jain et al. 2008). The mapping may over estimate the extent of the likelihood of wetlands associated with larger stream in the watershed. However, these are areas that have better mapping through the NWI than headwaters and also most likely do provide “wetland supportive” areas depending on the specific site-level characteristics.

KNOWN OCCURENCES

Finally, this work looked to predict known occurrences of wetlands from reach-level topographic variables. Using reach-level topographic variables (stream slope, valley width, specific stream power, and contributing area) as predictors and the occurrence of

wetland versus non-wetland reaches as the response variables, a classification tree was generated with an overall k-folded R^2=0.600. Though this is not a terribly high R^2, what is notable was the purity of two of the nodes of the tree, which together accounted for over 70% of the stream length in the model. The most constrained reaches (valley width < 39 m and stream slope >=0.023) were 94.4% non-wetland reaches and 33.1% of the stream length evaluated by the model. Similarly, the most unconstrained reaches (valley width >= 48m) were 99.2% wetland-associated reaches and made up 37.1% of the stream length evaluated by the model. Conversely, one of the classes had fairly low purity (53.6% wetland reaches) and made up 17.7% of the stream length evaluated by the model. These were not very wide (valley width < 39 m), flat (valley slope < 0.023), headwater reaches (<10.9 km^2). This seems to be an appropriate designation as floodplain occurrence in smaller streams seems to not only be a function of unconstrained reaches, but also very strongly influenced by groundwater discharge and stochastic events such as blow downs and beaver activity (Gomi et al. 2002, Olson & Burnett 2009). Additionally, the break in slope for longitudinally unconfined reaches at 0.023 is corroborated by other stream classification approaches for topographic settings where streams might have associated floodplains (Rosgen 1994, Kondolf et al. 2003). Additionally, other studies have found that a shift from a supply to capacity limited segments occurs at 3-2.5% stream slope (Flores et al. 2006, Bledsoe et al. 2008) supporting that the reach classification in this dissertation is process-based.

The classes generated by the partition tree indicate that unconstrained reaches can be identified through the use of freely available spatial data. The classes generated also have an associated probability of wetland occurrence, which is a useful way to think

about riparian wetland occurrence as, again, the drivers of actual occurrence are often stochastic and/or are not easily or readily mapped.

Noticeably, specific stream power was not used as a split criterion for the tree, though it has been proposed to be one of the most important variables in predicting stream morphology (Kondolf et al. 2003, Bledsoe et al. 2008). This may be due to the fact that it is highly correlated with other predictor variables that were better predictors for this set of data. Also, the ability to map specific stream power might not be good enough. This is somewhat difficult to directly validate, however, one of the modeled variables used in the specific stream power calculation is estimated stream width, which in the validation with stream survey data was shown to not be a tightly matched prediction.

Additionally, it is notable that the only slope criteria was a split in the laterally constrained branch of the tree with the split at a slightly higher slope than expected from field visits (2.3%). However, other studies have found that a shift from a supply to capacity limited segments occurs at 3% (Bledsoe et al. 2008) or at 2.5% (Flores et al. 2006) indicating that this break point is supported. Similarly, it is not clear whether or not valley width is the best predictor of wetland occurrence, as indicated by the analysis, or if it is the best modeled strong predictor of wetland occurrence. This could be isolated with more field assessment of the mapped predictor variables.

GENERAL

This approach proved effective at characterizing settings for riparian floodplains by quantifying laterally and longitudinally unconstrained reaches. This work could be

applicable to other regions through validation and verification with local data. Fortunately, the software used to generate the topographic metrics is freeware (Miller 2003), though a bit cryptically documented and requiring a fairly seasoned GIS operator to run. NetMap is a non-freeware, more advanced version of this software that will generate the topographic metrics (Benda et al. 2007).

More non-wetland reaches with larger contributing areas could be identified for the study, which might better refine the partitioning of the mid- to larger- streams (still relatively small compared to the greater Susquehanna basin). The issue of drained wetlands, however, is more of an issue as one moves down the watershed. For example, a wetland that occurs in the Shaver's Creek drainage just upstream of the confluence with Henry's Run was identified through field investigation as it does not occur on the NWI. It is a fairly unconstrained reach both laterally and longitudinally. The existing wetland is quite small and immediately adjacent to the stream with mowed field up to the edge of the wetland. There are clear indicators of some groundwater discharge in the mowed field and conversations with the landowner indicate that parts of the field were drained many years ago. This is an example of an area that is currently non-wetland (within the field), but seems to be a wetland-supporting setting historically.

It is also important to remember that the focus of this study is on riparian wetlands with a fairly specific definition. These do not include open water systems, isolated wetlands, and small seeps. Though these wetland types are not included in this study, they provide many wetland functions, but their occurrences are driven by different processes that were not modeled in this study (Julian et al. 2009).

There are pros and cons to using synthetic streams for spatial analysis (Clarke et al. 2008), with the biggest pro being that they are consistently generated and characterized within a study area. In this study, using synthetic stream was critical, because the program that was used to generate valley width required stream generated with specific numbering and ordering (Miller 2003). Additionally, the selection of the criteria used in generating streams can greatly influence the final synthetic stream network. For example, there is uncertainty around quantitative criteria for channel initiation (Benda et al. 2005).

FURTHER STUDY

It would be useful to identify areas where wetlands have occurred in the past and characterize the reach setting of those variables. One potential source for this information could be historic maps or Farm Bureau records of tile drainage projects. Aerial photo interpretation might also be a source of information, as these are available in this area and land cover in the watershed during some of the available time periods was predominantly agricultural, which might allow for easier interpretation of wetland locations.

Characterization or classification could be used in conjunction with other predictive methods relating to wetland setting and occurrence (Wardrop 2007). Several studies have successfully used satellite imagery to improve wetland detection (Ozesmi and Bauer 2002). This could be a useful approach to not only further validate or improve a reach classification scheme, but help to identify potentially hydrogeomorphically appropriate settings for restoration or creation of riparian wetlands.

Further studies could use differing methodology to identify non-mapped wetlands. One potential approach is random or stratified-random sampling, as riparian wetlands can be relatively rare in this physiographic province, a large number of sites would have to be visited to generate a sufficiently large population of wetlands on which to do analysis. Another potential approach is comprehensive mapping of stream reaches for example, one could walk and map all wetland occurrences in a 3.2-km stretch of stream. This approach would help to more comprehensively identify both wetland and non-wetland reaches. Further studies could be done on systems with contributing areas larger than that of Shaver's Creek, although wetlands associated with larger streams are generally better mapped by the NWI.

Future investigations are under way to look beyond the ability of these reach characteristics to predict riparian wetland occurrence, and use reach characterization to differentiate between predominantly wetland, floodplain, or mixed reaches beyond and reaches shown to have high complexity of floodplain, wetland, and stream habitats. Also, the characterization will be used in different physiographic settings to see if the predictive power holds or if the classification should be altered in different physiographic settings.

Knowing where wetlands occur in a watershed can help to understand their functioning and their delivery of ecosystem services. Understanding where ecologically and hydrologically complex systems occur or could occur in a watershed allows for the possibility of looking at a riverscape as a series of “hydrogeomorphic patches” that act as “Functional Process Zones” with varying degrees of connectivity in time and space (Standford et al. 2005, Thorp et al. 2006).

CONCLUSIONS

This study supports previous findings that though the NWI is an important data source for mapping wetlands, they do under represent wetland occurrence, in this case headwater riparian wetlands.

Additionally, the literature suggests that there are topographic reach characteristics that can help to predict reach types and, in turn, riparian wetland occurrence. This study suggests that some of these characteristics can be modeled and mapped using freely available spatial data and software. These modeled reach characteristics are fairly accurate, though there is room for improvement. Specifically, bankfull depth had a low correlation with field-measured data, but is used in modeling valley width, which appears to be a strong predictor of wetland occurrence. In turn, these mapped variables were indicative in predicting the occurrence of wetlands by reaches in the study area.

Finally, a predictive map of potential wetland occurrence can be generated using the techniques outlined above. These maps might be particularly useful when used in conjunction with other wetland predictive landscape variables or satellite imagery. Identifying these unconstrained reaches where riparian wetlands occur can be useful to managers and planners in efforts to protect and restore hydrologic and biological complexity within a watershed. This approach provides a step towards integrating streams and wetland studies.

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APPENDIX

Input “parameter” and “instructions” files for DEM processing programs (NetTrace and BLDGrid.

Input Parameters for Bld_grds and Trace
==========
2 flow direction algorithm (1 for Tarboton, 2 for Tarboton + convergence)
2 sl, number of pixels over which slope is calculated (> 1 to address "pocket terracing")
2.0 dig; depth of DEM incision for drainage enforcement
1 Channel threshold criteria:(1) Drainage area (2) Specific drainage area.
3000. channel_area_threshold ! maximum area for zero-order channel
3000. C_min, square meters ! (1500./1000000. for Oregon)
2.0 c_exp, slope exponent
0.25 S_max ! minimum slope for landslide potential, calibrated to DEM with landslide inventory
1.5 P_min, minimum number of inflowing cells for channel head
1 lcheckmax, number of contiguous pixels with Pin >= P_min for channel head
50. lstop_max ! maximum length for unchannelized, low-gradient debris flow runout, DEM-resolution dependent
200. Xmin, minimum window length for channel gradient estimation
200. Xmax, maximum window length
0.001 Smin, gradient at and below which Xmax applies
0.2 Smax, gradient at and above which Xmin applies
2 Fit Order, integer, polynomial order for fit
50. junction_length ! channel length used to estimate junction angles
2.094762 cw1_small, channel width function,
0.43832 cw2_small, channel width function
0.0 cw3_small
2.094762 cw1_big
0.43832 cw2_big
0.0 cw3_big
0.327968 depth_coefficient_1, bank-full depth = depth_coefficient_1*(area**depth_coefficient_2)
0.251918 depth_coefficient_2
2 reach method: 1) channel widths, 2) specified length !
20 # of channel widths for a reach, for reach-method 1
30. minimum reach length in meters, for reach-method 2
30. maximum reach length in meters, reach-method 2
0.04 area (km2) at and below which minimum reach length is enforced, reach-method 2
50.0 area (km2) at and above which maximum reach length is enforced, reach-method 2
150. minimum reach length for increasing max_grad_down

```
200.        maximum reach length for increasing max_grad_down
0.04        Drainage area (sq km) at and below which minimum reach length applies
50.         Drainage area (sq km) at and above which maximum reach length applies
1.0         Area weighting for reach breaks (larger values increase effect of tributary inputs)
1.8         vh, number of bank-full depths above channel to qualify as floodplain
0.15        ds_v, increase over channel gradient to qualify as floodplain
6.3187e-6   Mean annual flow, coefficient 1, AF = c1*(Area^c2)*(Precip^c3)
0.990       Mean annual flow, coefficient 2, Area in acres, Precip in inches
1.593       Mean annual flow, coefficient 3
0.0         gcoef1, field_gradient% = gcoef1 + gcoef2*(DEM_gradient%^gcoef3)
1.019785    gcoef2
0.825982    gcoef3
0.20        end of calibrated gradient
0.30        start of DEM gradient, linear combination in between
3.79        trib effects coefficient 1
1.96        trib effects coefficient 2
0.0437      trib patch size coefficient 1
0.3867      trib patch size coefficient 2
================================================================
UTM grid information
================================================================
27          datum, 83 for NAD83, 27 for NAD27
10          UTM zone number
================================================================
SOIL parameters (assuming metric units) used by SHALSTAB
================================================================
2000.       Soil Saturated Bulk Density (kilograms per cubic meter)
45.         Soil Friction Angle (degrees)
0.0         Soil Cohesion (Pascals)
65.0        Saturated Soil Conductivity (meters per day)
1.0         Soil Depth (meters)
0.05        Soil Porosity
================================================================
RASTER file generation for probability of debris-flow delivery
================================================================
3           number of fish-barring gradients
0.07        Downstream gradient to bar fish passage(coho)
0.20        Downstream gradient to bar fish passage(fish bearing)
100         all streams
```

Instruction file for netrace
==========
==========SHAPEFILE options
---
y ARCVIEW shape file output for channel reaches (y/n)
y ARCVIEW shape file output for tributary junctions (y/n)
y Link shape file (channel links only, used to create routed channel coverage) (y/n)
n Force reach breaks at channel junctions (y/n)
1 1) Fixed-length reaches, or 2) homogenous reaches
2 Gradient calculation method: 1) via contours, 2) poly fit over centered window, 3) none
2 Channel width estimation method: 1 = a*(Area^b)*(Prec^c), 2 = a*(Area^b), 3 = none
y Valley width (y/n)
y Valley side slopes (y/n)
2 Mean annual discharge (cfs) calculation method 1) = a*(Area^b)*(Prec^c), 2 = a*(Area^b), 3 = none
y Intrinsic Potential
1 Tributary effects method: 1) logistic equation, 2) linear, 3) none
3 Debris flow delivery model 1) the old one, 2) the new one, 3) none
1 Topographic index 1 = SHALSTAB, 2 = Slope/convergence
y Include lake attribute (even if no lake mask) (y/n)
n Specified reach endpoints (requires input file with endpoint locations) (y/n)
y Create channel mask (y/n)

RASTER output options
---
y Hillslope pixel distance to nearest stream channel, raster file (y/n)
y Hillslope pixel delivered-to-channel-reach ID, raster file (y/n) (requires reach shapefile)
y Create valley floor raster image vmask_ID.flt (.hdr) (y/n) (requires reach shapefile)

Chapter 4

CHARACTERIZING HYDROLOGIC REGIMES OF HEADWATER RIPARIAN WETLANDS: CASE STUDIES IN THE RIDGE AND VALLEY OF PENNSYLVANIA

ABSTRACT

Hydrology is considered to be the main driver of wetland ecology, function, and persistence, however, despite the recognized importance of hydrology on the form and function of wetlands, there is a lack of research in basic hydrological characterization of wetlands. In response there is a call for determining hydrologic behaviors of wetlands, particularly in response to changing climatic drivers. Past studies of hydrology in headwater wetlands of the Mid-Atlantic found that there were differences in proportion of time of saturation in the root zone hydrogeomorphic classes of wetlands. However, the water-level metrics used did not capture the "flashiness" or rate of movement of water into or out of the root zone or the responsiveness of these systems to precipitation events under varying antecedent moisture regimes. This study used a suite of time series analyses to explore the hydrographs of five headwater wetlands in terms of their dynamics and response to climatic drivers. Cross correlations between daily differences in water levels and precipitation showed significant correlations for most wetlands under dry and wet conditions on the same day time lag. Of the wetlands evaluated, all experienced a summer drawdown in water level except for the wettest sites in the wettest years. Further, the timing of the beginning of summer drawdown varied greatly for the period of record for the three studied wetlands (slope=61 days; headwater floodplain=58

days; and riparian depression=91 days) with the slope wetland drawing down earlier on average than headwater floodplain or slope wetlands (average day of the year 132, 156, 152 respectively). The moving averages of the water levels generally followed the trends of the downstream stream baseflow, except for the wettest site in the wettest year. Though the hydrologic data are only available as a highly discontinuous record over a 10-year period, more continuous records when analyzed as case studies with time series analyses can give insight into the dynamics and responses of hydrologic behavior of headwater wetlands to climatic drivers.

INTRODUCTION

The hydrologic regime of a river is considered the "master variable" in structuring river ecosystems (Poff et al. 1997). Likewise, hydrology is considered to be the main driver of wetland ecology, function, and persistence. Mitsch and Gosselink (1993:115) suggest that "hydrology is probably the single most important determinant of the maintenance of specific types of wetlands and wetland processes". Despite the recognized importance of hydrology on the form and function of wetlands, there is a lack of research in basic hydrological characterization of wetlands. Specifically, the National Research Council (National Research Council 1995: 91) stated that a "major technical challenge is 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".

The hydrologic regime of an aquatic system can be characterized in terms of 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 riparian ecosystems (Karr 1991, Karr and Chu 1999). Reduction in the magnitude and dynamics of flooding can result in a reduction in the biophysical complexity of an ecosystem (Richter et al. 2003), which can result in a shift in the invertebrate communities both on the surface (Richards and Host 1994, Lammert and Allan 1999) and in hyporheic zones (Poole et al. 2006). 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, Goslee et al. 1997, Miller and Zedler 2003, Magee and Kentula 2005). Further, the timing, duration, and dynamics of the hydrologic regime influences the biogeochemical environment of soils (Richardson and Vepraskas 2001).

With the recognition that hydrology is the driver of many structuring processes in river ecosystems and that hydrologic alteration is a major anthropogenic stressor on aquatic ecosystems (U.S. EPA 1998), there has been an effort to quantify what constitutes a “natural flow regime” in river systems (Grimm et al. 1997, Poff et al. 1997, Holland and Moore 2003, Richter et al. 2003). Richter et al. (1997) suggest a “Range of Variability Approach” to characterizing flow regimes using 32 indices in five categories: magnitude of monthly water conditions, magnitude and duration of annual extreme water conditions, timing of annual extreme water conditions, frequency and duration of high/low pulses, and rate/frequency of water condition change. Another study of 62 streams in New Zealand compiled 35 indices in three main categories: general, high, and low flow (Clausen and Biggs 2000). Olden and Poff (2003), suggesting a shift from a

paucity of indices to characterize flow regimes to an overabundance, compiled 171 flow indices from 13 papers, and grouped them into five main categories: magnitude (average, low, and high), frequency (low and high), duration (low and high), timing, and rate of change. They calculated these statistics for 420 stream gages in the United States and, in concordance with other studies (Poff et al. 1997, Toner and Keddy 1997, Cole and Brooks 2000) and found many of the indices were redundant and reduced the data set with a Principal Components Analysis.

In wetland hydrologic research a range of metrics and statistical approaches have been used to quantify characteristic wetland hydrologic regimes. Hunt et. al (1999:461) emphasized the importance of capturing the dynamics of water level fluctuations, and states that "(t)he challenge in assessing the hydroperiod for a particular site is to determine the appropriate statistics to characterize the temporal variability of the hydrograph". One study in southeastern Michigan compared variability in water levels with permeability of surficial deposits at the site (Merkey 2006). Another study in Arkansas and Missouri quantified changes in baseflow of stream gage data to infer changes in water regime to adjacent wetlands (Nestler and Long 1997). A study in the Prairie Pothole Region of the United States looked at long-term water level oscillations, and developed water regime indices for isolated wetlands (van der Valk 2005). For plant communities in British wetlands, researchers characterized the range of water depths on a site and the timing and duration of flooding (Mountford and Chapman 1993). Several studies look at the role of varying open water levels on fringing wetlands (Keddy and Reznicek 1986, Poiani and Johnson 1993, Winter and Rosenberry 1998). In the Ridge and Valley of Pennsylvania, water regimes were characterized in headwater floodplain

wetlands as percent time inundated, saturated, or dry; maximum, minimum, and median water depths; and the number of flooding events in three duration classes (Ryan 2005). A study of forested wetlands emphasized the importance of characterizing the timing of flooding (Toner and Keddy 1997).

Wetland classification is another approach used to bridge the gap between hydrologically-characterized and unstudied wetlands. Wetlands classification systems can aid in the comparison between similar wetlands, the sharing of information between managers and researchers, and the mapping and inventory of wetland resources (Brinson 1993). Further, if the classification is hydrologically based it can be a helpful framework for making projections about the impacts of hydrologic changes forced by climate or land use changes.

A range of wetland classification schemes exist. One early, ecologically-driven classification system was developed in 1890 that is based on the length of hydroperiod (Shaler 1890). The first truly national classification system in the United States was developed and used by the U.S. Fish and Wildlife Service in 1953, which classified wetlands based on the length of hydroperiod and a characterization of the plant community (Shaw and Fredine 1956). However, this classification did not include permanently inundated wetlands, wetlands that do not support hydrophytes, or wetlands that do not occur in topographic low spots. In 1979, the Cowardin et al. (1979) classification scheme was developed to establish consistency for the first National Wetlands Inventory. This scheme hierarchically splits wetlands into systems, subsystems, and classes based on the associated type of water body, roughly categorizing hydroperiod, substrate, and vegetation type.

In response to the expressed need for better understanding of wetland hydrology by type, a system of classifying wetlands based on their hydrogeomorphic properties was developed (Brinson 1993). The hydrogeomorphic classification system (HGM) is also hierarchical, and classifies wetlands based on water sources, hydrodynamics, and geomorphic or landscape position. The general HGM classification scheme can further be refined by region, which has been done in many parts of the world including the Mid-Atlantic region of North America (Cole et al. 1997, Brooks et al. submitted for publication). Since the development of the Mid-Atlantic regional HGM classification scheme in the mid-1990s, studies have evaluated this scheme using water level data from multiple sites in central Pennsylvania (Cole et al. 1997, Cole and Brooks 2000). Cole et al. (1997) first used monthly water level data at 24 reference wetlands during the growing season as a test of the a priori HGM classification. This work showed clear differences between HGM types, particularly in terms of percent soil saturation in the root zone (upper 30-cm) for the riparian depression class (groundwater dominated) versus the slope and headwater floodplain classes. Follow-up studies looked at more frequent water level data (6-hour intervals) at 30 wetlands (Cole and Brooks 2000) and at multiple intervals for seven wetlands in Pennsylvania and Oregon (Shaffer et al. 2000). This study similarly found that there were differences in proportion of time of saturation in the root zone, between HGM classes and also between disturbance classes. However, Cole and Brooks (2000) recognized that the water-level metrics they used to characterize their sites did not capture the “flashiness” or rate of movement of water into or out of the root zone. This study also did not look at responsiveness of these systems to precipitation events under varying antecedent moisture regimes.

TIME SERIES ANALYSIS OVERVIEW

A suite of statistical tools built on the concept of time series analysis can be useful in characterizing the hydroperiod or flow regime of an aquatic system, because they specifically focus on characterizing dynamics of a system (Gaucherel 2002). Also, time series analysis has been used to understand the tease apart the influence of climatic variables (Hanson et al. 2004, Kang and Lin 2007) and anthropogenic alterations (Jennings and Jarnagin 2002) on the hydrology of aquatic systems.

A time series is any dataset where the order is non-random and, typically, values are at equally spaced intervals (StatSoft 2006). Fundamentally, time series analysis is about recognizing either univariate or multivariate pattern, and separating out noise from pattern in time series data (Ostrom 1990). Pattern in time series takes two main forms: trend (or general structure) and seasonality (periodic structure within the time period represented) (StatSoft 2006). The evaluation of time series data can take many forms, ranging from simple exploratory and descriptive techniques to much more complex univariate or multivariate modeling.

Some of the commonly used basic exploratory techniques include simple time series plots and basic descriptive statistics by some relevant time period (annual, seasonal, etc.). Time series plots help to formulate understanding of the structure of the data and to formulate basic hypotheses. Examining the autocorrelation structure of the data is a fairly simple way to look at the seasonality of a dataset. Autocorrelation looks for correlations between values in the same time series over a series of time intervals or lags (Ostrom 1990). Another fairly simple and commonly used approach to time series analysis is to separate the data into relevant time periods and perform basic statistical

comparisons between the time periods. Changes in stream flow and precipitation with changes in impervious surface in the contributing watershed have been identified using this approach (Jennings and Jarnagin 2002). In this study, the analysis segmented the data into time periods and performed Kruskal-Walis One Way Analysis of variance on the groups. Another study looked at trends in lake and river ice off dates in the Northern Hemisphere, and they simply looked at rates of change in break up days in 100-year intervals (Magnuson et al. 2000). McCabe and Wolock (2002) performed an analysis of standard departures for the maximum, minimum, and median for flows at 400 stream gages over a 60-year period. They performed non-parametric correlations for all possible combinations of 10 years of data, and found an apparent break or step increase in the flow around 1970.

A more complex and commonly used suite of time series analysis tools is autoregressive-integrated moving average (ARIMA) (Storch and Zwiers 1999). This approach is powerful and flexible (StatSoft 2006), but it also requires that the time series meet several requirements including stationarity. A dataset is stationary if the mean is constant, it has a constant variance, and the covariance is independent of time (Gilgen 2006). There are several ways to achieve stationarity in a dataset; one of the most common is to take the first or second difference of the data (McCune and Grace 2002). This simply means taking the change (first) or the change in the change (second) in the raw values for a given time set. However, for some datasets these differences are less meaningful than the raw numbers, and looking at differences can make interpretation of results more difficult or in some cases meaningless. Interrupted time series ARIMA looks for the impacts of some change in the time series in the forms of: permanent abrupt,

permanent gradual, and abrupt temporary changes (StatSoft 2006). Many studies have used this approach to pattern detection, including a study of the cycles of drought in the Southwest (Pelletier and Turcotte 1997).

Another family of time series analysis techniques is spectral analysis. One of the first and most commonly used spectral analysis methods is single spectrum analysis or Fourier Analysis (Gilgen 2006). As the name infers, this technique looks for cyclic patterns in the time series. One of the main requirements of this technique is that the data must be continuous, and it must be stationary (StatSoft 2006). Also, as opposed to other techniques mentioned above including ARIMA, this technique looks for cycles at a variety of time scales without an a priori fixed time scale of interest (such as a season or hydrologic year) (Chatfield 2003). Fourier analysis has been used in the study of cycles in water level, stream flow, and precipitation data (Kang and Lin 2007). Wavelet analysis is another type of spectral analysis that overcomes some of the restrictions of a spectral analysis. Mainly it does not require the data to be stationary (Kang and Lin 2007). Fourier analysis not only requires a stationary signal, but also looks for linear, independent, and non-evolving signals (Labat 2005). In wavelet analysis, it looks for correlations between the time series and a family of similar curves that vary in shape or dilation through various times (Gaucherel 2002). The output can be plotted as a wavelet coefficient map that shows the strength of correlations at various times in the data set and at various dilations of the curve fitted to the data. Wavelet analysis has been used in several hydrologic studies (Gaucherel 2002, Kang and Lin 2007).

Many of the techniques of time series analysis require a complete record with no missing values. However, in many cases with hydrologic data, such as water quality

sampling (Kang and Lin 2007) or water levels (Cole et al. 1997, Cole and Brooks 2000), there are missing values. There are several ways to handle missing values in a time series. One approach is to use a median value as the test statistic (Cole and Brooks 2000) or in place of a mean in a moving window analysis (StatSoft 2006). This approach is less sensitive to outliers, however it may create less smooth curves and is not amenable to weighting. Another approach is to composite series from multiple sites to cover gaps in the series; this approach was used in a study of water levels in the arid Southwest of the United States (Hanson et al. 2004). Other approaches are to use filters, such as transfer-function noise and Kalman filter (Berendrecht et al. 2003, Yi and Lee 2004) or the weighted Z transform (Kang and Lin 2007). Other studies employ Artificial Neural Networking to fill gaps in series (Reusch and Alley 2002, 2004).

RESEARCH PURPOSE

The purpose of this research is to explore: the dynamics of headwater riparian wetland water regimes and the relationships between climatic variables and the wetland water levels. This exploration was done on a subset of the monitored wetlands that have several years of continuous records spanning both wet and dry periods. In addition, I looked at the possibility of separating the relative contributions of surface water flow and groundwater flow to these wetlands, and related that to their HGM classification.

Further, the hydrographs were explored for indicators of water source to the wetlands. For example, in a previous study, using cross correlation analysis abrupt changes identified in the response times of water levels to precipitation events were assumed to indicate changes in flow paths (fissure vs. matrix) (Lee et al. 2006). Additionally, distinguishing the fast and slow response of the baseflow recession can help

to determine whether the groundwater is coming from macropores or hillslopes. There has been some criticism of the assumptions and focus on pattern instead of process in using hydrographs to help to separate water sources in streams (Talaksen 1995), however, recent work using both oxygen isotopes and hydrograph separation techniques to estimate mean transit time show that for some watersheds the less costly hydrograph separation is just as effective as tracer-based δ18O models (James et al. 2007)

This research was part of a larger effort to attempt to understand the potential linear and non-linear responses of the ecological functions and services of freshwater wetlands in the Susquehanna River Basin to climate change. That study is examining at the interaction of climate change and land use change on the systems, using hydrology as the main driver of change. The overall goal is to build empirical relationships between measured wetland hydrology and ecological services, then use those relationships to forecast condition.

In the light of the goals of this overarching Susquehanna River Basin climate change study, and the general need for better characterization and understanding of wetland hydrology, the purpose of my study was to build on the existing hydrologic characterization of a set of headwater riparian wetlands studied in the Ridge and Valley Physiographic Province (Cole et al. 1997, Cole and Brooks 2000). Specifically, I sought to better characterize the dynamics of water level in a subset of wetlands in terms of the timing and magnitude of change and the relationship between water level changes and climatic drivers of precipitation and seasonal changes under varying drought regimes. The experienced range of climatic influences on the hydrologic regimes cannot be a true surrogate for projecting the influences of climate change on a system as actual scenarios

of climate change have not been experienced (Milly et al. 2008). However, characterizing the hydrologic response over a known period of record can help parameterize models used in forecasting and can be used to understand the hydrologic response of the systems to the experienced climatic variability (Poiani and Johnson 1993, Euliss et al. 2004). Further, five of the wetlands with more continuous records in both dry and wet years were evaluated in closer detail to guide interpretation for the hydrographs from the other sites.

RESEARCH QUESTIONS

The set of research questions that this research set out to address were:

- How well can changes in water level in headwater wetlands be predicted by off-site precipitation data?
- Does the timing of the summer drawdown differ between wetlands of different HGM classes and how does this vary between years of extreme wetness and dryness?

METHODS

STUDY SYSTEMS AND REGION

This study focused on headwater wetlands, because a previous study of Ridge and Valley watershed estimated that headwater **w**etlands comprise greater than 75% of the total wetland resources in the watershed (Wardrop et al. 2006). Riparia (formerly the Penn State Cooperative Wetlands Center (CWC)) has had collected water level data at several headwater wetlands found at 34 locations. The wetlands were classified into the three headwater hydrogeomorphic (HGM) subclasses: slope (n=10), riparian depression

(n=9), and headwater floodplain (n=14) (Cole and Brooks 2000) (Figure 1). The wetlands are fairly small (0.1 ha to 3 ha) and have varying degrees of human disturbance onsite and in their surrounding landscape. Based on the land use within the 1-km radius circle around the site, the sites range from high integrity to severely degraded landscape settings (Brooks et al. 2002).

All of the monitoring wells were located in the Ridge and Valley physiographic province. Though precipitation occurs evenly throughout the year, the study area experiences seasonality in the water balance. There is a water deficit in the summer months, recharge in the fall months, storage in winter into spring, and soil water utilization in the summer due to cycles in the evapotranspiration demand (Figure 2) (Waltman et al. 1997).

The periods of record vary greatly: the earliest begin in 1994 and the latest begin in 2003. Discontinuities in data were caused by a variety of factors including bears, vandals, and battery failure. Given these limitations, these data are still one of the longest and most detailed libraries of information on water levels for headwater wetlands.

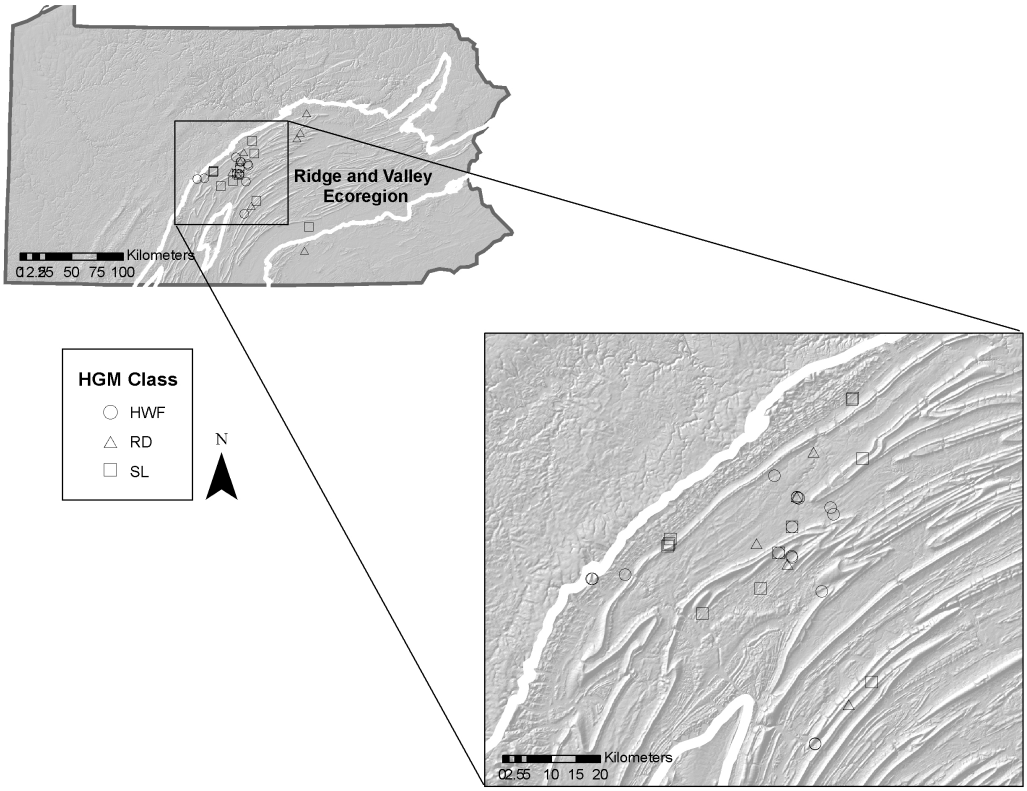


Figure 1. Location of the water level wells sites in Pennsylvania by HGM class: headwater floodplain (HWF), riparian depression (RD), and slope (SL).

STUDY SITES

The primary sites for the detailed analysis are a headwater floodplain (Laurel Run 329- from here forward Laurel Run), a riparian depression (Sand Springs), and a slope wetland (Swamp White Oak). All three of the sites have relative high ecological integrity for Pennsylvania wetlands and occur in landscape settings with low levels of human disturbance (Brooks et al. 2004). As in most of Pennsylvania, their surrounding landscapes had been deforested at least once since the late 1800's (Schein and Miller 1995).

QUALITY CONTROL OF WATER LEVEL DATA

Before the water level data was used in the analysis, it was run through a series of quality control checks and processing. The first processing step was to remove the calibration height from the water level data. The wells record water level at a calibration point which is typically at the ground level. If this calibration point is above the ground level, this height was periodically checked in the field and subtracted from the water level reading so that all of the data is in centimeters below the ground surface. Second, the reading-intervals were standardized across the sites, as water levels were originally measured at six-hour intervals and then switched to 3-hour intervals (Cole et al. 2000). The data was checked for data gaps and anomalous readings. First, the data were plotted as a time series to identify obvious anomalies. Additionally, routines were used in Excel to check for gaps or redundancies in the data. Scripts were further used to identify water level outliers if a level was beyond the measurement range of the wells (<-1.5 m or >1 m).

Specifically, dates were checked to see if they were within the period of record for the well and were successive. Redundant data were eliminated and data with impossible date stamps were either corrected if possible or changed to no data. Scripts were also used to identify "runs" in the data where the exact same or nearly the same water level was recorded multiple days in a row. This was a somewhat subjective assessment, but, for example, when all other wells at that time period were fluctuating or that most wells typically showed at least some diurnal fluctuations. These periods were removed from the record and recorded as missing data.

These sites were instrumented with one or two automatic water level recorders per site. Two types of Remote Data Systems water level recorders were used: WL40 and WL20 (Cole and Brooks 2000). The WL40 casing was a 5-cm diameter, 140 cm long PVC pipe with a 51-cm slotted, recording length. The WL20 casing was a 9-cm diameter, 168 cm long PVC pipe with a 102-cm slotted, recording length. Wells were installed into hand-augered holes, lined with sand. Soil profiles were recorded for most wells. Water levels were initially recorded at 3-hour intervals, and the reduced to 6-hour intervals to extend data storage time. Water level measurements have an accuracy of 1% of the full scale and a measurement resolution of 0.5 cm.

CLIMATIC VARIABLES

As mentioned above, the climate in the study area during the period of record varied from periods of extreme drought to extreme wetness according to the Palmer Drought Index (Table 1) (Palmer 1968). This facilitated simulations of changing hydrologic regimes. Further, studies in of wetland hydrology in the Prairie Pothole region have shown relationships between wetland hydrologic regimes and the PDHI (Winter and Rosenberry 1998).

The additional climatic drivers used in this analysis are precipitation and downstream flow. Sites did not have on-site climatic information, so precipitation and temperature data from National Weather Service cooperative stations were used (Daly et al. 2000). Well sites were within 2.7 to 21.7 km of long-term weather stations with precipitation data. Similar studies that compared climatic drivers to water level data have indicated the importance of proximate climate data, especially for comparisons over short time periods such as the responsiveness of water levels to storm events (Hunt et al. 1999,

Conly and Van der Kamp 2001). Downstream flow data was collected from U.S. Geological Survey gaging stations as available.

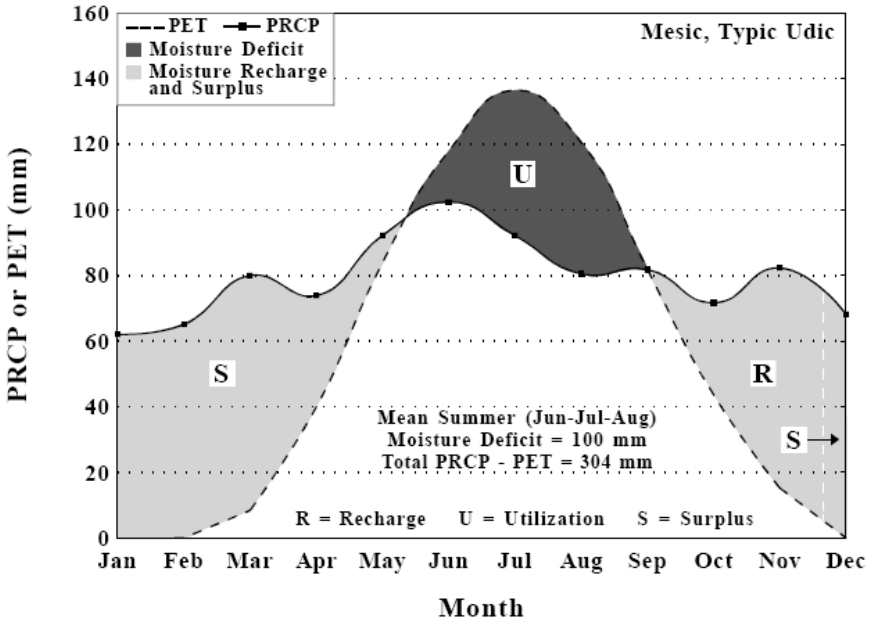


Figure 2. Moisture balance for State College, PA based upon data from 1961-1990. PET calculated using the Newhall Simulation Mode. (from Waltman et al. 1997).

Table 1. Palmer Hydrologic Drought Index for the Central Mountain Climatological Division in Pennsylvania.

Palmer Hydrologic Drought Index

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1995	3.21	2.45	1.16	0.93	0.95	-1.35	-1.97	-3.28	-3.83	-2.55	-1.55	-1.66
1996	2.41	2.28	2.19	2.26	2.13	2.00	2.15	1.53	3.95	4.54	4.68	4.95
1997	4.25	3.69	3.61	2.48	2.22	1.59	0.85	1.69	1.99	1.15	2.54	2.06
1998	2.37	2.27	1.65	2.74	2.09	2.03	1.02	-1.49	-2.22	-2.34	-3.47	-4.28
1999	-2.33	-2.64	-2.49	-2.01	-2.89	-3.35	-4.16	-4.22	-3.32	-3.52	-3.31	-3.27
2000	-3.34	-2.91	-3.20	-2.23	-2.19	-1.30	-2.04	-1.91	-1.92	-1.84	-2.20	-2.23
2001	-2.56	-2.90	-2.09	-2.48	-2.86	-2.67	-3.11	-3.41	-2.60	-2.47	-2.87	-2.77
2002	-2.85	-2.95	-2.56	-2.37	-1.07	1.52	-1.06	-1.88	-1.90	-1.35	-1.39	-0.99
2003	-0.97	-0.60	-0.78	-1.20	-0.84	-0.55	1.57	2.39	4.20	4.23	4.53	4.68
2004	4.73	4.22	3.42	3.69	3.47	2.95	4.91	5.16	7.31	6.52	5.88	5.54
2005	6.34	5.83	5.12	4.00	2.94	1.69	0.74	-2.78	-3.60	-2.68	-1.73	-1.80
2006	-0.95	-1.35	-2.08	-2.21	-2.49	-1.33	-0.86	1.88	2.63	3.71	3.46	2.61

> 4.00	Extreme wetness
3.00 to 3.99	Severe wetness
1.50 to 2.99	Mild to moderate wetness
-1.49 to 1.49	Near normal
-1.50 to -2.99	Mild to moderate drought
-3.00 to -3.99	Severe drought
<-4.00	Extreme drought

PERIOD OF RECORD

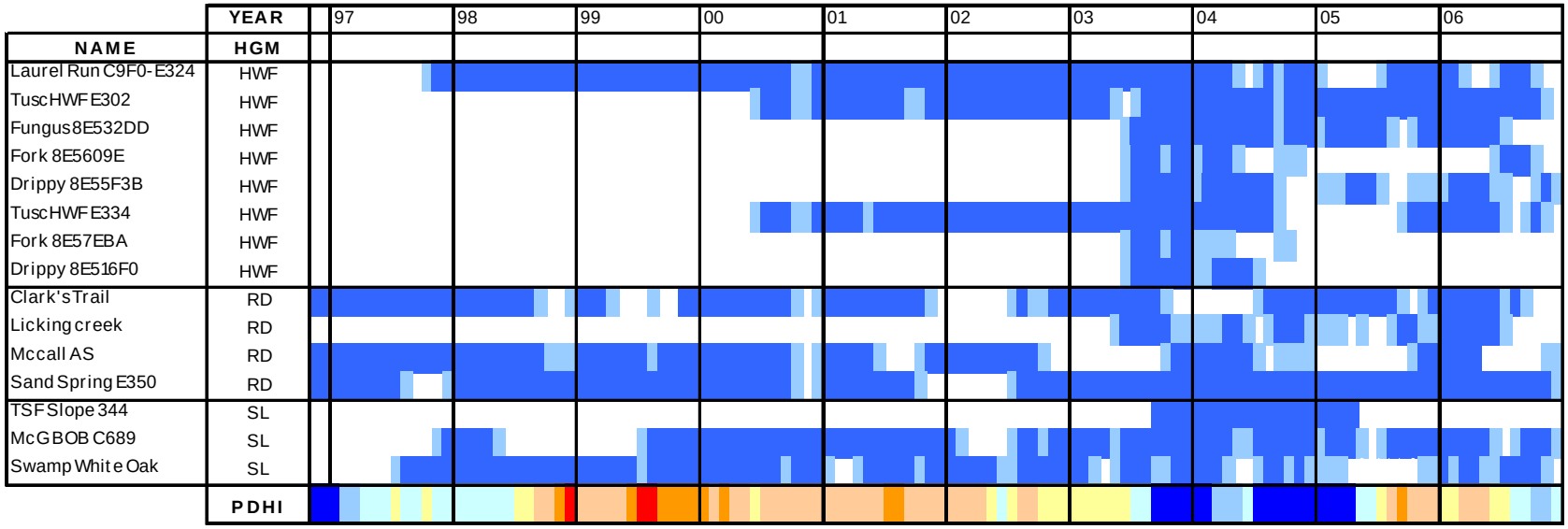
The wetland water level data span a relatively short period of time: 10 years of interrupted data in comparison to 50 to 60 years of continuous or fairly continuous records for stream gages. However, it is one of the most comprehensive water level datasets for these types of small head water wetlands. Relatively short records of water level (1-15 years of data) have been used in time series analysis studies (Kang and Lin 2007). Also, within the dataset there is a range in the completeness of the data with some records covering just over one year and others spanning the entire 10 years (Table 2).

ANALYSIS

The data were explored using cross-correlation between first difference (or daily change) in water levels and daily precipitation; moving average analysis; and local-minimum. These analyses were performed on the datasets in Excel and Minitab. Baseflow separation was estimated for the stream discharge data and for Laurel Run, one of the

most complete wetland hydrographs (Arnold et al. 1995, Sloto and Crouse 1996) using the Web-based Hydrograph Analysis Tool (Lim et al. 2005).

Table 2. Period of record for available water level data for headwater riparian wetlands in the least disturbed class (Brooks et al. 2004). Showing months with complete, partial, or no data (dark to light).



RESULTS

CROSS-CORRELATION WITH PRECIPITATION

Cross correlations were generated for these low disturbance sites with complete records for a moderate to very wet year (2003) and a dry year (1999). The climatic conditions for 2003, based on the PDHI (Table 1), were slightly drier than average in the spring and early summer and severely to extremely wet in the later summer and fall. The conditions in 1999 were dry throughout the year with an extremely dry summer. Cumulative precipitation for 1999 was less than 100 cm while 2003 had approximately 140 cm (Figure 3). The distribution throughout the year was fairly even or linear, with only difference being an increase in the second half of 2003.

The data were examined seasonally. Some sites only had complete records for some of the seasons (Table 3). Seasons were simply defined by three month blocks: spring (March, April, May), summer (June, July, August), fall (Sept, October, November), and winter (December of the previous year, January, February) corresponding to typical temperature regimes for the four seasons in this geographic region.

Table 3. Pennsylvania wetlands used in cross-correlation analysis with precipitation by HGM type, year, and season. Analyses were only performed where there was a complete dataset for that season.

		Headwater Floodplain		Riparian Depression		Slope
		Laurel Run	Tuscarora HWFP	McCall	Sand Spring	Swamp White Oak
Dry Year 1999	Winter	x	-	-	x	x
	Spring	x	-	x	x	x
	Summer	x	-	*	x	-
	Fall	x	-	x	x	x
Wet Year 2003	Winter	x	x	-	x	-
	Spring	x	x	-	x	-
	Summer	x	x	-	x	-
	Fall	x	x	-	x	-

The autocorrelation patterns for precipitation have little significant structure in any of the seasonal data for 1999 or 2003 (Figure 7). The data for 1999, in the winter there was a slight, significant autocorrelation at the 9-day time lag; the spring had no significant autocorrelation; the summer had slight significant autocorrelation at the 12- and 23-day lags; and the fall showed slight significant correlations at the 10- and 23-day lags. As for the 2003 precipitation data, there were no significant autocorrelations in any season except for a slight significant autocorrelation at the 6-day lag in winter.

Though using autocorrelated time series in cross correlations can yield spurious results, some of the methods for removal of autocorrelation can remove the actual trends of interest in a time series (Burn et al. 2004, Aziz and Burn 2006). Also, Monte Carlo simulation of errors generated from use of autocorrelated time series regression analysis indicate that correction for autocorrelation least squares regression should be corrected if it is the correlation coefficients are > or = 0.6 (Dielman and Rose 1994). Since the largest correlation coefficient in the precipitation data autocorrelation structure was ~ 0.3, the raw precipitation time series was used in the cross correlation analysis. Further, it shows

that there are relatively few multi-day events in the dataset, which would show up as 1- or 2-day lag correlations. There are no significant sub-monthly cycles to the precipitation events as the largest lag tested was for 23 days. Again, this region does not typically have high seasonal variability in precipitation.

Cross-correlation significance levels were found for the 95% confidence interval by calculating the standard error ($2/n^{-2}$) with n=number of values in the time series) and assuming that there is no correlation (Lee et al. 2006, Diggle 1990). Since these data had periods of 90-92 days, the 95% confidence significance level is R^2 of ~0.21, so any $R^2 >$ 0.2 was considered significant.

Daily Change Time Series Because the water level data has a high degree of auto-correlation, the first difference or the daily change in water level was calculated and used in the cross correlation with precipitation. These cross correlations between precipitation and change in water level reveal relationships between these variables and show temporal structure in these relationships. However, they do not necessarily relate the magnitude of changes in water level. This magnitude of change represents the flashiness of the hydrograph, and is similar to the “flashiness index” used in a previous wetland study (Fennessey et al. 2004). It is useful to inspect the times series of daily changes in water levels to see seasonal patterns in magnitude of water level changes (Figures 4-6).

The headwater floodplain, Laurel Run, is a relatively stable or non-flashy system showing the highest magnitude of change in the summer. The drier year showed fewer moderate sized changes overall and was markedly less variable in the fall than in 1999. The other headwater floodplain site, Tuscarora, showed a similar magnitude and distribution of water level change to Laurel Run, but with an overall larger magnitude of

changes especially in the fall and winter. The riparian depression, Sand Spring, showed a marked decrease in the frequency of mid-magnitude changes (approximately 5 cm) throughout the year from the dry year to the wetter year. The summer water levels were much more variable in the wet year than the dry year. During the dry year, the slope wetland site, Swamp White Oak, showed a moderate distribution of water level changes in both distribution and frequency compared to the other sites. Notably the largest changes were in the fall of the dry year.

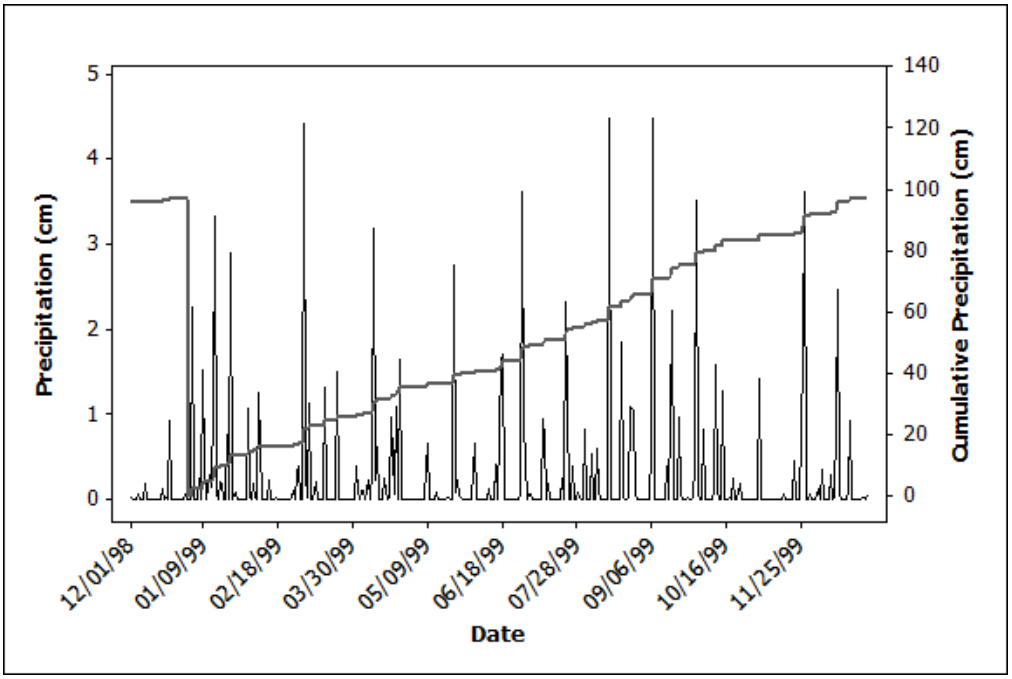
Overall, there are clearly differences in the magnitude and distribution of changes in water level between wet and dry years and between seasons for many sites.

Cross Correlations between Water Level and Precipitation Headwater floodplain, Laurel Run, showed correlations with same day precipitation and across all seasons of 1999 with high same-day lag correlations in the spring and summer (0.6 and 0.8 respectively) (Figure 9). There was also a significant 1-day lag correlation in the summer of 1999 at the 13-day lag and in the fall at the 10-day lag (Figure 10). Riparian depression, Sand Spring showed very different structures of correlation for winter and spring. It showed only a fairly strong same day correlation in the winter. The summer had a slight same day correlation, a larger negative 10-day correlation, and a slight positive correlation at 11 days. The only slope wetland, Swamp White Oak, showed little signification correlation with precipitation in the winter, but same day correlations in the spring and fall (Figure 8).

The wetter year (2003) showed very different correlative relationships across the sites. Laurel Run showed little correlation in the winter with only a slight same day and 19-day significant correlations (Figure 11). Spring also had little structure with only a

slight same day correlation, but there were strong same day correlations in the summer and fall. Tuscarora headwater floodplain showed little correlation except in the first day across all seasons (Figure 12). Sand Spring riparian depression showed same day correlations in all seasons except spring, with the only other significant correlation occurring on the 13-day lag in the winter (Figure 13). Sand Spring generally had standing water in the spring, so it is possible that the water levels were above the recording limit, so changes in water level would not be recorded and, therefore, would not be correlated with changes in precipitation.

(a)



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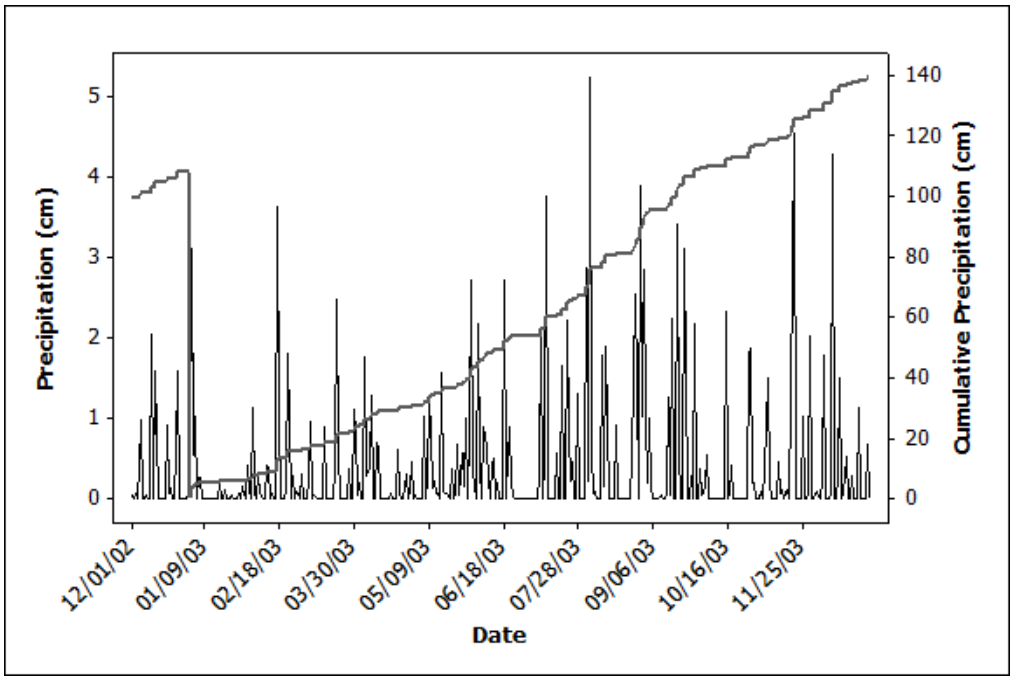
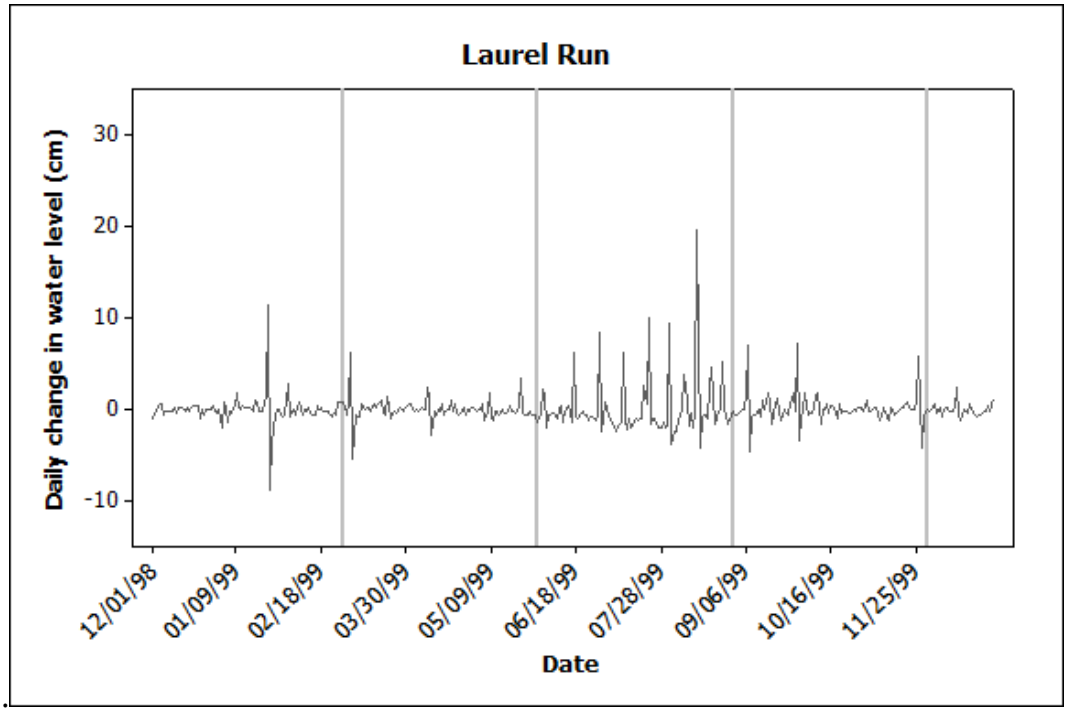
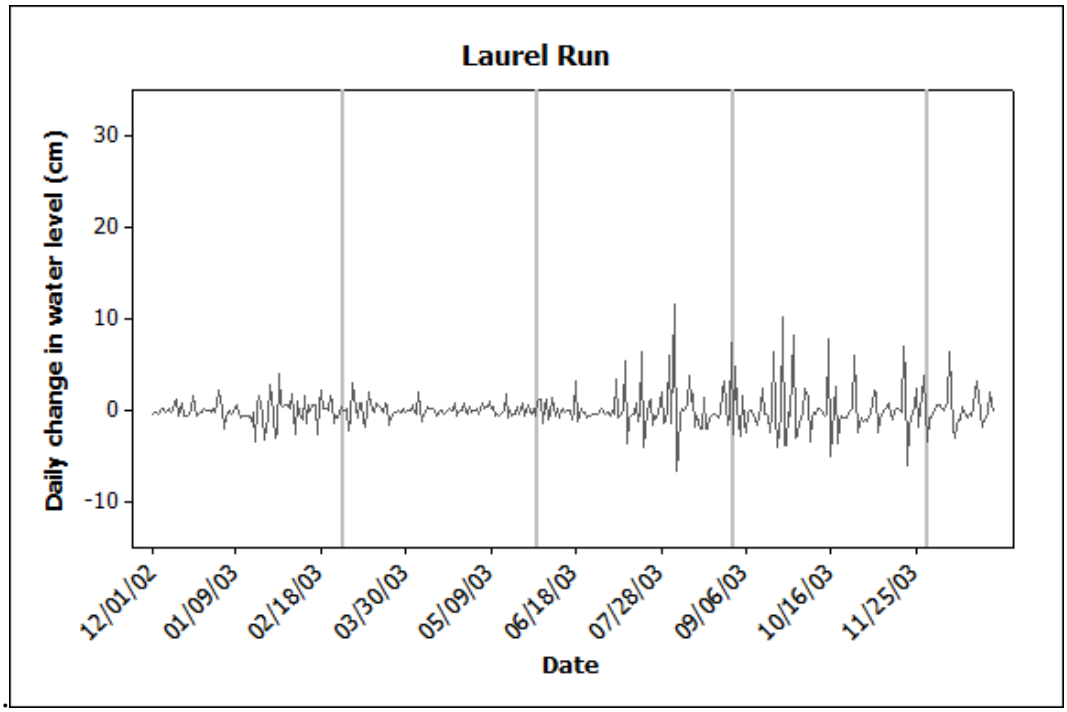


Figure 3. Daily and cumulative precipitation (cm) for State College in Central Pennsylvania during 1999 (a) and 2003 (b).



(a).



(b).

Figure 4. Daily change in water level (cm) in Laurel Run wetland in Central Pennsylvania during 1999 (a) and 2003 (b); 0=ground level.

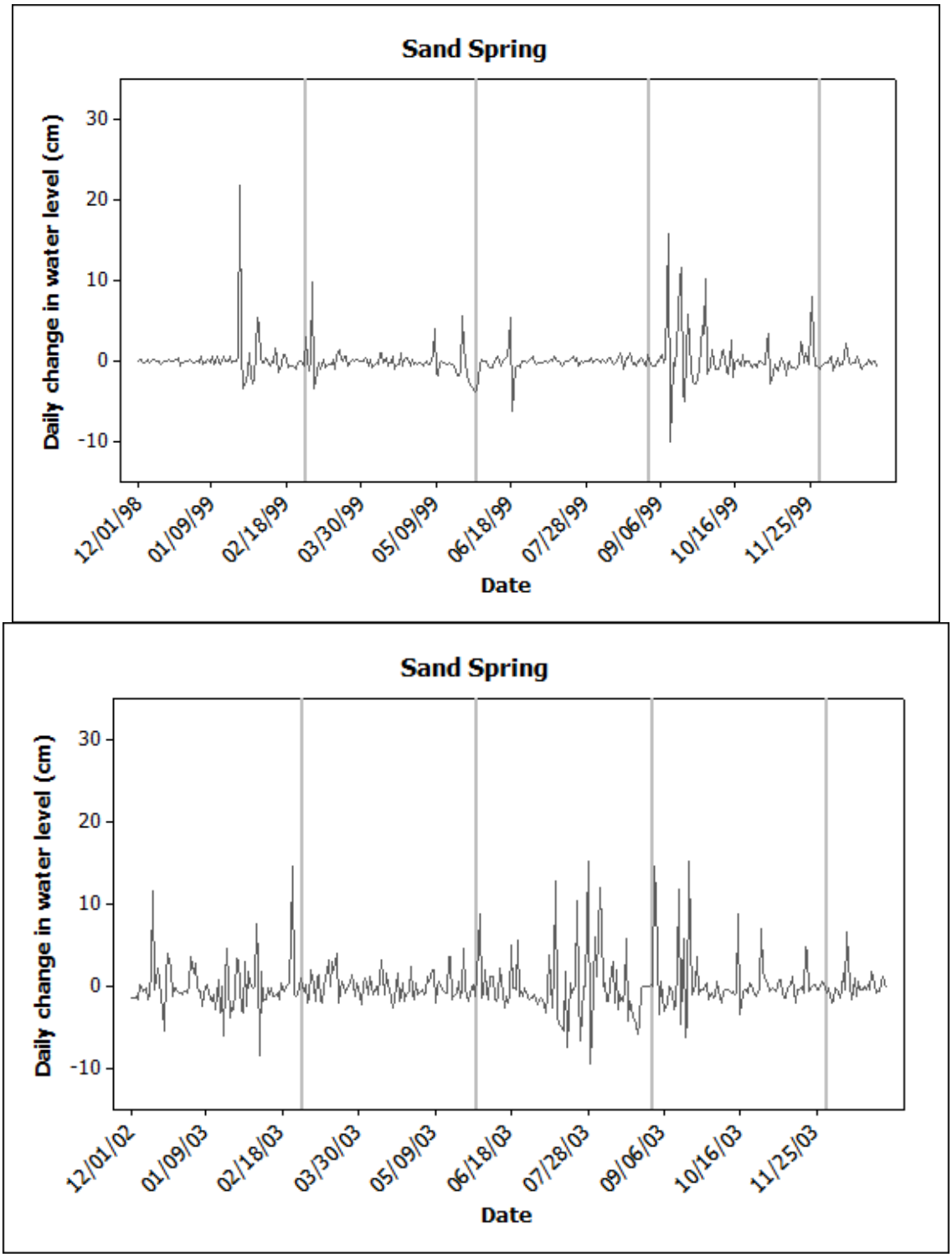
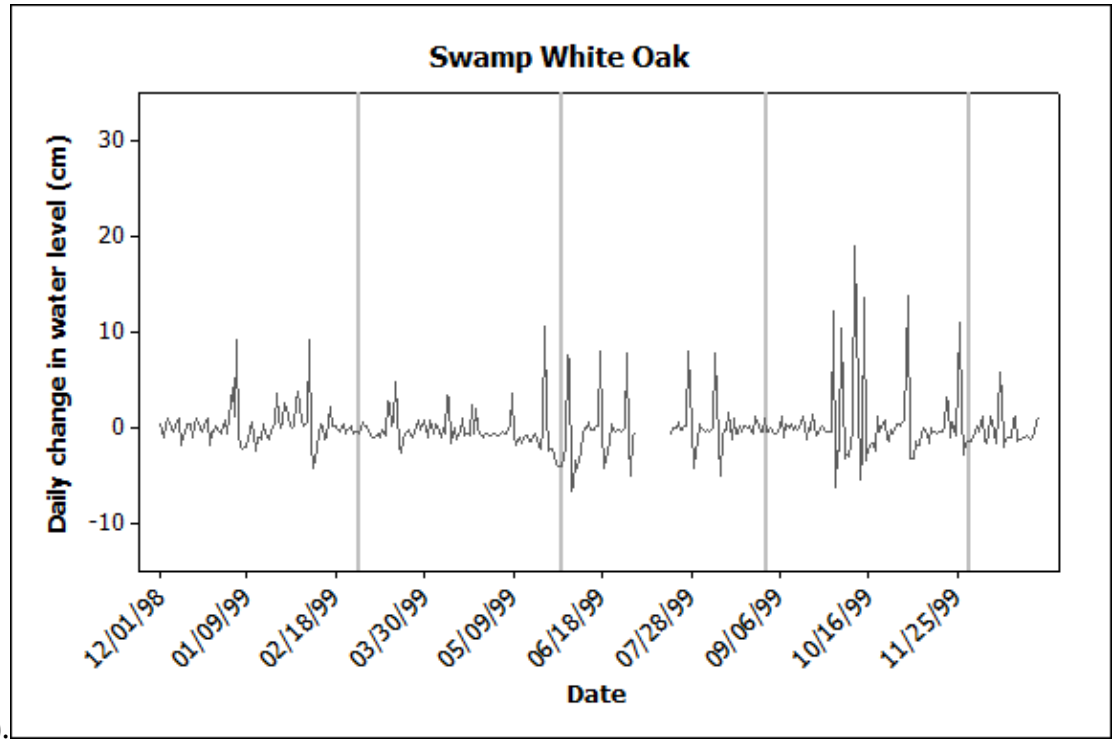


Figure 5. Daily change in water level (cm) Sand Spring wetland in Central Pennsylvania during 1999 (a) and 2003 (b); 0=ground level.

(a).



(b).

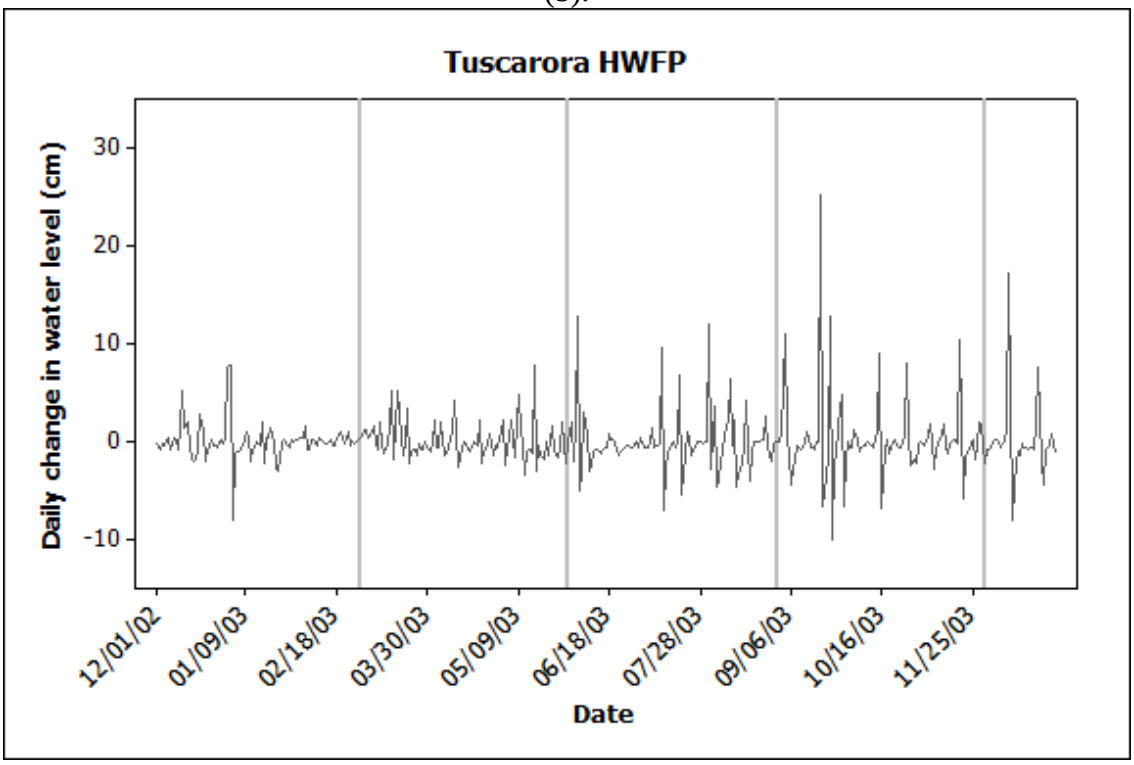


Figure 6. Daily change in water level (cm) in two slope wetlands in central Pennsylvania, (a) Swamp White Oak in1999 and (b) Tuscarora HWFP in 2003; 0 = ground level.

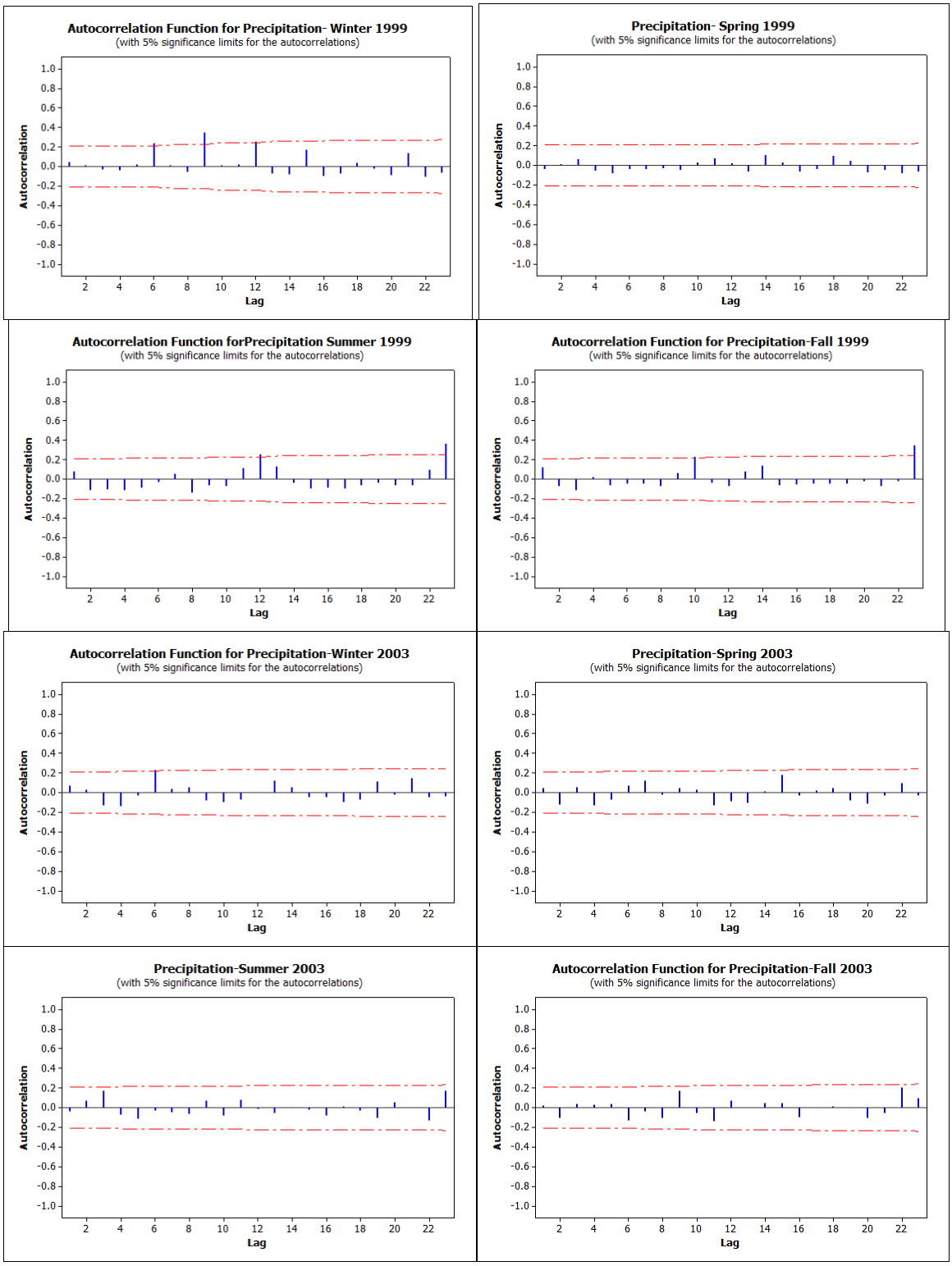


Figure 7. Autocorreleograms for precipitation by season for (a-d) 1999 and 2003 (e-h).

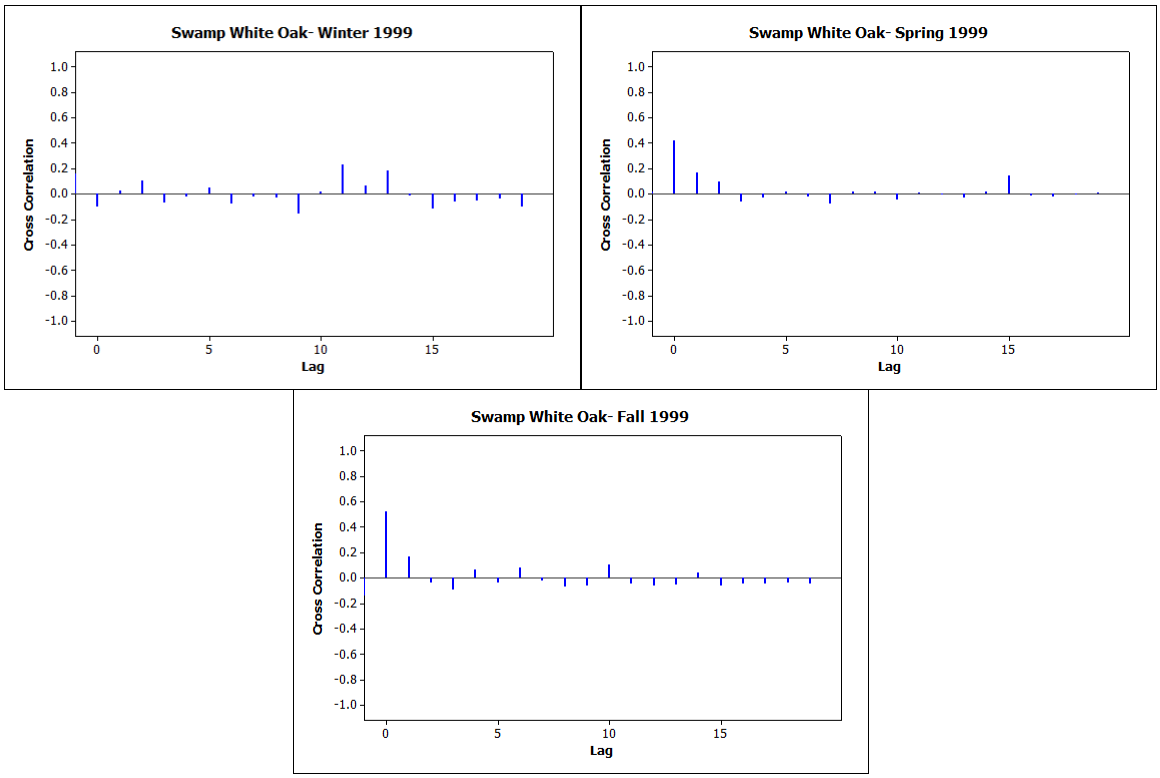


Figure 8. Cross-correlations between water levels in the Swamp White Oak wetland and precipitation in 1999 by season for (a-c) showing the correlation coefficient against the time lag (in days).

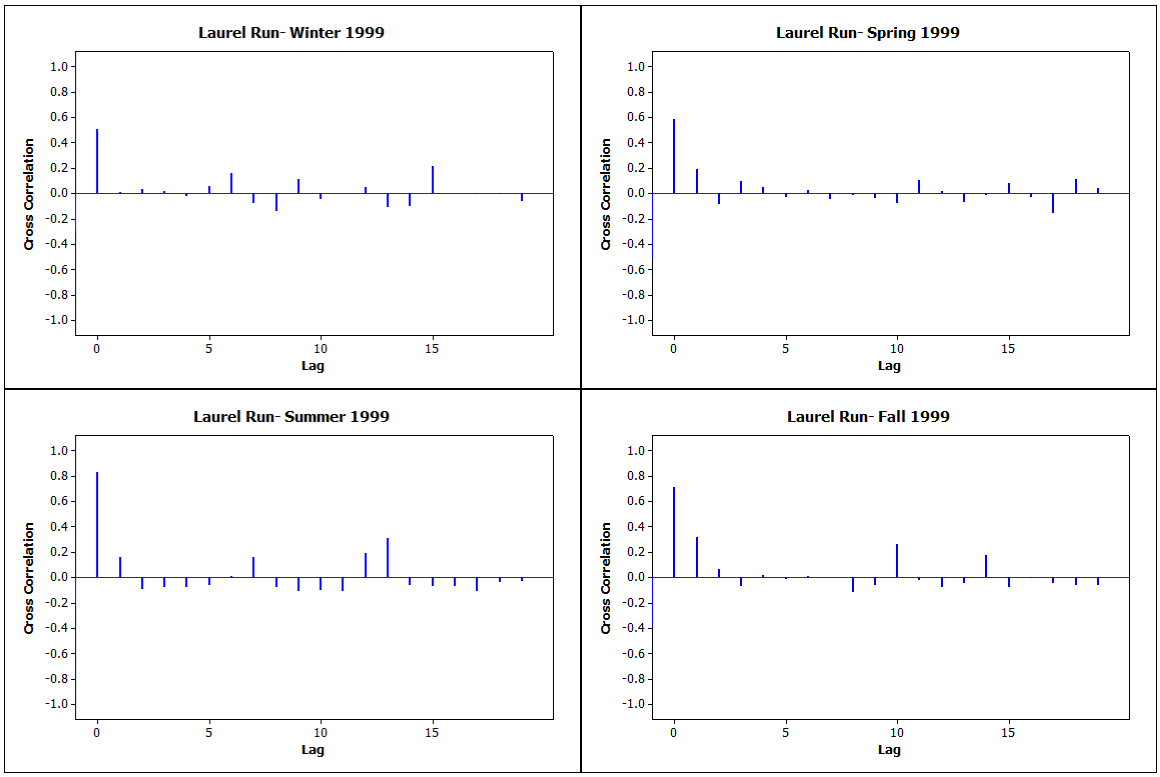


Figure 9. Cross-correlations between water levels in the Laurel Run wetland and precipitation in 1999 by season for (a-c) showing the correlation coefficient against the time lag (in days).

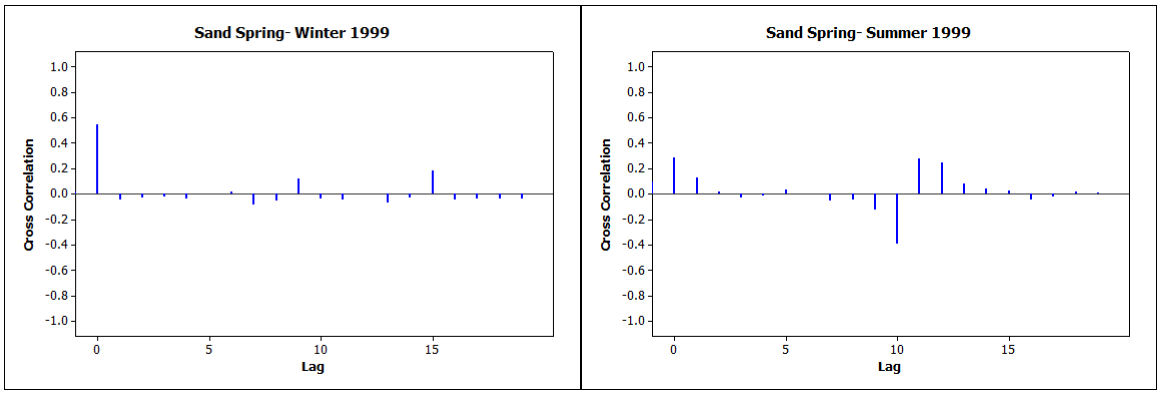


Figure 10. Cross-correlations between water levels in Sand Spring wetland and precipitation in 1999 by season for (a-c) showing the correlation coefficient against the time lag (in days).

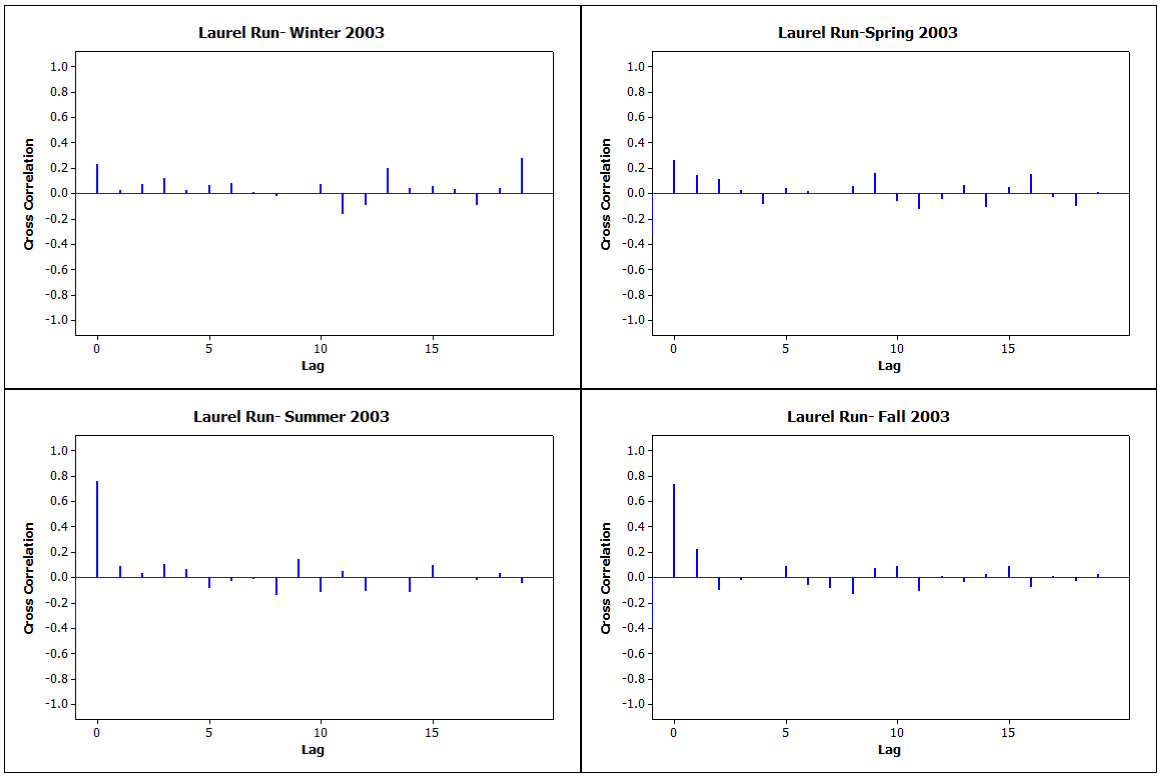


Figure 11. Cross-correlations between water levels in the Laurel Run wetland and precipitation in 2003 by season for (a-c)**.**

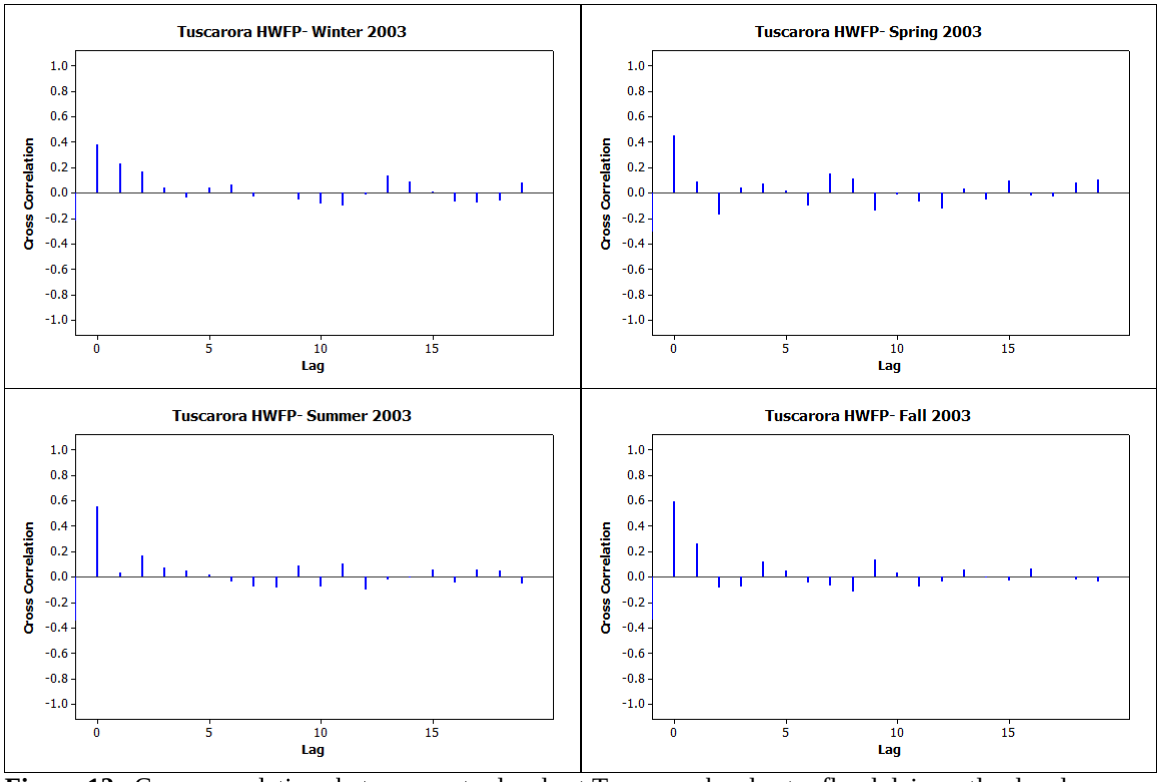


Figure 12. Cross-correlations between water levels at Tuscarora headwater floodplain wetland and precipitation in 2003 by season (a-d) showing the correlation coefficient against the time lag (in days).

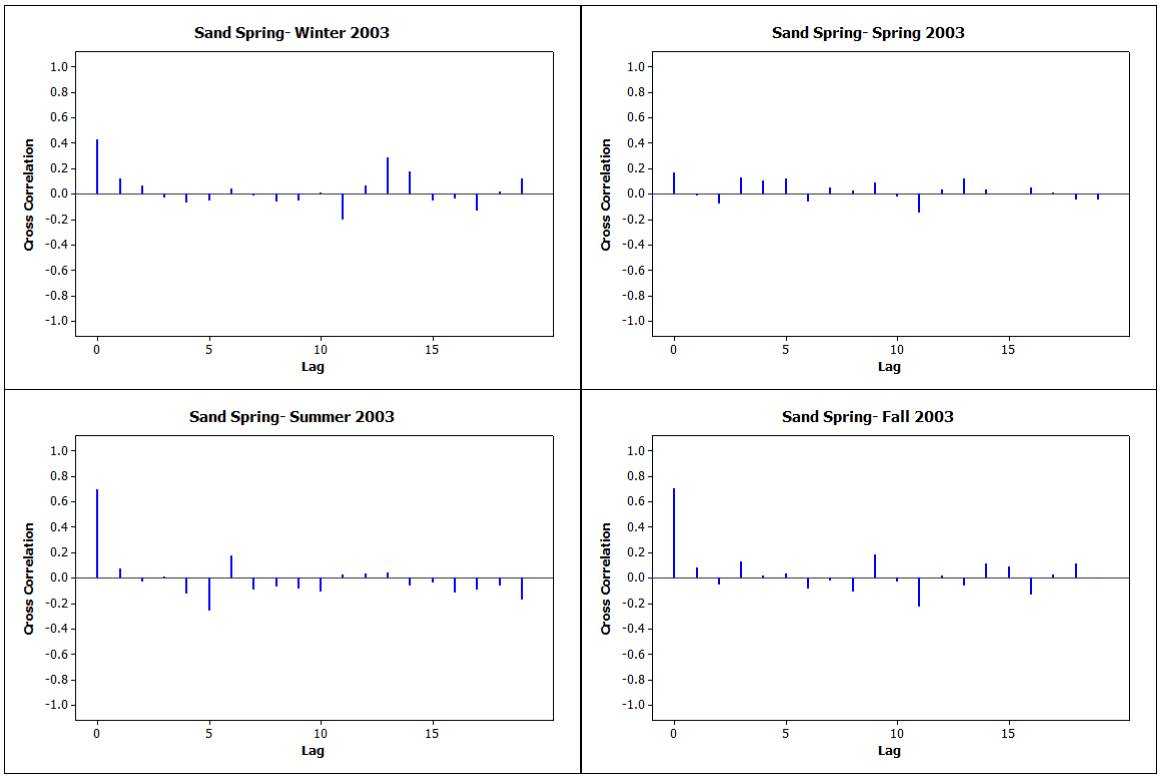


Figure 13. Cross-correlations between water levels in Sand Spring wetland and precipitation in 2003 by season (a-d) showing the correlation coefficient against the time lag (in days).

		Headwater Floodplain		Riparian Depression	Slope
		Laurel Run	Tuscarora HWFP	Sand Spring	Swamp White Oak
Dry Year 1999	Winter	**0**	no data	**0**	-
	Spring	**0**	no data	0, 11	**0**
	Summer	0, 13	no data	no data	no data
	Fall	0,1,10	no data	no data	**0**
Wet Year 2003	Winter	0,19	0,1	1, 13	no data
	Spring	0	**0**	-	no data
	Summer	**0**	**0**	0	no data
	Fall	0,1	0,1	0	no data

Figure 14. Time lags for which there were significant correlations between precipitation and change in water level by season in 1999 and 2003 Bold indicates r>0.4 and '-' indicates no significant correlation.

MOVING AVERAGE

Another approach to time series analysis of water level is the use of moving window statistics (Chatfield 2003). These allow for expression of general trends in data through time, especially in noisy datasets like shallow ground water levels. They also can be used on data sets with some missing data. This is fundamentally the opposite of analyzing magnitude and distribution of daily change, but instead looks at patterns over longer time periods. This approach allows for looking at differences through time without imposing *a priori* bins of time (like seasons) except for the selection of the window used for smoothing.

Baseflow was calculated for the stream discharge hydrograph using the local minimum method. This method mathematically attempts to separate baseflow from event flow by tracing the minimum values through time of a series (Arnold et al. 1995, Sloto and Crouse 1996). This was done with the local minimum setting in the Web-based Hydrograph Analysis Tool (Lim et al. 2005). As this method is a rough estimation of baseflow, it has been noted that it should not be used for exact estimates from specific storm events. It does, however, provide a plausible general trend in the baseflow from a simple hydrograph. The stream baseflow was used in this analysis because it is a long-term record that is an indicator of ground water conditions regionally. Also, stream hydrographs are one of the variables most frequently modeled and carefully calibrated in hydrologic models.

Moving averages were generated for the time series of the water level hydrographs. The moving averages were plotted for the sites (a headwater floodplain, a riparian depression, and a slope (Laurel Run, Sand Spring, and Swamp White Oak

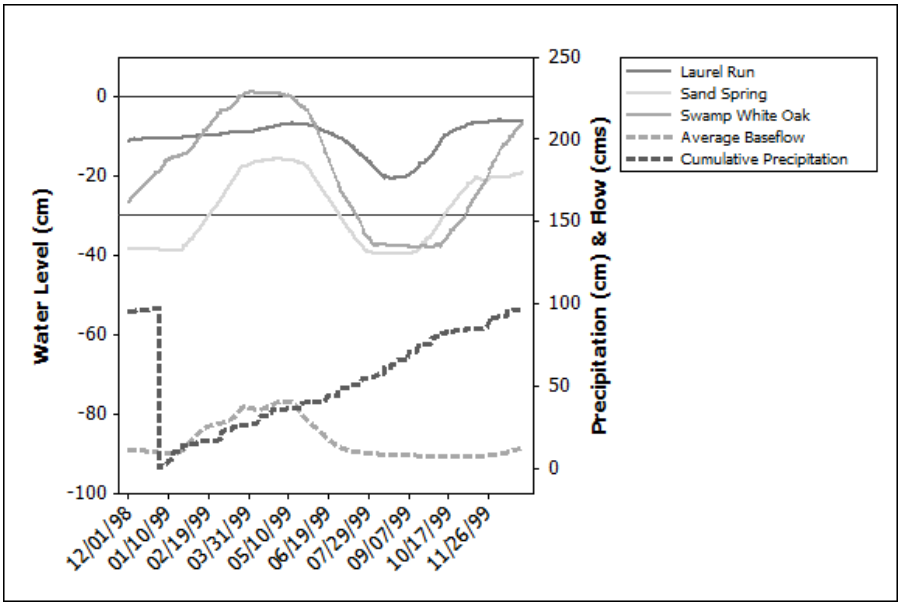
respectively) and years (1999 and 2003) used in the cross correlation analysis (Figure 16). In this analysis, the actual water levels were looked at instead of the change in water level, to look for general trends in water table height. Again, more of sites can be looked with moving window statistics than in the cross correlation analysis, because moving averages can handle gaps in a time series. Notably, Swamp White Oak was explored for both years instead of just 1999, since it was left out of the cross correlation because it has small gaps throughout 2003. The averages are calculated on a subset of the data around a single data point. In this case, a 90-day window was used, which was identified as a useful window of time through a trial-and-error, iterative process.

In the dry year, 1999, all of the wetlands showed a draw-down in the summer. The drawdown was earlier and more extreme in the Swamp White Oak and Sand Spring sites, with the smoothed water level dropping below the rooting zone (<30 cm) for most of the summer and into the fall. The lowest water levels in all sites generally followed the stream baseflow levels, but the lowest levels occurred later in the season for the wetlands. Laurel Run showed a drawdown in the summer, but the smoothed water levels did not drop out of the rooting zone.

In the wet year, 2003, there is a noticeable drawn-down of water level in the summer for all of the wetlands except for the wettest, Laurel Run. Laurel Run actually showed a slight increase during the summer. However, the smoothed water levels for the wetland never drop out of the rooting zone. With the exception of Laurel Run, the wetland water levels typically follow the water levels of downstream base flow. The lowest water levels in all sites generally followed the stream baseflow levels, but the lowest levels occurred later in the season for the wetlands. Laurel Run showed a

drawdown in the summer, but the smoothed water levels did not drop out of the rooting zone.

(a).



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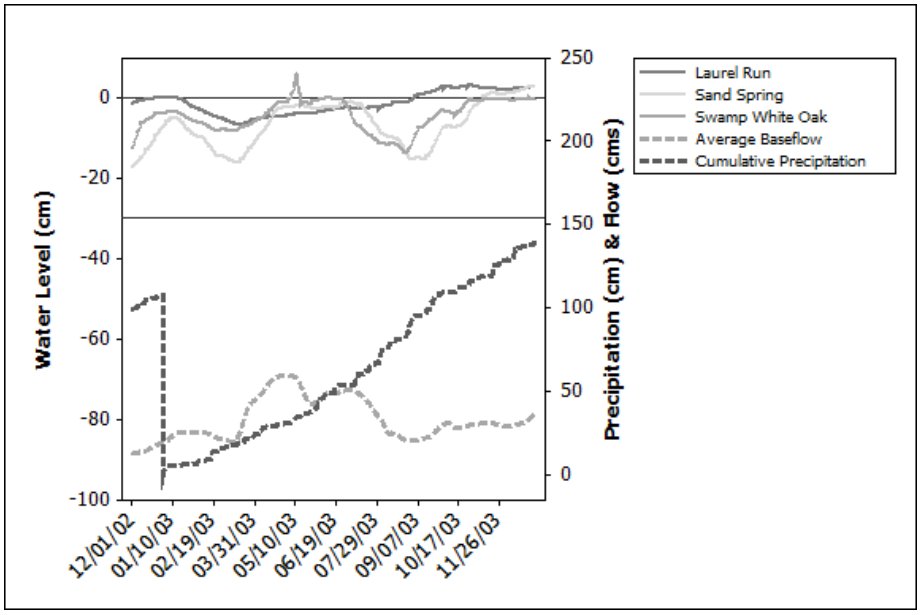
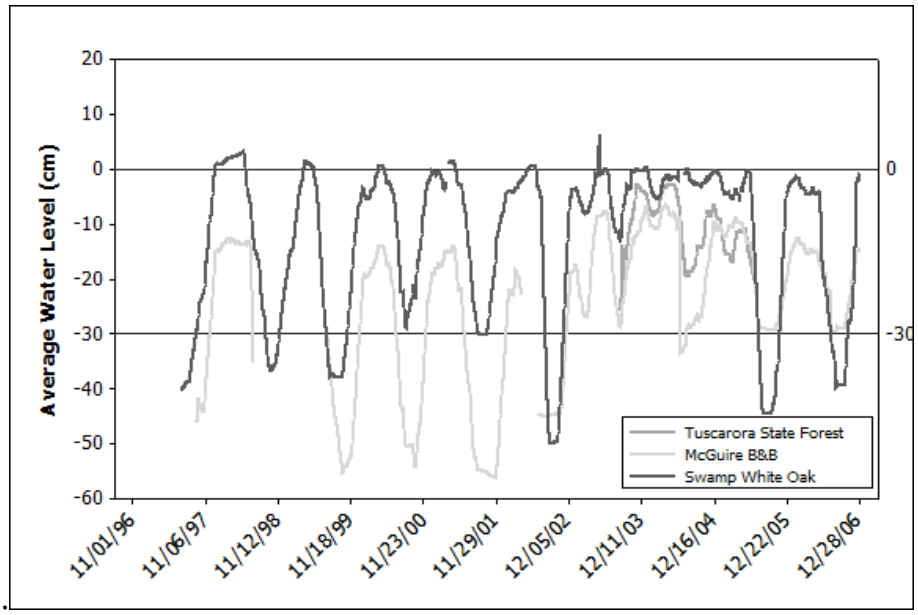
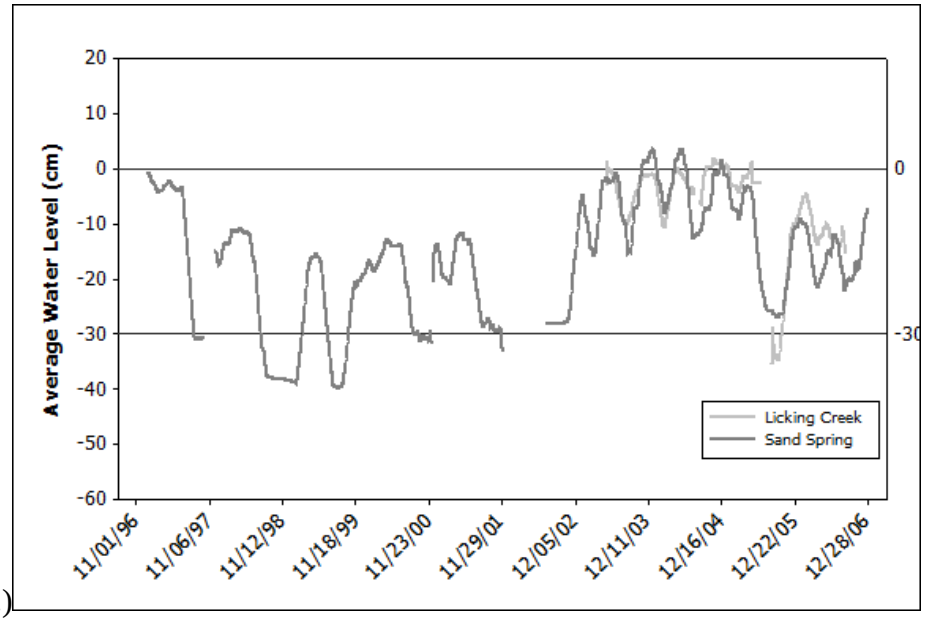


Figure 15. Cumulative precipitation (cm) and stream baseflow (cms) and 90-day moving average wetland water levels for Laurel Run, Sand Spring, and Swamp White Oak (cm) for (a) 1999 and (b) 2003.

(a).


Average Water Level (cm)
20
10
0
-10
-20
-30
-40
-50
-60
0
-30
11/01/96
11/06/97
11/12/98
11/18/99
11/23/00
11/29/01
12/05/02
12/11/03
12/16/04
12/22/05
12/28/06
Tuscarora State Forest
McGuire B&B
Swamp White Oak

(b.)


Average Water Level (cm)
20
10
0
-10
-20
-30
-40
-50
-60
0
-30
11/01/96
11/06/97
11/12/98
11/18/99
11/23/00
11/29/01
12/05/02
12/11/03
12/16/04
12/22/05
12/28/06
Licking Creek
Sand Spring

(c.)

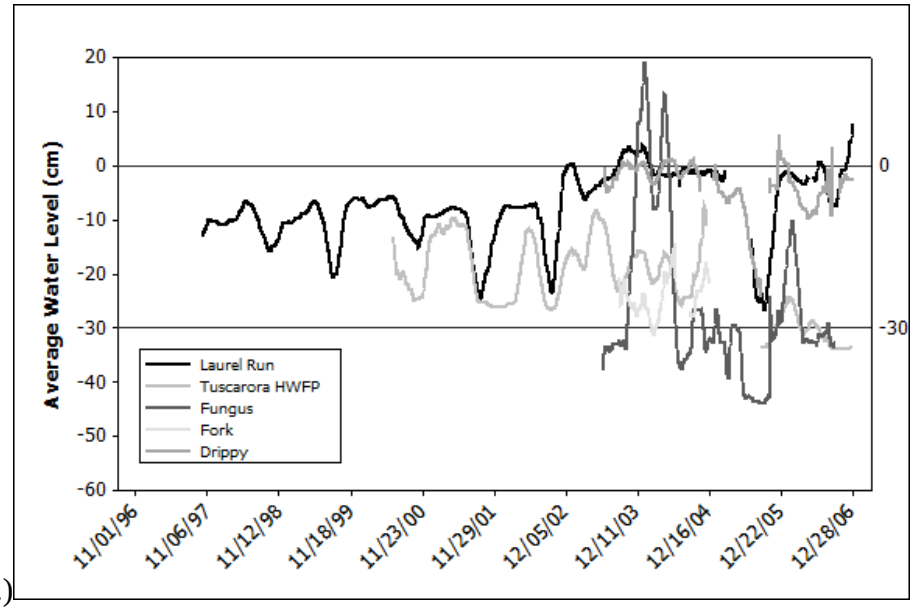


Figure 16. The 90-day moving average wetland water levels (cm) for (a) slope, (b) riparian depression, and (c) head water floodplain in Central Pennsylvania from 1996-2006.

SUMMER DRAWDOWN

Another aspect of the temporal dynamics of a wetland hydrograph is that where there is not sufficient, continuous surface or groundwater input to match this demand, wetland hydrographs in this region demonstrate a seasonal drawdown in the water level. The onset of vegetation growth (or green up) and increasing day length, intensity of solar radiation, and temperatures in a watershed changes the water balance by increasing the transpiration and evaporation demand. This is noticeable in the moving average time series shown above (Figure 16). Estimating the timing of drawdown dates and magnitude can indicate the influence of groundwater on a site. It is difficult to estimate a particular date from the raw time series, because of the flashiness of some systems in response to single rain events. The smoothed, moving average times series can also be useful in identifying this change date, but it is influenced by the peak in a flashy system. A time

series of the non-event or the slower response water levels would be useful in identifying these dates.

Slow response- or baseline-water level were calculated using the same technique for baseflow separation in the surface water hydrographs using the local minimum method in the Web-based Hydrograph Analysis Tool (Sloto and Crouse 1996 and Lim et al. 2005). This was done for the Laurel Run hydrograph, as it has a fairly long, continuous record (10/1997-11/2000). Because this approach only works with continuous datasets, the output was compared to using a moving window minimum statistic in 3-day, 5-day, and 8-day increments. All are strongly significantly correlated with the local minimum baseflow time series (R^2= 0.983, 0.976, 0.941 ($p<0.001$) respectively). Since it can be calculated on time series with missing data and is highly correlated with non-event estimates from estimates of baseflow, the 5-day minimum was used for the daily time series for all of the wells.

The local minimum time series was examined to identify the timing of the Spring draw-down for multiple years for some of the wetlands with long time series (Laurel Run, Sand Spring, and Swamp White Oak) (Figure 18). Roughly following the approach used for identifying changes in the leaf area index from remotely sensed data (Zhang et al 2003), the timing of the first decrease in slope of the trend (first sign) and the beginning of the steepest slope of drawdown was identified. Though estimating the end of the summer drawdown is useful, it is not consistently possible to extract that information from this dataset, as many of the water levels drop below the monitoring level during the summer months such that the initiation of recovery of the water table cannot be identified.

Notably, drawdown occurred in all of the wetlands except in the wettest years (2003 and 2004). The timing of the first sign of and the steepest change in drawdown were both, on average, the earliest for the slope wetland, Swamp White Oak. The magnitude of the drawdown dropped the water level below the rooting zone every year for which there were data, except for 2003 which was an extremely wet year. Laurel Run did not show drawdown in the wettest years (only a very slight drawdown in 2003 and no drawdown in 2004 and 2005). In magnitude, the drawdown only dropped below the rooting zone briefly in the driest years (1999 and 2001). Notably, 2000 which was considered a mild to moderate drought preceded by a year of drought conditions, did show some drawdown, but the level did not get below -15 cm. Sand Spring, the riparian depression, showed the most variation in drawdown in both magnitude and timing. During all years, this riparian depression had standing water in the spring with a rapid and marked drawdown to below the rooting zone. In the summer of 1997, a moderately wet summer preceded by an extremely wet winter, showed a bounce back of the minimum water levels in into the rooting zone, but not to standing water.

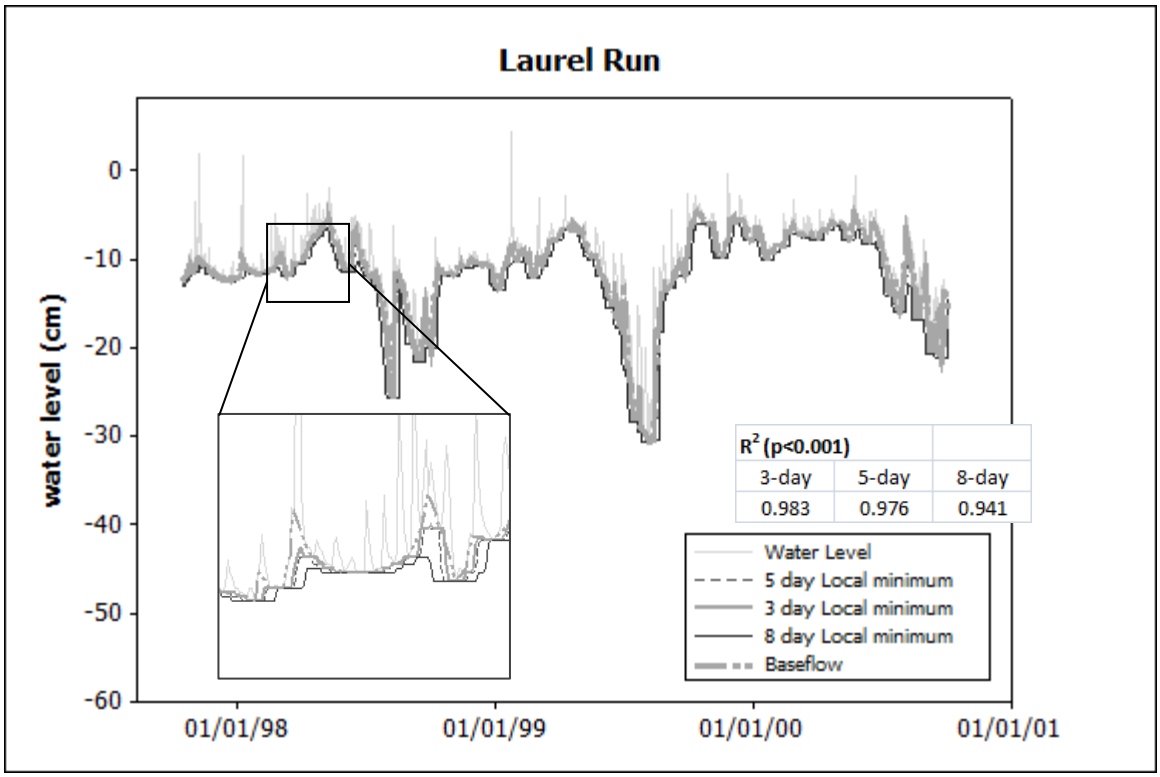


Figure 17. Cumulative precipitation (cm) and stream baseflow (cms) and 90-day moving average wetland water levels for Laurel Run, Sand Spring, and Swamp White Oak (cm) for (a) 1999 and (b) 2003.

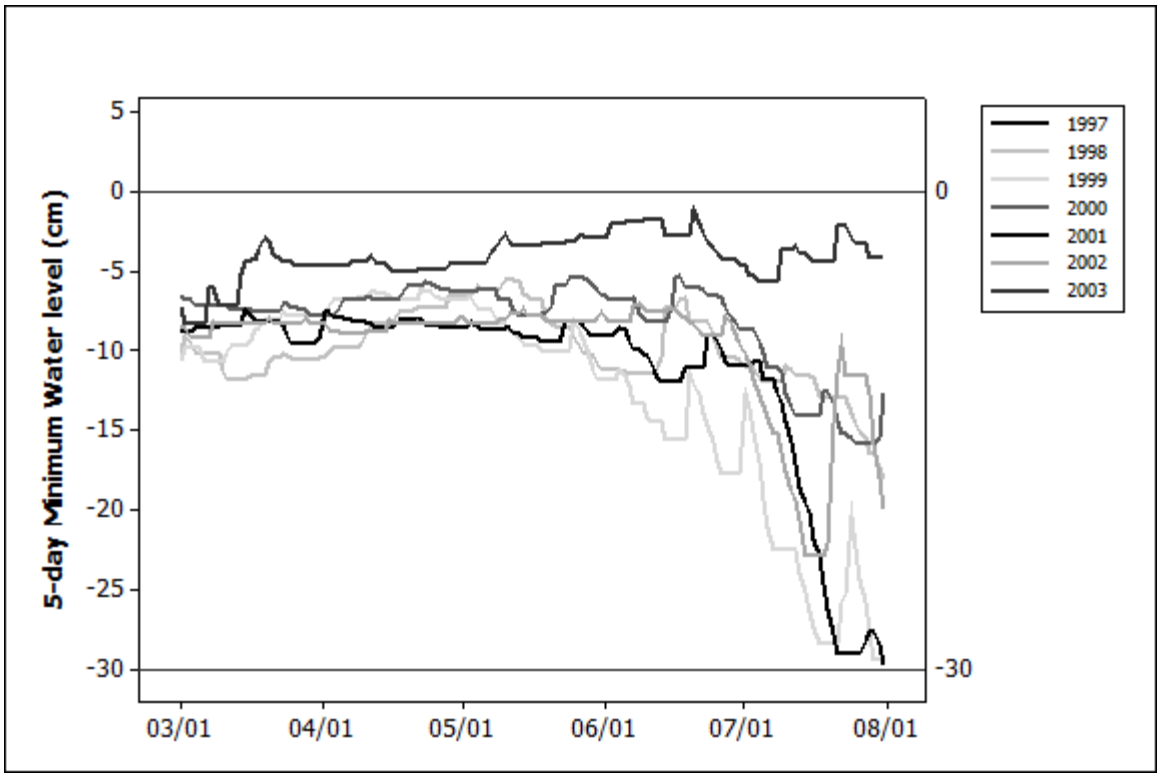


Figure 18. The 5-day minimum water level (cm) for Laurel Run from March through August for years with nearly complete data records.

Table 4. Summary of the seasonal drawdown of the water levels in Sand Spring, Laure Run, and Swamp White Oak for 1996-2006. The events identified are the first sign of change in and the beginning of the steepest change the base water level (date and day of year (DOY)) indicated by the moving window 5-day minimum water level (cm).

	Sand Spring				Laurel Run				Swamp White Oak			
Year	First Sign		Steepest Change		First Sign		Steepest Change		First Sign		Steepest Change	
	Date	DOY	Date	DOY	Date	DOY	Date	DOY	Date	DOY	Date	DOY
1997	7/1	182	7/23	204	no data		no data		no data		no data	
1998	after 6/24		after 6/24		5/16	136	7/23	204	5/10	130	5/16	136
1999	5/24	144	6/9	160	5/5	125	6/20	171	4/25	115	5/25	145
2000	6/29	181	7/24	206	5/28	149	7/6	188	4/24	115	6/7	159
2001	5/11	131	6/4	155	6/6	157	7/8	189	4/20	110	6/5	156
2002	6/16	167	7/18	199	6/15	166	6/29	180	5/26	146	6/18	169
2003	no drop		no drop		6/23	174	no drop		6/20	171	6/23	174
2004	no data		no data		no drop		no drop		no data		no data	
2005	6/21	172	7/21	202	no drop		no drop		6/5	156	6/15	166
2006	4/1	91	5/12	132	7/2	183	8/1	213	4/24	114	6/7	158
AVG	6/1	153	6/28	180	6/4	156	7/9	191	5/11	132	6/6	158
STD		33		30		21		15		23		13

DISCUSSION

CROSS-CORRELATION WITH PRECIPITATION

Cross correlations between water levels and precipitation can be insightful. Noticeably, the responses of the water level changes are primarily on the first day of a precipitation event (Figure 14). This means that typically the soils are close to being saturated and water can move quickly into the wetland. Further, some wells respond to an event over the course of several days. The secondary response may indicate water entering the site from a less permeable aquifer, while the primary response may be from flow in fissures or macropores, overbank flooding, or simply a rapidly responding local water table. Further, the first response could be "early time near stream saturation", and the second or later response of the hydrograph could be the movement of upslope groundwater to the riparian area after a slower build up of saturation in the hillslope (Duffy 1996).

Notably, Laurel Run, the most complete series analyzed, showed delayed responses during drier seasons of a dry year (summer and fall of 1999), but did not show the longer lag response in the dry seasons of the wet year (Table 3). This is a similar finding to earlier work that showed delayed responses of water level to precipitation for dry seasons (Lee et al. 2006).

Examining the cross correlation between wetland water levels from a well and local precipitation can help to infer information about water source and its water movement through the site. However, there are some caveats for this type of analysis. First, it should be noted that this is clearly inferential and does not account for other

potential drivers of response time of water level to precipitation, such as differing transmissivities between sites (Lee et al. 2006). It should also be noted that though the wells were placed within the wetlands to capture a “representative hydrograph” for each wetland (Cole and Brooks 2000), yet these wetlands can be highly heterogeneous within a single wetland and inferring the hydrologic behavior of the wetland from one well is limiting. Further, these relationships also only look at liquid precipitation and not snow or snow melt. In the winter of some years, much of the precipitation can be in the form of snow and often is stored in the watershed. This does melt, usually happening fairly gradually in the absence of precipitation, however it makes interpretation of the winter correlations a little more difficult.

MINIMUM WATER LEVELS AND DRAWDOWN

Of the wetlands evaluated, all experienced a summer drawdown in water level except for the wettest sites in the wettest years. Further, the timing of the beginning of summer drawdown varied greatly for the period of record for the three studied wetlands (slope=61 days; headwater floodplain=58 days; and riparian depression=91 days) with the slope wetland drawing down earlier on average than headwater floodplain or slope wetlands (average day of the year 132, 156, and 152 respectively). This indicates that the slope wetland is the most susceptible to the summer increase in evapotranspiration demand and maybe be fed by a shallower groundwater source.

First leaf date in central Pennsylvania has been modeled from historic lilac phenology data to be between March 30 and April 29 (Fitzjarrald et al. 2001), so most likely the onset of the drawdown and the steepest changes in the water level records are not associated with first leaf date. Due to their timing, they are more likely driven by

combination of evapotranspiration occurring when sites are fully vegetated and during the longer and warmer days of summer.

MOVING AVERAGES AND TRENDS IN WATER LEVELS

Moving average analysis can be a helpful early step in water-level analysis. It makes it easier to see general trends in the data and allows for easier recognition of unusual trends. These trends often warrant further investigation, as they may be indicative of a particular seasonal response, or sometimes spurious segments of data.

It should be noted that by definition a moving average is a smoothing function, so as it shows general trends, it illustrates conservative magnitude of events. Other analyses are needed to examine this kind of useful information, such as statistics about the length of time water level is in the root zone (Cole and Brooks 1997, Cole et al. 2000) and number of daily changes over a certain magnitude within a season (Hunt et al. 1999).

As there appears to be a fairly strong response of water levels in many of the wetland hydrographs to seasonal changes in evapotranspiration (Figure 17), especially during dry years, earlier springs projected under many climate change scenarios (Walther et al. 2002) in the absence of greater seasonal precipitation will likely cause earlier drawdowns and less saturation in the rooting zone during the growing season.

CONCLUSIONS

In general it is clear from these case studies that it is important to take into account the wetness of the year of record when performing analysis on and generalizing findings from wetland water level data. The years of extreme dryness and extreme

wetness showed markedly different hydrologic behaviors in the same wetlands in terms of presence and timing of summer drawdown and the response of water levels to precipitation.

The literature suggests that wetlands predominantly driven by surface water are more vulnerable to the impacts of climate change than those driven by groundwater (Winter and Rosenberry 1998, Conly and Van der Kamp 2001, Merkey 2006). However, for the HGM classes in this study, it is possible that these systems in the Ridge and Valley Province of the Mid-Atlantic Region are driven by shallow groundwater or unsaturated zone water that may be nearly as vulnerable to climate change or land use change on hillslopes as surface water (Grenfell et al. 2005, Lee et al. 2006).

There are many avenues of future study that could build upon the results of this work. Obviously, as more long-term datasets become available there could be more comparisons across HGM wetland types. Additionally, this study did not look at the role of landscape or site level human disturbance on the hydrologic behaviors by wetland type, and this would be a relevant and important avenue for extending this research. Moreover, the use of a coupled surface, ground water, and soil water model might help to corroborate some of the assumptions made about potential groundwater sources based on the statistical interpretations. Future studies could look at the timing of drawdown in relation to yearly evapotranspiration or the modeled long-term soil climate regime of the site (Newhall 1972, Waltman et al. 1997, Miller et al. 2006).

Overall, this research demonstrates the importance of multi-year hydrologic data in understanding wetland hydrologic behaviors. It is critical that continued effort and

support go to maintaining and expanding the monitoring of these critical, headwater wetland ecosystems.

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Chapter 5

SUMMARY AND CONCLUSIONS

SUMMARY

This work sought to enhance the incorporation of headwater riparian wetlands into the River Ecosystem Synthesis framework by exploring the four dimensional nature of their hydrologic settings and hydrologic regimes. The River Ecosystem Synthesis seeks to understand river ecosystems (including the main channel and slack waters and sub- and supra-bankfull inundation areas) as a functioning whole (Thorp et al. 2006).

The context of this research was a larger study of ecosystem services provided by wetlands of the Susquehanna River Basin. An early scaling exercise indicated that a revision of the study's scaling approach was needed. In Chapter 2, the framework of the four-dimensional RES was used to generate a revised approach to scaling for the study (Figure 1, Table 1). One aspect of the revised scaling hierarchy was the development and use of a reach-scale. Chapter 3 furthered the understanding that in river ecosystems wetlands and, in turn high, aquatic bio-complexity, occurs where there are laterally and longitudinally unconstrained reaches. The topographic characteristics that defined unconstrained reaches in this physiographic setting were identified based on the known occurrence of wetlands in the study area. Chapter 4 explored the vertical and temporal dimensions of wetlands in this physiographic setting by exploring the relationships between water level, wetland type, and seasonal fluctuations across years with a range of drought and deluge conditions using time series analysis.

Table 1. General findings of this dissertation research summarized by chapter objectives and research questions.

Chapter	Objective	Research Question
2	Articulate an ecologically-relevant spatial hierarchy for the study of wetland services in the Susquehanna River Basin	Does the existing hierarchy allow for integration across multi-disciplinary efforts and data sources?
		Are there spatial scales of study that will facilitate 'scaling up' of ecosystem studies to a small-watershed or basin-wide scale?
		Can process-based tools from geomorphology help to quantify size class breaks in contributing area to the study ecosytems?
3	Identify and map reach-level characteristics associated with riparian wetland occurrence	How accurately do modeled geomorphic variables derived from digital elevation models predict quantitative and qualitative reach-level variables?
		Can these topographic characteristics be mapped to help identify 'wetland supportive environments' or unconstrained reaches?
		Can known occurrences of wetlands be predicted from reach-level topographic variables?
4	Explore the temporal dynamics of the water levels in headwater riparian wetlands	How well can changes in water level in head water wetlands be predicted by off-site precipitation data?
		Does the timing of the summer draw down differ between wetlands of different HGM classes and how does this vary between years of extreme wetness and dryness?

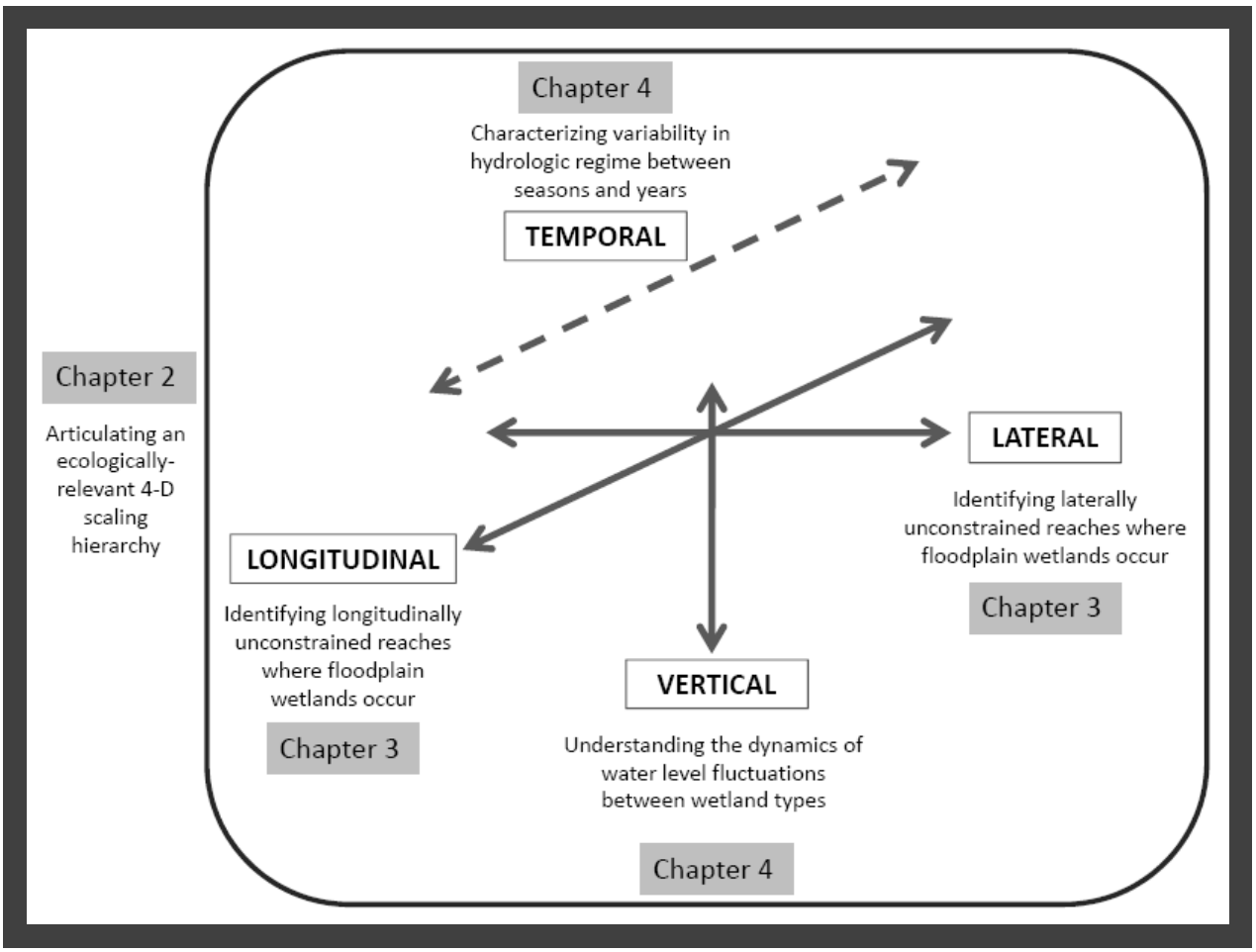

Figure 1. General findings of this dissertation research relative to the 4-dimensional River Ecosystem Synthesis Framework.

CONCLUSIONS BY CHAPTER

CHAPTER 2

In Chapter 2, the overall objective was to articulate an ecologically-relevant spatial hierarchy for the study of wetland services in the Susquehanna River Basin that incorporated the four dimensions of the River Ecosystem Synthesis Framework. One of the key research questions was, would the existing, proposed hierarchy for the study

allow for integration across disciplines and data sources in the study? By following the scaling exercise suggested for interdisciplinary studies of environmental issues (Benda et al. 2002), there was a gap identified in spatial scale between the hydrologic model output and the scale at which *in situ* wetland data, both hydrologic and biologic, had been and were proposed to be collected. Wetland data had been collected at the site- and plot-levels, while hydrologic output could be consistently generated in small-watersheds by stream segment- or reach-scales for this study.

Also for the study, there were issues with how the data could be scaled up from intensively studied sites to the small-watershed and basin-scales. One major issue in scaling up from site-level data was that there is a problem with inventory with an underrepresentation of the area of wetlands in the existing wetland inventory (Wardrop et al. 2007). This meant that using model output with the National Wetland Inventory wetlands alone would greatly under-represent findings. Also, some of the rapid assessment field work for the study locations were to be generated following a stratified, random sampling approach. Using the NWI to stratify the sampling would leave out much of the wetland resource, particularly the floodplain habitats this sampling was attempting to characterize. Borrowing primarily stream ecology literature, the reach was proposed as a base unit for analysis for intensive site-level data collection as an intermediate scale between the sub-watershed and the habitat scales.

Additionally, the habitats of the study were further classified at the macro- and micro-habitat sales. At the macro-scale the wetlands in the study were classified as mainstem or headwater floodplain wetlands using an approach from fluvial geomorphology. Traditionally, classification efforts break mainstem and headwater

floodplain wetlands using Strahler stream order, with wetlands associated with 1st through 3rd order as headwater floodplains and 4th order and larger as mainstem wetlands. However, the contributing area varies greatly within a stream order in the Susquehanna River Basin and at a landscape scale streams are not often consistently mapped at the same resolution (Clarke et al. 2008). This study borrowed the slope-area plot approach from fluvial geomorphology to attempt to form this break on process-based criteria instead of on relatively coarse predictors such as stream order. All stream reaches in the watershed were mapped as synthetic streams and slope and contributing area were calculated for each segment. The slope versus the area for each study reach was plotted and non-linearities or discontinuities were identified in the plot. These discontinuities were guides for the breaks in size classes in study area reaches based on contributing area: 0.4 to <4 km^2 are small headwaters, 4 to <40 km^2 are headwaters, and > or = 40 km^2 are mainstems. These were not only helpful in identifying the break between headwater and mainstem sites, but it allowed for the identification of an extreme headwater class (<0.4 km^2). This class of stream was very unlikely to have floodplains, because they do not have sufficient stream power and in the Ridge and Valley they were typically fairly steep where there was little lateral or longitudinal connection from the stream to the riparian area. At the micro-habitat scale, a process-based classification for stream, wetland, and floodplain habitats was developed for use in a rapid assessment. Again, few studies focus on wetlands, streams, and floodplains collectively, whereas this classification allowed for the assessment across macro-habitat types within one reach. Also, the presence or absence of such habitats acts as indicators for processes that occur in that setting.

CHAPTER 3

Chapter 3 was a further exploration of the quantification of the characteristics of the reach scale identified in Chapter 2. The overall goal was to enhance the ability to predict floodplain wetland occurrence using the knowledge that floodplains occur where there is lateral and longitudinal connectivity between the stream and riparian area, and then identify and map reach-level characteristics associated with riparian wetland occurrence. First, an approach was identified to generate synthetic streams for the study watershed and model basic topographic variables from digital elevation models. Conceptually, valley width is one of the most important variables in identifying unconstrained reaches where wetlands might occur (Ward et al. 2002), but it is one of the hardest topographic variables to quantify. There are many programs and routines that generate synthetic stream and topographic indices (Tarboton 1997, Wilson and Gallant 2000), however, identifying a tool that could calculate valley width proved to be difficult. A series of freeware, Fortran-based programs (BLDGRDS and NETTRACE) had been used and validated in a study of streams in the Coast Range of Oregon to calculate valley and stream metrics including valley width for use in a multi-disciplinary aquatic ecology study (Clarke et al. 2008). Once mapped, this chapter addressed how accurately these modeled geomorphic variables derived from digital elevation models predicted reach-level variables. Notably, modeled valley width closely predicted manually measured valley widths ($r=0.823$, $p<0.001$) and 100-year floodplain widths ($r=0.809$, $p<0.001$). Also, modeled valley slope closely predicted valley slope measured from 1:24,000 USGS topographic maps ($r=0.857$, $p<0.001$). Manually measuring reach characteristics introduces its own set of errors: for example observer judgment in determining the limits

of the valley and very widely separated iso-topo lines in low relief settings giving slope for a much larger stream segment than the target reach. But many studies blindly use GIS-measured topographic variables without any validation or verification (Clarke et al. 2008), so some positive corroboration that fairly coarse-level DEMs highly correlate with independent measures of reach characteristics is important.

The research further examined whether known occurrences of wetlands could be predicted from reach-level geomorphic variables. Using reach-level topographic variables (stream slope, valley width, specific stream power, and contributing area) as predictors and the occurrence of wetland versus non-wetland reaches as the response variables, a classification tree was generated with an overall k-folded R^2=0.600. Though this is not a terribly high R^2, what is notable was the purity of two of the nodes of the tree, which together accounted for over 70% of the stream length in the model. The most constrained reaches (valley width < 39 m and stream slope >=0.023) were 94.4% non-wetland reaches and 33.1% of the stream length evaluated by the model. Similarly, the most unconstrained reaches (valley width >= 48m) were 99.2% wetland-associated reaches and made up 37.1% of the stream length evaluated by the model. Conversely, one of the classes had fairly low purity (53.6% wetland reaches) and made up 17.7% of the stream length evaluated by the model. These were not very wide (valley width < 39 m), flat (valley slope < 0.023), headwater reaches (<10.9 km^2). These headwater reaches are particularly likely to have wetlands occurring within the reach if stochastic processes beyond the topographic setting allow for the establishment of wetlands: a large blow down or a clay lens that encourages pooling of water and, in turn, organic material. Additionally, the break in slope for longitudinally unconfined reaches at 0.023 is

corroborated by other stream classification approaches for topographic settings where streams might have associated floodplains (Rosgen 1994, Kondolf et al. 2003). Additionally, other studies have found that a shift from a supply to capacity limited segments occurs at 3-2.5% stream slope (Flores et al. 2006, Bledsoe et al. 2008) supporting that the reach classification in this dissertation is process-based.

CHAPTER 4

Chapter 4 explored the vertical and temporal dimensions within wetland habitats. Specifically, it looked at how well off-site data, particularly precipitation data, can predict on-site changes in wetland water level. And, further, it looked at how the timing of summer drawdown varied between HGM types and between years.

Precipitation was shown to be significantly correlated with the same-day water levels for most wetlands in most seasons, with correlations as high as $r = 0.8$ (Laurel Run in the summer of 1999). Further, the separated baseflow for downstream flow measurements tracked closely with absolute wetland water levels (not daily change) except for the wettest sites in the wettest years. Both of these findings are useful for regional wetland studies, because there is little existing wetland water level monitoring data. Unlike streams, which in some cases have > 100 years of continuous data, the best monitored headwater wetlands in the region have a spotty record of just over 10 years. Finding ways to use surrogates such as statistical models using off-site data, can help in the reconstruction of past and forecasting of potential wetland hydrologic regimes.

Smoothing the noisy wetland water level with a moving average filter allowed for the identification of the timing of the summer drawdown. What this work showed was

that there was a difference in the timing of drawdown between years for the period of record (range: slope=61 days; headwater floodplain=58 days; and riparian depression=91 days) with the slope wetland drawing down earlier on average than headwater floodplain or slope wetlands.

IMPORTANCE AND IMPLICATIONS

The hierarchical scaling effort described in this dissertation supports the notion that explicitly "mapping" the temporal and spatial scales of disciplinary knowledge and data in a multi-disciplinary effort to study environmental problems can be quite effective in identifying potential gaps (Benda et al. 2002).

The question of "where wetlands occur" is critically important to understanding how they function in a changing landscape and changing environment. This work helps us understand where wetlands occur within a river network, and hopefully moves us towards improving our understanding of them as a system of inter-connected Functional Process Zones. Much of the approaches used in studying Functional Process Zones require spatially-explicit data for zones or geomorphic patches. Helping to define what constitutes an unconfined reach for a headwater floodplain wetland helps to identify where wetlands are likely to occur and where might be an appropriate setting of restoration and mitigation efforts.

Since there are only short, discontinuous records of headwater wetland water levels, off-site data that has been more consistently monitored (such as precipitation and stream flow) can help to reconstruct on-site wetland hydrologic regimes. Additionally, analyzing the dynamics of hydrographs (such as the interpretation of cross-correlation

structure between water level changes and precipitation) can help to understand the potential sources of water to a wetland, while the water source has been linked to the resilience of a wetland to changes driven by climate change (Winter 2000).

Further, understanding the timing of drawdown in empirical data wetland water level data, can help us to predict changes in the timing of drawdown and water availability with changes in the hydrologic cycle predicted due to changes in climate and changes in land use.

In general, the literature about stream ecology only makes tangential reference to wetlands. This research, by characterizing a spatially-explicit reach-based scaling approach and by identifying fluvial geomorphic characteristics associated with wetland occurrence, helps to situate wetlands in the same River Ecosystem Synthesis framework as stream studies.

AVENUES FOR FUTURE RESEARCH

The work in this dissertation points to relevant future work, some of which is already under way.

CHAPTER 2

There is on-going field data collection on the rapid assessment of macro- and micro-habitats by reach types across several physiographic provinces. Field data collected in these efforts can help to validate or modify the identified cut offs between the size classes of wetland (extreme headwater, small headwater, headwater, and mainstem).

In general, the hierarchical framework can be useful for future work in studying wetlands as Functional Process Zones within the overarching river ecosystem. Though, not always spatially predictable, understanding how riparian wetlands as Functional Process Zones are distributed in a watershed will allow for analysis of intra-patch flows of energy, materials, and biota. Using this knowledge, watershed profiles could be generated as to where in a small-watershed 'wetland dependent settings' occur. This could be done using hypsometric curves.

Finally, a *post-hoc* 'disciplinary mapping' might be useful to see if given all that the study has learned and explored, what would the scaling approach be any different if the work was done again? This would allow for a way to document and pass on some of the lessons learned.

CHAPTER 3

The field work mentioned above will also contribute to the validation and refinement of the reach characterization. Again, the work is being done across several physiographic provinces. Specifically, this will give more points for building an understanding of where wetlands, and even specific wetland micro-habitats, occur and in what topographic setting.

Another future avenue of research is to use reach characterization in conjunction with output from an integrated surface-unsaturated-saturated zone hydrologic model to improve prediction of wetland occurrence. Some early efforts on this front indicate that

this might be a useful approach, for example, looking for wetlands where there is a modeled groundwater discharge during even a dry year in an unconstrained reach.

Another very logical extension of this research would be to higher-resolution base data. There are now LiDAR generated DEMs for most of Pennsylvania with approximately 1-m horizontal resolution (Delano and Braun 2006) versus 10-m resolution used in this study. Higher resolution DEMs could be useful on at least a couple of fronts: primarily they could simply help to more accurately capture topographic reach variables and they also might be used in identifying geomorphically-based reach breaks.

CHAPTER 4

The hydrologic regime analysis pointed to many further avenues of research. One obvious direction would be to do some of the same analysis (cross correlation with precipitation, timing of draw downs) across more sites and across a disturbance gradient. Conceptually, these analyses might indicate wetlands that may have lost connection with groundwater inputs due to disturbance in the adjacent stream geomorphology or to changes in hillslope hydrology due to changes in land use patterns.

Another useful avenue of research would be to couple these results with hydrologic model output. Hydrologic models can give an estimate of water source to a specific wetland, and they may or may not corroborate the timing of or implications of the drawdown analysis. Running models to estimate water source where there is empirical well data could corroborate inferences about water source to differing wetland types. Further, running ‘what if’ scenarios with land use and climate change at sites

where models can be partially calibrated might further allow for understanding of how these changes could impact wetland hydrologic regimes.

Also, this study qualitatively compared smoothed wetland hydrographs with downstream stream baseflows. Further, work could quantify these observed patterns through further cross-correlations.

Additional work could be done on interpreting the autocorrelation structure of the water level data itself. The water level data showed significant autocorrelation, therefore, the first difference had to be used in the cross correlation analysis. However, analyzing the structure of the water level autocorrelations themselves might give insight into dynamics of the hydrographs, which would be interesting to explore by season and across years with ranges PDHIs.

Another interesting extension to this work would be to look at evapotranspiration at the wetlands. With some of the newly collected data, the temporal resolution (hourly) would be appropriate to estimate ET evapotranspiration from diurnal fluctuations of water level data. Further, looking at the timing of the drawdown in conjunction with temporally explicit spatial data of leaf on/leaf off or modeled potential evapotranspiration would be helpful in further understanding the source of water to sites and in understanding the causes of seasonal drawdown.

OVERALL CONCLUSIONS

This work is one step in a process to integrate scientific understanding and management of wetlands, floodplains, and streams as one inter-connected system. The River Ecosystem Synthesis Framework encourages this multi-system approach to

understanding the 4-dimensional nature of river ecosystems through the integration of principles and approaches of fluvial geomorphology and landscape ecology into stream ecology. However, the nature and functioning of wetlands in the river ecosystem needs to be incorporated into and advanced by this framework. The work in this study helps to move this integration forward by exploring the four-dimensional nature of headwater wetlands in the river ecosystem, which can contribute to this critical, but limited literature at the nexus between streams and wetlands.

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