Hydrogeomorphic Functional Assessment Models – Riparian Depressions

Robert P. Brooks, Denice Heller Wardrop, Jennifer Masina Rubbo, Wendy M. Mahaney
Penn State Cooperative Wetlands Center, Department of Geography
302 Walker Building, University Park, PA 16802
and
Charles Andrew Cole, Center for Watershed Stewardship, Pennsylvania State University
227 East Calder Way, State College, PA 16801

INTRODUCTION

Hydrogeomorphic functional assessment models were developed for six regional wetland subclasses in the Commonwealth of Pennsylvania. The following is a description of each of the applicable functions for the specified subclass. This description is made up of six parts which are explained below.

1. Definition and applicability – Briefly defines the function and identifies which subclasses the function should be applied.
2. Rationale for selecting the function – Explains why the function is relevant to the regional subclass.
3. Characteristics and processes that influence the function – A brief literature review describing important characteristics of the function.
4. General form of the assessment model – Identifies what variables are used in the functional assessment model and describes how they are aggregated in the model equation. Additional information about the transferability of functions and variables to different ecoregions can be found in section II.B.3.b.2 (Hydrogeomorphic Model Building Process). Detailed information about the variables can be found in section II.B.3.b.2 (Hydrogeomorphic Variables: Definitions, Rationale, and Scoring).
5. Subclass rigor – Identifies major differences in the assessment model when it is used in a different subclass. Helps the user to understand the importance of correct classification of the wetland for each model.

6. FCI graphs – Each function was calculated for reference wetlands across the state. The final Functional Capacity Index for each function is plotted against our disturbance score. Although there may not be a clear relationship with disturbance, we believe these graphs are valuable in characterizing the functional capacity of the wetlands in our reference collection for use as a comparison with new sites that may be evaluated.

SUMMARY OF FUNCTIONS FOR RIPARIAN DEPRESSIONS

Hydrologic Functions

F1 – Energy Dissipation/Short term Surface Water Detention

Not applicable to Riparian Depressions

F2 – Long-term Surface Water Storage

Not applicable to Riparian Depressions

F3 – Maintain Characteristic Hydrology

$FCI = (V_{HYDROCHAR} * V_{HYDROSTRESS})^{1/2}$

F4 – Reserved for alternate hydrologic function

Biogeochemical Functions

F5 – Removal of Imported Inorganic Nitrogen:

$FCI = (V_{REDOX} + V_{BIOMASS} + V_{ORGMA})/3$

F6 – Solute Adsorption Capacity

$FCI = (V_{HYDROSTRESS}) * [(V_{ROUGH} + V_{REDOX} + V_{ORGM} + V_{TEX})/4$

F7 – Retention of Inorganic Particulates

Not applicable to Riparian Depressions

F8 – Export of Organic Carbon (dissolved and particulate)

$FCI = V_{HYDROSTRESS} * (V_{REDOX} + V_{ORGM} + V_{FWD} + (V_{CWD-BA} + V_{CWD-SZ})/2) + V_{SNAGS})/5$

<u>Biodiversity Functions</u>

F9 – Maintain Characteristic Native Plant Community Composition

$FCI = [(V_{SPPCOMP} * 0.66 + V_{REGEN} * 0.33) + V_{EXOTIC}]/2$

F10 – Maintain Characteristic Detrital Biomass

$FCI = [(V_{CWD-BA} + V_{CWD-SIZE}/2) + V_{FWD} + V_{SNAGS} + V_{ORGMA}]/4$

F11 – Vertebrate Community Structure and Composition

Use HSI models

F12 – Maintain Landscape Scale Biodiversity

$FCI = (V_{AQCON} + V_{UNDEVEL} + V_{SDI} + V_{MPS})/4$

MODEL DESCRIPTIONS

Function 1: Energy Dissipation/Short-term Surface Water Storage

Not applicable to Riparian Depressions

Function 2. Long-term Surface Water Storage

Not applicable to Riparian Depressions

Function 3. Maintain Characteristic Hydrology

<u>Definition and applicability</u>

This function is a supplemental hydrologic function for subclasses not sufficiently covered by the preceding two functions. It assesses subclasses in which the dominant source of water is not flooding related. Sites are assessed by looking at indicators of human alteration of the natural hydrologic regime of the system. This function is assessed for the following regional wetland subclasses:

a. Slopes
b. Riparian Depressions
c. Isolated Depressions
d. Fringing

Rationale for selecting the function

Hydrology is one of the defining characteristics of wetlands. However, when compared to wetlands that receive overbank flooding, the hydrology of groundwater supported wetlands is very different. As a result, the hydrologic functions the wetlands perform also change. The best way to monitor the hydrology of these systems is through a more quantitative approach, such as the collection of monitoring well data. This is not practical, however, using the rapid assessment approach outlined here. This model uses the presence of hydrologic modifications as an indicator that the hydrologic regime has been somehow altered from reference standard conditions.

Characteristics and processes that influence the function

According to HGM classification methods, wetlands that are primarily supported by groundwater are grouped separately from those that receive water from overland flow (Brinson 1993). These types of wetlands are generally grouped in the Depression and Slope classes. In Pennsylvania, depression wetlands have been further split into Isolated Depressions and Riparian Depressions. Riparian Depressions have a characteristic hydrology that is primarily supported by groundwater. The fact that Riparian Depressions are located in the riparian corridor and have an outlet to a stream separates them hydrologically from Isolated Depressions (Cole et al. 1997). Slopes are groundwater supported systems typically found on an elevational gradient, resulting in a mixture of vertical movement of ground water and horizontal movement of surface water through the system (Cole et al. 1997).

Groundwater flow wetlands are affected by human alterations occurring directly in or adjacent to the wetland and in the corresponding recharge area. Alterations such as ditches or drains, inputs from stormwater culverts, and addition of fill, have the direct effect of causing the wetland to be wetter or drier than what is characteristic of unaltered sites. Riparian Depression wetlands in Pennsylvania with moderate amounts of human disturbance were found to have greater median depth to water and a greater duration of wet conditions than pristine Riparian Depressions (Cole et al. 1997). In surrounding areas, activities that increase the rate or quantity of groundwater used, such as irrigation, industrial uses, and uses for private homes may decrease overall levels of groundwater available for wetlands (Novitzki 1989).

Isolated Depressions, many of which are commonly referred to as vernal or temporary pools, are typically supplied by precipitation runoff from a relatively small contributing area. Some receive groundwater inputs, but at rates that do not require an outlet. They often occur in clusters of depressions with members displaying variable rates of drawdown depending on their area, depth and the mixture of precipitation and groundwater. Alterations such as ditches or drains, inputs from stormwater culverts, and addition of fill, have the direct effect of causing the wetland to be wetter or drier then what is characteristic of unaltered sites Alterations of surrounding, vegetated buffers by cutting and operating heavy equipment can change the amount and flow path of waters supporting these wetlands.

General form of the assessment model

The model for the assessing the maintenance of characteristic hydrology includes the following variables:

Riparian Depressions:

$V_{HYDROCHAR}$: represents characteristic hydrology of groundwater supported systems
$V_{HYDROSTRESS}$: indicators of hydrologic modifications from stressor checklist

The general form of the assessment model is:

Riparian Depressions:

$FCI = (V_{HYDROCHAR} * V_{HYDROSTRESS})^{1/2}$

This function is split into two components. The first $V_{HYDROCHAR}$ is representative of site characteristics that would indicate the hydrologic regime. At the time this model was developed, extensive datasets to describe these characteristics were unavailable for the relevant subclasses.

Therefore, the variable was left as a placeholder, to indicate that any available information could be used here. All sites are assumed to receive a score of one for this variable unless other information is available. A typical hydrograph is provided to help define the expected hydrologic regime of each subclass (Figure 1). Deviations from this expected pattern could be used to justify assignment of a score less then one. The second variable $V_{HYDROSTRESS}$ indicates modifications to the hydrology that would cause it to deviate from reference standard conditions. The geometric mean of the two variables is calculated with both variables contributing equally to the final score.

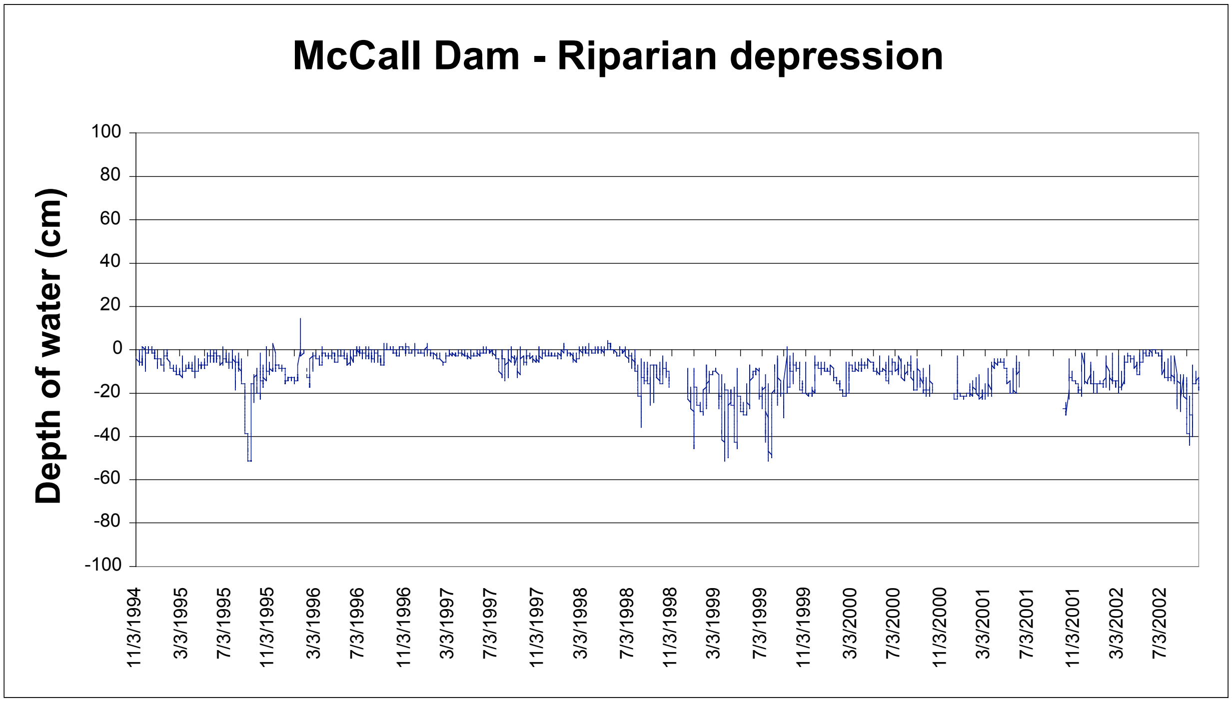


Figure 1. Typical hydrograph for HGM Wetland Subclass - Riparian depression.

Subclass rigor

This function is dependent on subclass classification since it should only be evaluated for non-riverine wetland types. Otherwise, variables that make up the function were calibrated independent of subclass.

The ability of a function to reflect disturbance is directly related to the ability of the individual variables to predict disturbance, which depends on the method of calibration used for each variable. The different methods of variable calibration and details on individual variable scores across ecoregions are discussed in Section II.B.3.b.2 (Hydrogeomorphic Model Building Process). Figures 2-4 show the relationship between the FCI and the degree of human alteration at the site, illustrating the how well the model responds to human disturbance for each ecoregion.

Figures 2-4. Relationship of Riparian Depression FCI and disturbance for sites in the Ridge and Valley, Allegheny Plateau and Glaciated Poconos (no data was available for the Piedmont). _ = Reference Standard Sites

Figure 2.

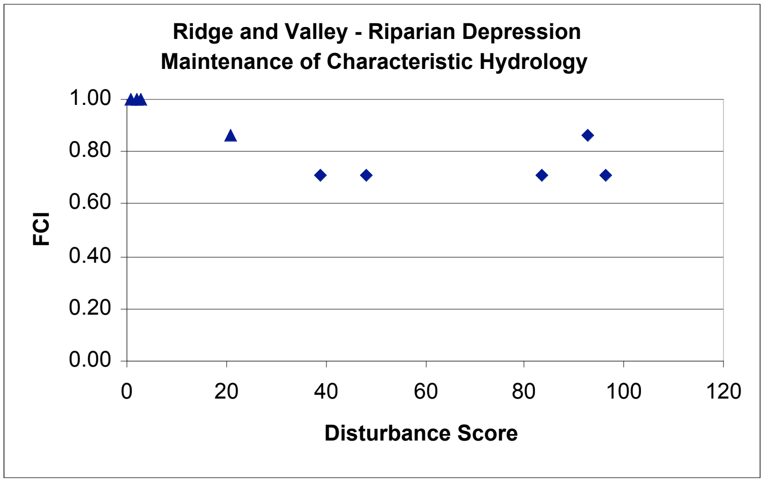


Figure 3.

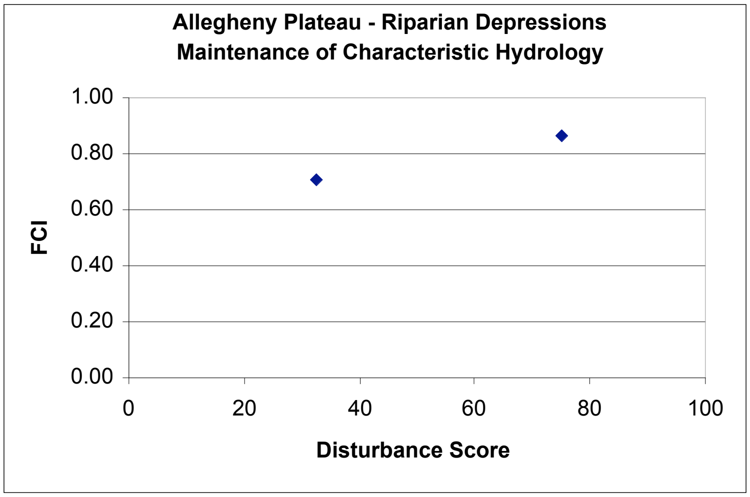
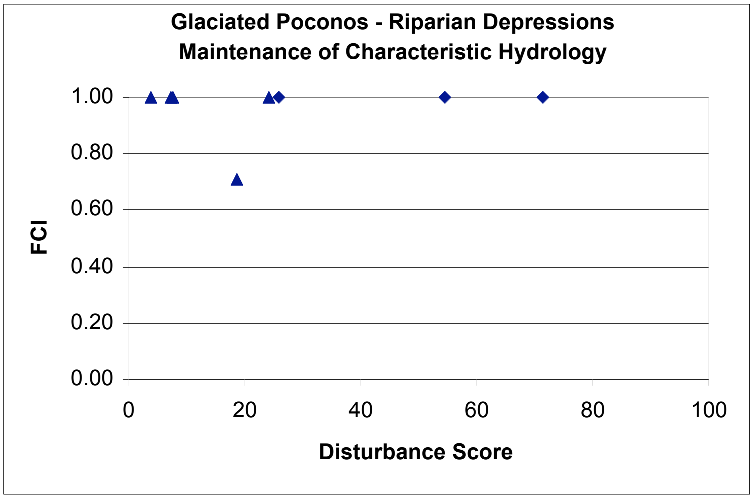


Figure 4.


Glaciated Poconos - Riparian Depressions
Maintenance of Characteristic Hydrology
FCI
1.00
0.80
0.60
0.40
0.20
0.00
0
20
40
60
80
100
Disturbance Score

Function 4. Reserved for alternate hydrologic functions

Function 5. Removal of Imported Inorganic Nitrogen

Definition and applicability

Removal of imported nitrogen is defined as the ability of a wetland to permanently remove inorganic nitrogen through chemical processes or temporarily sequester inorganic nitrogen through the plant community. The function is assessed for the following regional subclasses:

a. Riparian Depressions
b. Isolated Depressions
c. Slopes
d. Headwater Floodplains
e. Mainstem Floodplains
f. Fringing

The assessment of this function incorporates the three basic components of inorganic nitrogen removal. Permanent removal from the system is represented by the amount of organic matter in the soil and the anaerobic characteristics of the soil. Temporary removal from the system is indicated by the plant biomass at the site.

Rationale for selecting the function

Nitrogen is one of the largest non-point source pollutants of stream systems. Often, this nitrogen passes through wetlands before reaching the stream, so the ability of a wetland to remove nitrogen is extremely important to stream water quality. In many countries, agriculture is the biggest non-point source polluter, causing elevated levels of sediment, nutrients, and pesticides (Vought et al. 1994). While the application of fertilizer in general has increased since the 1960's, nitrogen fertilizers have by far been the element with the greatest increase (9 million metric tons) (Crumpton et al. 1993, Vought et al. 1994, Kadlec 2001). Studies show that as much as 50 - 90% of nitrogen fertilizer added to a cultivated crop is transported from the fields in runoff. (Crumpton et al. 1993, Seitzinger 1994). Wetlands play an important role in improving water quality due to their capacity to permanently and temporarily remove nitrogen. Denitrification is the primary process of long-term nitrogen removal from wetland systems (Davidsson and Stahl 2000). In areas impacted by agriculture, denitrification may remove a significant amount of the nitrogen transported to wetland from fields, thus preventing its movement into streams (Groffman 1994). Research has shown a 90% or more reduction in NO_3^- concentrations in water as it flows through riparian areas (Gilliam 1994).

Characteristics and processes that influence the function

The three main controls on denitrification are: oxygen levels, carbon availability and NO^{2-} supply (Groffman 1994). The majority of inorganic N present in sediments is in the form of NH_4^+ (Bowden 1987). Microbes then transform NH_4^+ to NO_3^-, which is rapidly denitrified in anaerobic zones. Along with the absence of oxygen, a nitrogen pool must be present in the system, usually found in the organic layer. Organic matter is also important in providing a substrate necessary for microbes to perform the process of denitrification. Plant uptake is an additional means of nitrogen removal from the system. Marshes show evidence that nitrogen export is small compared to uptake and internal transformations (Bowden 1987). However, this is considered only temporary removal since the nitrogen taken up by plants will eventually return to the system through leaf litter and other vegetative sources of organic matter.

Anthropogenic impacts often lead to increases in nutrient inputs to nearby wetlands, thus, altering nutrient dynamics within the wetland. Nitrogen fertilizer, one of the more common nutrient inputs in an agricultural setting, enters wetlands through groundwater and surface water runoff (Schlesinger 1997). Vought et al. (1994) found that nitrogen transport from fields was primarily in the form of NO_3^- in subsurface flows, where removal occurs mainly via denitrification. Riparian forest retained 89% of total nitrogen inputs as compared to 8% for cropland, and the nitrogen loss from the forest was primarily via groundwater (Peterjohn and Correll 1984). Nitrate was an order of magnitude higher in streams draining agricultural watersheds compared to forested and wetland landscapes (Cronan et al. 1999). Riparian wetlands can retain large amounts of nitrogen originating in upland agricultural areas. Jordan et al..

(1993) found that riparian forests retained 70-90% of the total nitrogen inputs from adjacent croplands, most of which occurred within the first 20 m from the forest-field boundary.

Channelization is a common feature associated with human activity in and around wetlands. This feature may be evident as channels in the actual wetland, or as channels leading into the wetland from the upland. Channelization of wetlands increases annual stream flow yields of nitrogen (Cooper et al. 1986). The channels funnel water rather than spreading it across the wetland (Brown 1988). Channelization also decreases the sinuosity of the river and increases channel gradient, which results in sharper pulses in flow (Brinson 1990). These impacts reduce the frequency and duration of water contact with the wetland soil, which leads to a decrease in opportunity for the wetland to remove nitrogen originating in the upland and an increase in nitrogen entering the stream.

General form of the assessment model

The model for assessing the export of imported inorganic nitrogen includes the following variables:

V_{REDOX}: presence of redoximorphic features in the upper soil profile
$V_{BIOMASS}$: estimate of amount of plant biomass
V_{ORGMA}: amount of organic matter in the upper soil profile

The general form of the assessment model is:

$$FCI = (V_{REDOX} + V_{ORGMA} + V_{BIOMASS})/3$$

The variables included in this equation estimate the controlling factors for the dominant removal mechanisms. $V_{BIOMASS}$ estimates vegetative uptake of nitrogen, and V_{SORGM} and V_{REDOX} represent conditions that affect denitrification rates. At present, there is no clear evidence that one removal mechanism is more important than the other. Thus, the variables are given equal weight in this equation. Additional information in the future may suggest that one removal mechanism dominates under different conditions, warranting a reconsideration of the equation.

Subclass rigor

This function is assessed for all subclasses using the same function equation for each. The variable $V_{BIOMASS}$ was calibrated based on a linear relationship with disturbance, so that sites with the least alteration received the highest scores. It is subclass specific since the scoring is based on the average of the reference standard sites for each subclass. V_{ORGMA} was scored differently depending on both ecoregion and subclass. For all Glaciated Poconos subclasses, as well as Fringing sites and Isolated Depressions across the state, this variable was scored based on a linear relationship with disturbance and the average of reference standard sites. It showed little response to disturbance and reference standard sites were highly variable in Mainstem Floodplains, Headwater Floodplains, Riparian Depressions and Slope subclasses in all of the other ecoregions. This resulted in a categorical scoring system that was based on deviations from the reference standard average. V_{REDOX} also showed no relationship with disturbance and was scored categorically. This variable was calibrated based on conditions that result in higher level of function. Since the variable V_{ORGMA} is both ecoregion and subclass specific, and the variable $V_{BIOMASS}$ is subclass specific, this function is highly sensitive to subclass misclassification.

The ability of a function to reflect disturbance is directly related to the ability of the individual variables to predict disturbance, which depends on the method of calibration used for each variable. The different methods of variable calibration and details on individual variable scores across ecoregions are discussed in Section II.B.3.b.2 (Hydrogeomorphic Model Building Process). Figures 5-7 show the relationship between the FCI and the degree of human alteration at the site, illustrating the how well the model responds to human disturbance for each ecoregion.

Figures 5-7. Relationship of Riparian Depression FCI and disturbance for sites in the Ridge and Valley, Allegheny Plateau and Glaciated Poconos (no data was available for the Piedmont). _ = Reference Standard Sites

Figure 5.

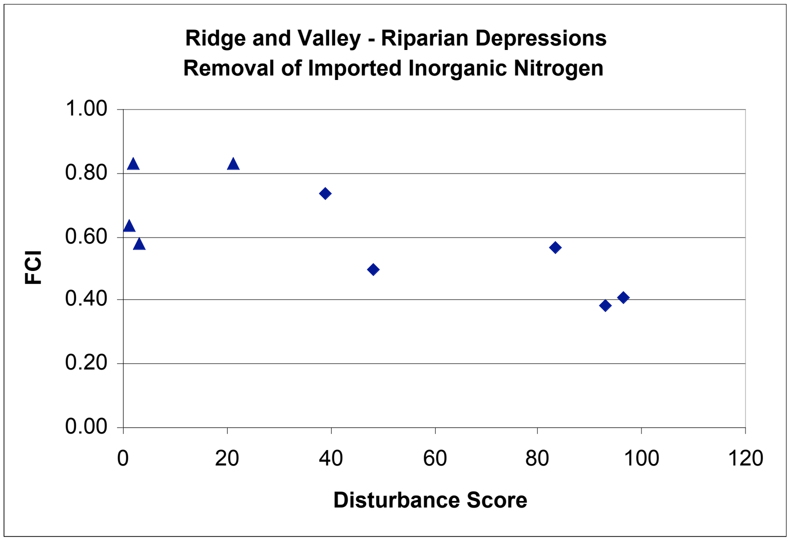


Figure 6.

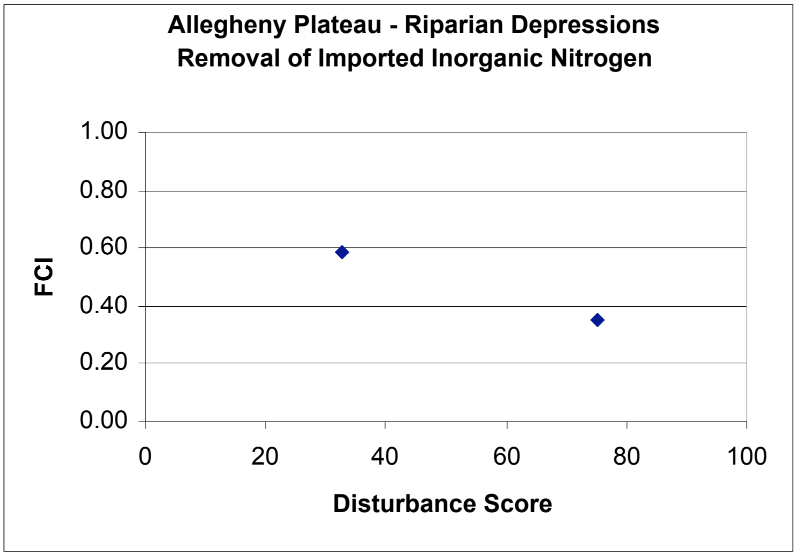
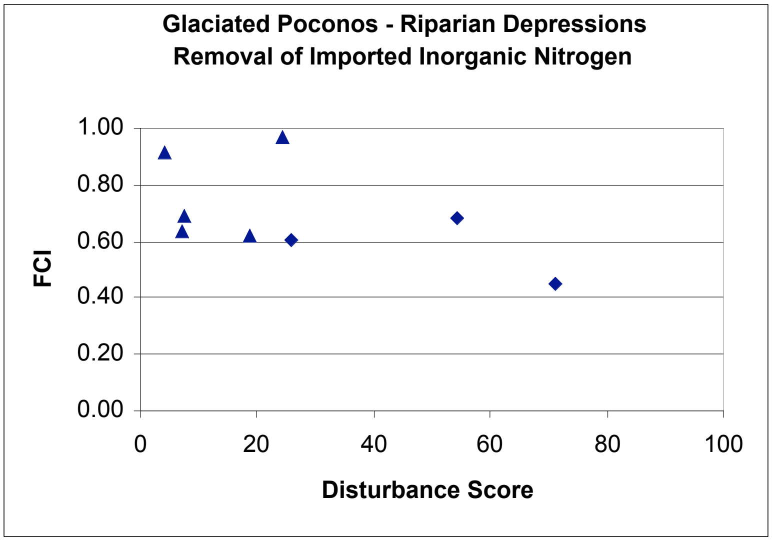


Figure 7.


Glaciated Poconos - Riparian Depressions
Removal of Imported Inorganic Nitrogen
FCI
1.00
0.80
0.60
0.40
0.20
0.00
0
20
40
60
80
100
Disturbance Score

Function 6. Solute Adsorption Capacity

Definition and applicability

This function evaluates the ability of a wetland to permanently remove and temporarily immobilize elements, such as phosphorus, metals, and other imported elements and compounds. Metals include lead, zinc, chromium, etc. Compounds include herbicides, pesticides, etc. Nitrogen, which is considered in function 5, is not included in this function. Mechanisms for retention include burial, adsorption, sedimentation, vegetation and microbial uptake, and precipitation. This function is assessed for the following regional wetland subclasses:

a. Riparian Depression
b. Isolated Depression
c. Slopes
d. Headwater Floodplains
e. Mainstem Floodplains
f. Fringing

The procedure for assessing this function is slightly different between subclasses, although all models contain similar fundamental components. The model is split into three primary components representative of hydrologic regime at the site, residence time of water at the site, and soil characteristics. It is in the first component that variables differ among subclasses due to the fact that the hydrology of subclasses is fundamentally different.

Rationale for selecting the function

As pollution due to urbanization and agriculture increases, ponds, lakes and rivers begin experiencing a decrease in water quality. Wetland systems often act as buffers to these water sources due to their ability to filter out contaminants. In many countries, agriculture is the biggest non-point source polluter, elevating levels of pesticides, herbicides and nutrients (Vought et al. 1994). Concentrations of Ca, Mg, NO_3, Cl, SO_4, and suspended solid were up to one order of magnitude higher in streams draining agricultural areas compared to forested or wetland areas

(Cronan et al. 1999). Phosphorus loads tend to increase with increasing disturbance, with the greatest loading associated with agriculture (Soranno et al. 1996). Riparian areas can remove significant amounts of imported phosphorus. For example, in a floodplain wetland in Sweden, 95% of phosphorus entering the wetland in surface runoff was removed within 16 m (Vought et al. 1994). In North Carolina, approximately 50% of the phosphorus leaving agricultural fields in runoff was removed in riparian areas (Cooper and Gilliam 1987).

Characteristics and processes that influence the function

The primary removal mechanisms for metal and phosphorus are the settling of particles out of the water column and adsorption to organic matter and clay. Phosphorus and metal removal in wetlands is controlled by several factors: adsorption to soil organic matter (SOM) and clay particles, complexation with Fe and Al, adsorption with Fe and Al, vegetation and microbial uptake. Biological uptake of phosphorus and metals is considerably smaller than the other removal mechanisms, and is relatively short-term. Chemical properties, such as pH and redox potential, greatly influence metal and phosphorus retention (Gambrell 1994, Reddy et al. 1998).

Wetlands are usually sinks for metals via three primary mechanisms: 1) precipitation of insoluble salts, 2) sorption of metal ions, and 3) vegetation uptake (Johnston et al. 1990). Detention and transformation of elements depends on SOM content, clay content and type, soil pH, and roughness (Scott et al. 1990). Chemical properties affecting metal retention include the redox potential, pH, SOM, salinity, and Al, Fe, Mn oxide concentrations (Gambrell 1994).

Phosphorus retention is influenced by plant and microbial uptake, sorption to soil particles, sedimentation (sediment-bound phosphorus), and precipitation in the water column with Ca, Al, Fe (Reddy et al. 1999). There is a dominance of geochemical sorption reactions on phosphorus, and unlike nitrogen removal, long-term phosphorus retention is predominantly geochemical rather than biological (Walbridge and Struthers 1993, Bridgham et al. 2001). Long-term removal can be through roots, buried leaves, and sediment deposition (Richardson and Craft 1993). Finer soil particles carry more phosphorus than larger particles, and slower water movement will increase particulate phosphorus settling to the soil surface (Reddy et al. 1999, Mitsch and Gosselink 2000).

The phosphorus removal ability of wetlands is assumed finite (Cooper and Gilliam 1987). After 25 years of receiving sewage effluent, a wetland in Michigan was considered phosphorus saturated (Kadlec and Bevis 1990). In contrast, phosphorus removal in an 11-year-old wastewater treatment wetland was 96%, occurring mainly via burial (Kadlec and Alvord 1989). Part of the ability to retain phosphorus is dependent upon fresh sediment entering the system. The role of incoming sediment is two-fold; deposition of sediment-bound phosphorus and deposition of fresh soil particles to bind with dissolved phosphorus (Cooper and Gilliam 1987). A large amount of the phosphorus entering and being retained in wetlands is particle-bound. More than 70% of agricultural phosphorus export is particle-bound (Vought et al. 1994). Detenbeck et al. (1993) found that much of the phosphorus transport from urban areas, as well as agricultural areas, is associated with fine particles. As a result, phosphorus retention in wetlands should increase with increasing retention time, as the settling of finer particles increases (Detenbeck et al. 1993). While the residence time in a wetland may be adequate to promote the settling of sediment, much longer times are needed for dissolved elements to settle out of the water column. The residence time of water in an urban-placed riparian depression in Minnesota was sufficient to remove 50% of the sediment-bound phosphorus, but was too short for dissolved phosphorus to be deposited (Brown 1985).

Channelization negatively impacts the ability of wetlands to remove phosphorus by funneling water rather than spreading it across the wetland. Channelization increases loading and runoff, while also decreasing load retention, resulting in increased flow yields of phosphorus into receiving waters (Cooper et al. 1986, Brown 1988).

There was not much information available on metals and other contaminants associated with disturbance, but it is likely that metal, pesticide, and herbicide levels increase with disturbance. All three are likely associated with nearby urban development. Pesticides and herbicides are also likely to be associated with agricultural production in the surrounding watershed.

General form of the assessment model

The model for assessing the solute adsorption capacity of a wetland includes the following variables:

Riparian Depressions:

$V_{HYDROSTRESS}$: indicators of hydrologic modifications at the site

V_{ROUGH}: composite score based on coarse woody debris, microtopography and vegetation
V_{REDOX}: presence of redoximorphic features in the upper soil profile
V_{MACRO}: presence of macrotopographic depressions
V_{ORGMA}: amount of organic matter in the upper soil profile
V_{TEX}: soil texture

The general form of the assessment model is:

Riparian Depressions:

$FCI = (V_{HYDROSTRESS}) * [(V_{ROUGH}+V_{REDOX})/2 + (V_{ORGMA}+V_{TEX})]/2$

The assessment models for this function have been split into three primary components representative of hydrologic regime at the site, residence time of water at the site, and soil characteristics. The first component is representative of the hydrologic regime in the system.. For Riparian Depressions it takes into account human alterations at the site that may cause the movement of water to deviate from reference standard conditions. These hydrologic variables are used as controlling factors in the Riparian Depression model and are multiplied with the remaining two components.

The second component of the model is representative of variables that indicate the residence time of water at a site. V_{ROUGH} represents a physical characteristic at the site that slows the rate of water moving through the system, therefore, increasing contact time between the soil and water. The variable V_{REDOX} denotes the level of soil saturation, indicating the duration that water and soil surface are actually in contact with each other, allowing the adsorption of solutes to actually take place. The variables in this part of the equation are considered to contribute equally to the level of function and are averaged together using the arithmetic mean.

The final component of the model takes into account soil characteristics that promote the adsorption of solutes. Greater solute adsorption occurs when there is a higher clay content in the soil, and at when organic matter is present. V_{TEX} and V_{ORGMA} are considered to contribute equally to the level of function so these two variables were combined by taking the arithmetic mean. The second and third components were combined by taking the arithmetic mean of components two and three.

Subclass rigor

While this function is assessed for all subclasses, the variables and model equations differ depending on subclass identity. Therefore, proper classification of the wetland is of great importance so that appropriate data can be collected to assess the relevant variables.

Most of the variables in the model equations were calibrated based on a categorical basis, with scores given to values that helped the wetland to perform the function at a high level instead of at reference standard conditions. This was due to a poor relationship of these indicators with disturbance. The variable that was calibrated based on a linear relationship with disturbance was $V_{ROUGH.}$ The subindex scores remain the same among subclasses for most of the variables with the exception of V_{ROUGH}, and V_{ORGMA}. For these two variables, scores differ between subclasses, which is another reason that accurate site classification is essential.

The ability of a function to reflect disturbance is directly related to the ability of the individual variables to predict disturbance, which depends on the method of calibration used for each variable. The different methods of variable calibration and details on individual variable scores across ecoregions are discussed in Section II.B.3.b.2 (Hydrogeomorphic Model Building Process). Figures 8-10 show the relationship between the FCI and the degree of human alteration at the site, illustrating the how well the model responds to human disturbance for each ecoregion.

Figures 8-10. Relationship of Riparian Depression FCI and disturbance for sites in the Ridge and Valley, Allegheny Plateau and Glaciated Poconos (no data was available for the Piedmont). _ = Reference Standard Sites

Figure 8.

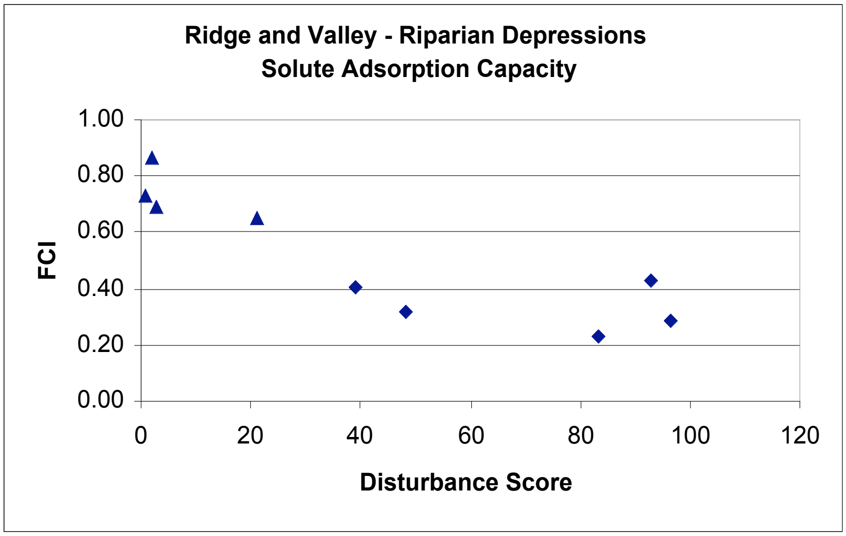


Figure 9.

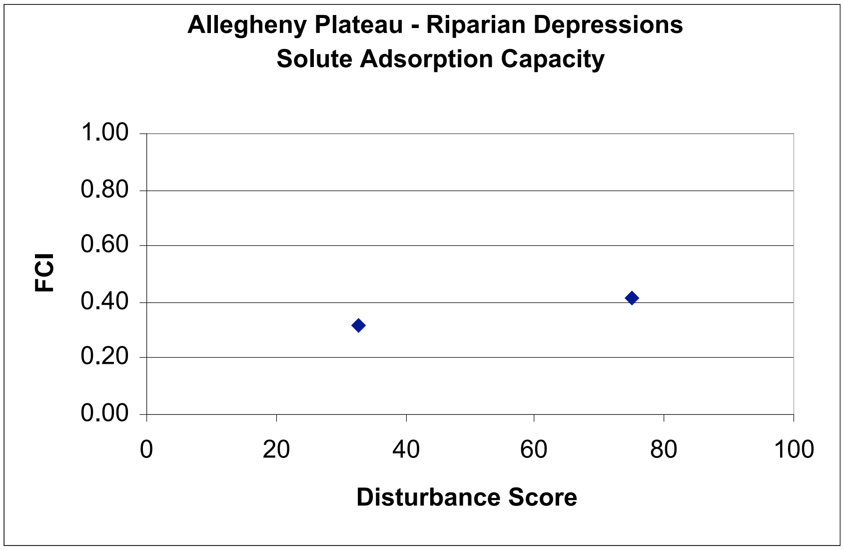
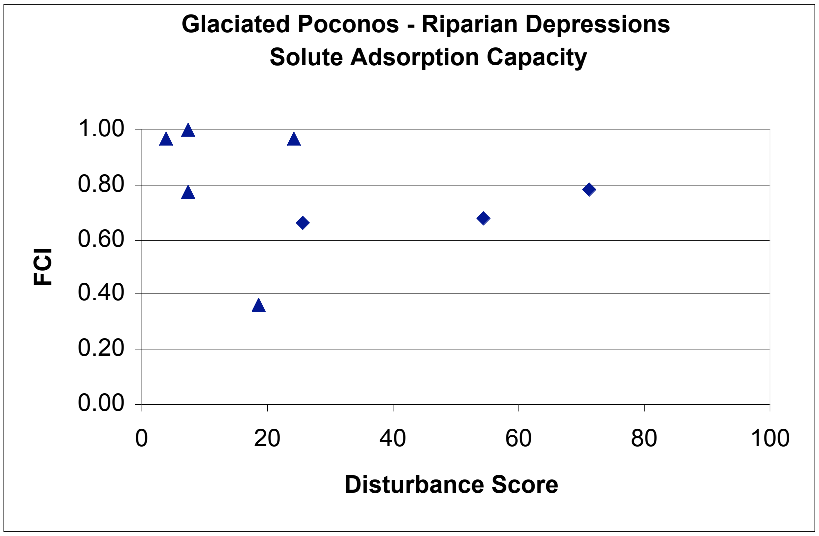


Figure 10.


Glaciated Poconos - Riparian Depressions
Solute Adsorption Capacity
FCI
1.00
0.80
0.60
0.40
0.20
0.00
0
20
40
60
80
100
Disturbance Score

Function 7. Retention of Inorganic Particulates

Not applicable to Riparian Depressions

Function 8. Export of Organic Carbon (Dissolved and Particulate)

Definition and applicability

This function evaluates the ability of wetlands to export both dissolved organic carbon and organic particulates. The total export of dissolved organic carbon from a wetland consists of three main processes: DOC loss in water leaving the wetland, microbial respiration (CO_2), and photosynthesis. Particulate organic matter includes soil organic matter and leaf litter, as well as sources of particulate organic matter such as coarse woody debris and standing dead trees. Originally, this function was split into two separate functions. However, to avoid redundancy the two functions have been combined. We are assuming that if a site is successfully exporting organic particulates it is likely that it is exporting dissolved organic carbon. This function is assessed for the following regional wetland subclasses:

a. Riparian Depressions
b. Slopes
c. Headwater Floodplains
d. Mainstem Floodplains
e. Fringing

The assessment model for this function is split into two components. Since this function is dependent on water moving through the site, as with previous functions, the first component is representative of the hydrologic regime at the site. The second component has represent residence time of water at the site, availability of organic matter and sources of organic matter. It is in the first component where variables differ among subclasses, reflecting fundamental differences in the hydrology of the different subclasses.

Rationale for selecting the function

Wetlands are a major source of particulate organic matter (POM). Woody debris is a nutritional substrate, provides habitat for microbes, invertebrates, and vertebrates, is a substrate for seedling growth, and serves as a long-term nutrient reservoir, a consistent source of organic material (Harmon et al. 1986, Brown 1990). Particulate carbon is a small fraction of total organic carbon (TOC), but is of a disproportionately higher importance as a food source for fish and invertebrates (Taylor et al. 1990). POM is a nutritional source for stream fauna. Particulate organic carbon (POC) from wetlands contributes substantial amounts of carbon to stream channels (Dosskey and Bertsch 1994). In fact, POC comprises between 24 and 46% of the total organic carbon in streams (Dosskey and Bertsch 1994). Detrital inputs to the stream during peak inundation periods support microbial and macroinvertebrate communities in the stream channel (Smock 1990).

Jones and Smock (1991) referred to floodplain wetlands as integral processing components of low-gradient stream systems. They suggested that most of the coarse particulate organic matter (CPOM) entering floodplains is received, retained, at least partially processed, and then exported in large quantities as a finer form more readily useable by aquatic invertebrate communities(Smock 1990, Jones and Smock 1991). On floodplains, CPOM is processed to fine particulate organic matter (FPOM) and dissolved organic matter (DOM), which may be transported to channels or respired as CO_2 (Bilby and Likens 1979, Jones and Smock 1991).

Dissolved organic carbon (DOC) is composed of humic substances, which are a group of complex organic compounds (fulvic acid, humic acid, and humin) that vary in solubility and molecular weight. DOC originates from recent soil organic matter (less than 45 years old), woody debris, and leaching from stems and leaves. DOC is an intermediary in the decomposition process; it is both produced and consumed (Schiff et al. 1998). The net DOC produced is the balance between these two processes (Schiff et al. 1998). Wetlands contribute large quantities of organic matter to streams, especially in the form of DOC. Dosskey and Bertsch (1994) found that while wetlands made up only 6% of the watershed, they contributed 93% of the total organic carbon flux. The majority of organic carbon is in dissolved form and export may be an order of magnitude greater than particulate export (Bilby and Likens 1979, Naiman 1982). Several studies done in mainstem floodplain systems show that approximately

95% of the annual export to the stream, and up to 93% of the TOC of the stream consisted of DOC (Mulholland 1981, Cuffney 1988).

DOC levels influence the chemical and the biological processes of wetlands. DOC provides energy to microbes, which in turn form the base of the detrital food web in river ecosystems (Dosskey and Bertsch 1994). DOC also plays an important role in the transport and toxicity of metallic ions (Schiff et al. 1990, Dillon and Molot 1997, Schiff et al. 1998). DOC is a strong complexing agent for Fe, Cu, Al, Zn, and Hg (Schiff et al. 1990). Organic acids can have acidifying effects on natural water bodies (Schiff et al. 1990, Dillon and Molot 1997). DOC affects light penetration into water and protects aquatic organisms from damaging UV radiation (Dillon and Molot 1997, Schiff et al. 1998). DOC export to streams influences the carbon cycle of both systems. Although DOC export is small compared to carbon respiration (Schiff et al. 1998), the export of carbon decreases carbon sequestration in the wetland, possibly turning the wetland into a source rather than sink for carbon.

Characteristics and processes that influence the function

The export of organic particulates requires a mechanism of transportation (usually water) and an accessible route (i.e., physical outlets) through which to exit the wetland. It is generally thought that a larger percentage of the fine organic particulates are exported from wetland systems than of the coarse fraction (Jones and Smock 1991), however, medium sized particles frequently exit wetlands due to their lower specific gravity and inherent tendency to float (Benke and Wallace 1990). In fact, Collier Creek, a large frequently inundated floodplain, was an exporter of medium-sized CPOM pieces, such as leaf litter that had been processed into smaller particles (Smock 1990). This trend was not observed in their less frequently flooded site (Smock 1990). Jones and Smock (1991) found that during overbank flow, much of the coarse particulate organic matter moved from the stream channel onto the floodplains, whereas FPOM (the primary size of particulate organic matter that supports invertebrate communities) was exported in large quantities. This was supported by the fact that FPOM comprised 99% of the POM present in the water column of the stream (Jones and Smock 1991).

The rate of particulate matter degradation depends on many factors, including soil moisture levels. According to Bilby et al. (1999), when compared to either fully submerged or terrestrial conditions, wood decays at a much faster rate when periodically wetted and dried, conditions typical of many wetlands. Floodplains had higher decomposition rates for wood than streams (Cuffney 1988). CPOM on dry areas of a streambed undergoes no processing or breakdown, and thus, FPOM formation is low (Bilby and Likens 1979). The breakdown of CPOM into smaller particle sizes influences POM export. Finer particulates are more susceptible to export because they are smaller and lighter and thus move easier with water flow. Heavier or larger pieces need higher water volume and velocity to be moved.

The duration of time between flooding events determines the amount of POM that can build-up. There was a 10-fold variation in FPOC at similar discharge levels in a headwater floodplain, in part because of the variation in duration of dry weather for FPOC to accumulate in the wetland (Bilby and Likens 1979). In winter, there is little biological processing due to low temperatures, so CPOM is able to accumulate. Storage of non-wood CPOM on floodplains increased and peaked in January, and the highest CPOM storage was mainly large pieces >16mm (Smock 1990). FPOM concentrations and export is low during snowmelt because winter decreases the biological breakdown of CPOM and thus formation of FPOM (Bilby and Likens 1979).

Export of DOC is controlled by several factors. The variables used relate to the amount of carbon potentially contributing to the DOC pool, frequency and duration of inundation, and intensity of water flow. The various types of organic carbon present in the wetland are important sources of DOC. Sources of organic matter determine the amount of carbon potentially available for export. Potential DOC production increases with increasing total organic carbon in the system. Long-term surface water storage controls the duration of water contact with DOC sources. The export of DOC from wetlands is affected by the ability of the wetland to store water for short periods of time and control the export volume and rate of the water.

High DOC concentrations found in swamps may be due to: DOC leaching from soil litter during water contact, swamp surface water receiving additional DOC-rich inputs of throughfall and stemflow, low DOC utilization because of the refractory nature of some compounds, and high evapotranspiration rates in wetlands compared to uplands (Marion and Brient 1998). Wetland size and position in the landscape may also be important in terms of surface area for water-soil contact and surface water inflow. The number of wetlands in a watershed has been strongly positively correlated with DOC levels in streams (Mulholland 1981, Dalva and Moore 1991, Dillon and Molot 1997). Disturbances impact many characteristics of wetlands, including wetland productivity and hydrology. Both of which can have an affect on the amount and export of DOC. If a disturbance causes a reduction in the health of the vegetation, the productivity should decrease, resulting in a reduction in the potential amount of DOC available at a site. Human alterations may lower the water table and cause streams to become incised. These changes result in drier, less frequently flooded wetlands. The lack of flood events will lead to lower export opportunities and the dryness will cause carbon mineralization rates to decrease, leading to lower DOC concentrations. Channelization decreases the sinuosity of the river and increases channel gradient, which results in sharper pulses in flow (Brinson 1990). This leads to higher water flow velocities, which can cause more scouring and export of POM, instead of DOC, from the wetland.

General form of the assessment model

The model for assessing the export of organic particulates includes the following variables:

Riparian Depressions:

$V_{HYDROSTRESS}$: number of hydrologic modification indicators from stressor checklist
V_{REDOX}: presence of redoximorphic features in the upper soil profile
V_{ORGMA}: amount of organic matter in the upper soil profile
V_{FWD}: visual estimate of depth of litter layer from HSI models
$V_{CWD\text{-}BA}$: estimate of coverage of coarse woody debris along a transect
$V_{CWD\text{-}SIZE}$: presence of coarse woody debris in three size classes
V_{SNAGS}: presence of dead standing wood in four size classes

The general form of the assessment model is:

Riparian Depressions:

$$FCI = V_{HYDROSTRESS} * [(V_{REDOX} + V_{ORGMA} + V_{FWD} + (V_{CWD\text{-}BA}+V_{CWD\text{-}SIZE})/2 + V_{SNAGS})/5$$

As in previous functions where movement of water through the system is a determining factor on the performance of the function, this function also is split into two major components. The first component is representative of the hydrologic regime of the wetland. The second component represents the different source of organic matter at the site. V_{ORGMA} and V_{FWD}

represent organic matter that is presently available for export. The variables $V_{CWD\text{-}BA}$, $V_{CWD\text{-}SIZE,}$ and V_{SNAGS} represent organic matter at the site that will potentially be available in the future as decomposition proceeds. The variable V_{REDOX} represents the residence time of water at the site, which is necessary for the formation of dissolved organic matter. The variables $V_{CWD\text{-}BA}$ and $V_{CWD\text{-}SZ}$ are averaged together since we believe that both are necessary to properly indicate the presence of CWD at the site. Otherwise, the arithmetic mean is taken for all the variables in the second component of the function, since each contributes equally to the overall functioning of the site. This function is very similar among subclasses, with the first component of the equations being the only difference between subclasses.

<u>Subclass rigor</u>

Although the differences in the models appear to be minimal, this function is class specific due to the inherent differences in hydrodynamics among subclasses. Therefore, the first component of the model is different depending on the wetland subclass. The variables V_{FWD}, $V_{CWD\text{-}SIZE}$, $V_{SNAGS,}$ $V_{HYDROSTRESS}$, and V_{REDOX} are all scored categorically based on highest level of functioning. V_{ORGMA} was also scored categorically, but with subclass differences. The remaining variables in the equation are calibrated based on a linear relationship to disturbance and are scored differently depending on subclass type.

Riparian depressions are assessed differently then other subclasses since the hydrology of these systems is significantly different. The variables representing the movement of water through the site are based on variables used in function three, maintenance of characteristic hydrology.

The ability of a function to reflect disturbance is directly related to the ability of the individual variables to predict disturbance, which depends on the method of calibration used for each variable. The different methods of variable calibration and details on individual variable scores across ecoregions are discussed in Section II.B.3.b.2 (Hydrogeomorphic Model Building Process). Figures 11-13 show the relationship between the FCI and the degree of human alteration at the site, illustrating the how well the model responds to human disturbance for each ecoregion.

Figures 11-13. Relationship of Riparian Depression FCI and disturbance for sites in the Ridge and Valley, Allegheny Plateau and Glaciated Poconos (no data was available for the Piedmont). _ = Reference Standard Sites

Figure 11.

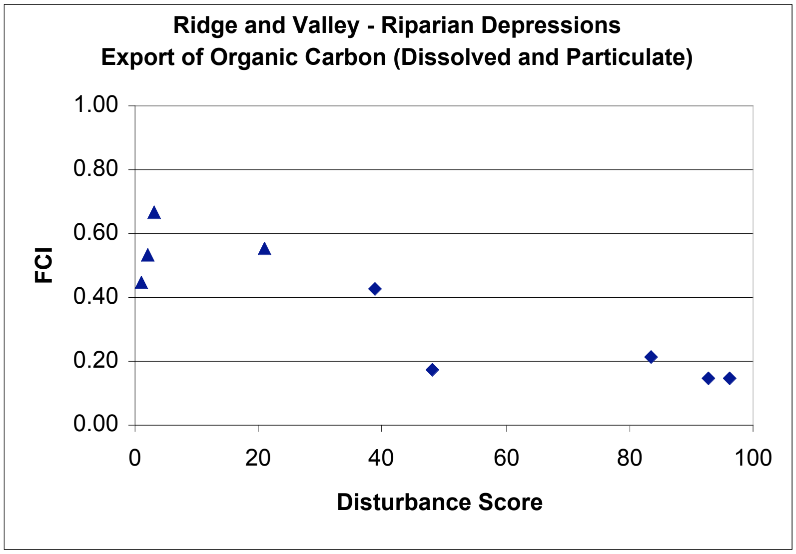


Figure 12.

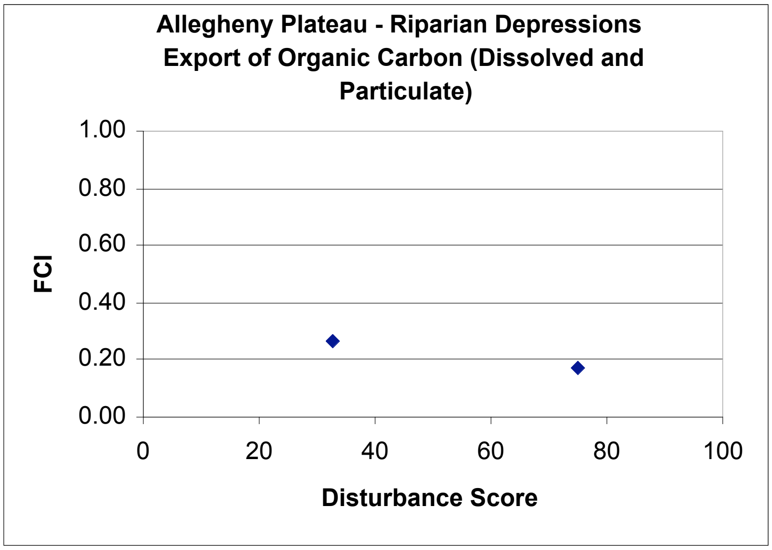
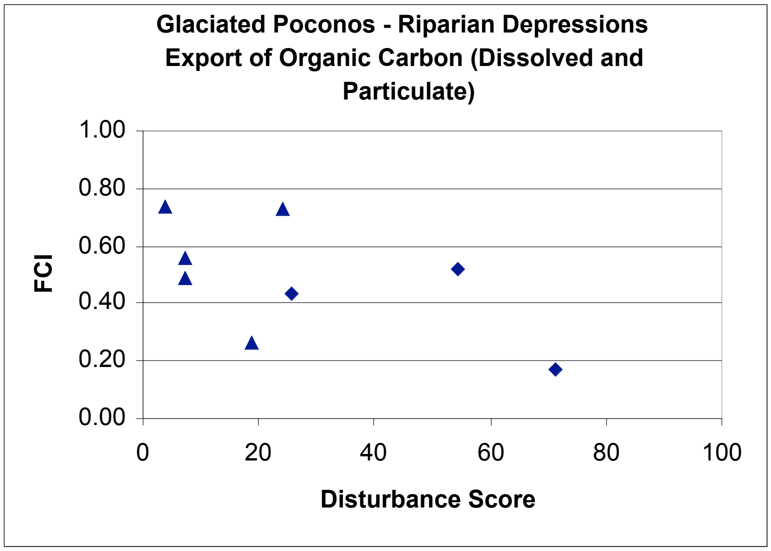


Figure 13.


Glaciated Poconos - Riparian Depressions
Export of Organic Carbon (Dissolved and Particulate)
1.00
0.80
0.60
0.40
0.20
0.00
FCI
0
20
40
60
80
100
Disturbance Score

Function 9. Maintain Characteristic Native Plant Community Composition

Definition and applicability

This function is defined as the ability of a wetland to support native plant species while taking into consideration the presence of invasive species (defined as exotics and native aggressive species). Due to the rapid assessment approach used in the HGM approach, the function looks at the extant plant community as an indicator of sites ability to maintain characteristic conditions. Since Pennsylvania is historically in a forested region, regeneration of native woody species is used as an indicator of the ability to maintain or develop a characteristic forested plant community. This function is assessed for the following regional wetland subclasses:

a. Riparian Depressions
b. Isolated Depressions
c. Slopes
d. Headwater Floodplains
e. Mainstem Floodplains
f. Fringing

The procedure for assessing this function incorporates three characteristics that define the state of the plant community. Species composition and percent invasives represent the quality of the present plant community. Regeneration of the forest community provides an indication that the site is moving toward or maintaining conditions typical of reference standard sites.

Rationale for selecting the function

The composition of vascular plant communities have long been used to characterize wetlands (Cowardin et al. 1979, Mitsch and Gosselink 2000). Plant community composition influences many ecosystem properties, such as primary productivity, nutrient cycling and hydrology (Hobbie 1992, Ainslie et al. 1999). Plant species composition plays an important role in determining soil fertility (Wedin and Tilman 1990, Hobbie 1992). Individual plant species effects on ecosystem fertility can be as or more important than abiotic factors, such as climate (Hobbie 1992). Resource uptake and allocation differs between species, as does tissue quality, and differences in litter quality affects nutrient cycling (Wedin and Tilman 1990, Hobbie 1992). Community composition also influences the habitat quality for invertebrate, vertebrate, and microbial communities (Gregory et al. 1991, Norokorpi 1997, Ainslie et al. 1999). The maintenance of a characteristic plant community can also be related to other HGM functions such as: energy dissipation via roughness, detrital production and nutrient cycling, and biodiversity and habitat functions.

Characteristics and processes that influence the function

Plant communities are highly influenced by human disturbance due to the fact that human alterations generally act as a means of establishment for invasive and aggressive species. Invasive species change competitive interactions, which result in changes in species composition (Walker and Smith 1997, Woods 1997). These changes in species composition often lead to changes in mineral and hydrologic cycling (Woods 1997). Impacts of invasive species include: simple competitive replacement of one or a few native species to the loss of an entire plant guild, modification of one stratum, and a change in plant community structure (Woods 1997). Very little information is available regarding the rates and spatial patterns of species invasion and spread (Higgins et al. 1996). However, it is generally accepted that disturbed sites, both natural and anthropogenic, are more easily invaded (Elton 1958, Mooney and Drake 1986, Huenneke et al. 1990, Burke and Grime 1996). The susceptibility of an indigenous community to invasive species is strongly related to the availability of bare ground and increased fertility (Burke and Grime 1996).

General form of the assessment model

The model for assessing the maintenance of a native plant community includes the following variables:

Riparian Depressions:

$V_{SPPCOMP}$: Floristic Quality Assessment Index (FQAI)

V_{REGEN}: regeneration of native tree species

V_{EXOTIC}: percent exotic species

The general form of the assessment model is:

Riparian Depressions:

$FCI = [(V_{SPPCOMP} * 0.66 + V_{REGEN} * 0.33) + V_{EXOTIC}]/2$

To evaluate this function, three metrics that indicate the present state of the plant community, $V_{SPPCOMP}$, V_{REGEN}, and V_{EXOTIC} have been selected. All three variables were calibrated based on characteristic conditions at reference standard sites. In this equation, $V_{SPPCOMP}$ and V_{REGEN} are first considered together in a cumulative interaction. These two components represent the plant community at the present time as well as what the potential canopy tree community may be in the future. $V_{SPPCOMP}$ was weighted more heavily then V_{REGEN} since present conditions at the site are more reliable and relevant than what conditions may be like if the site remains undisturbed. Also, V_{REGEN} only indicates the canopy tree community while $V_{SPPCOMP}$ considers the entire plant community. The variable V_{EXOTIC} is then assumed to be contributing equally and independently to the outcome of the function. The arithmetic mean of the two terms is then calculated to avoid a score of zero if invasive species cover exceeds 50%.

Subclass rigor

This function is assessed the same for all HGM subclasses, except Fringing sites. All three variables were calibrated based on reference standard conditions. V_{REGEN} and $V_{SPPCOMP}$ are both scored on reference standard conditions specific to subclass. V_{EXOTIC} is based on thresholds of % non-native species and is independent of subclass type. Since the majority of variables are scored based on subclass, the classification of the wetland becomes relevant when assessing a site.

The ability of a function to reflect disturbance is directly related to the ability of the individual variables to predict disturbance, which depends on the method of calibration used for each variable. The different methods of variable calibration and details on individual variable scores across ecoregions are discussed in Section II.B.3.b.2 (Hydrogeomorphic Model Building Process). Figures 14-16 show the relationship between the FCI and the degree of human alteration at the site, illustrating the how well the model responds to human disturbance for each ecoregion.

Figures 14-16. Relationship of Riparian Depression FCI and disturbance for sites in the Ridge and Valley, Allegheny Plateau and Glaciated Poconos (no data was available for the Piedmont). _ = Reference Standard Sites

Figure 14.

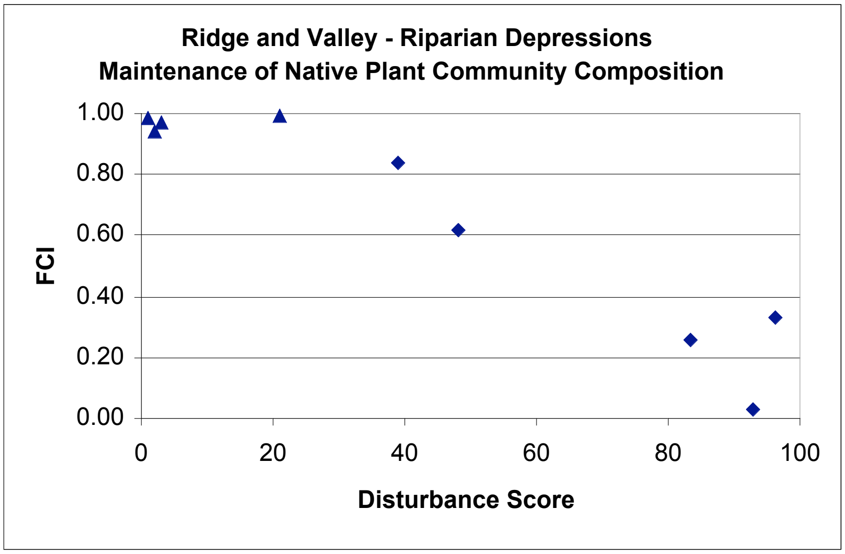


Figure 15.

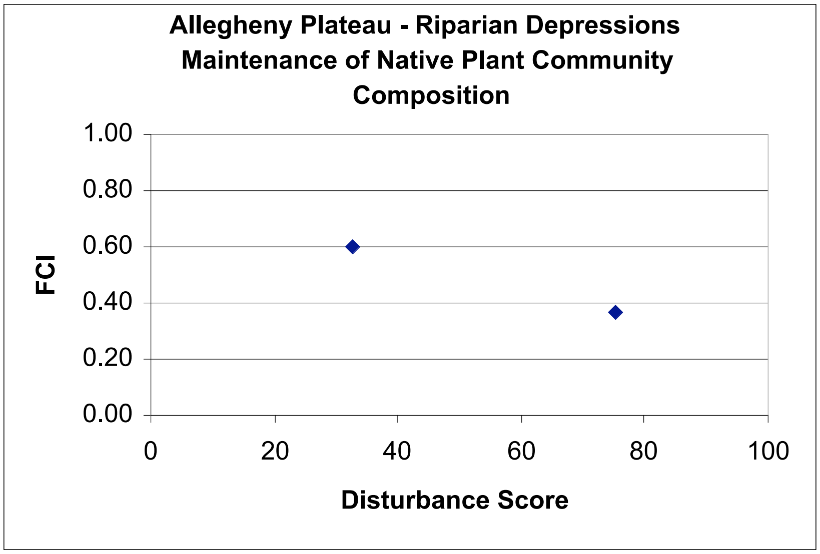
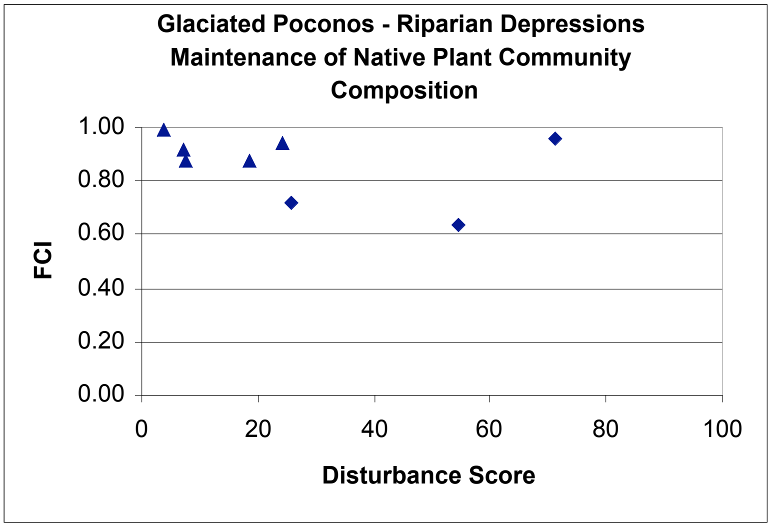


Figure 16.


Glaciated Poconos - Riparian Depressions
Maintenance of Native Plant Community Composition
FCI
1.00
0.80
0.60
0.40
0.20
0.00
0
20
40
60
80
100
Disturbance Score

Function 10. Maintain Characteristic Detrital Biomass

Definition and applicability

Detrital biomass is an important component of wetland ecosystems. It plays a role in nutrient cycling as well as providing habitat and substrate for plant and animal communities. Detrital biomass is represented by snags, down and dead woody debris, organic debris on the forest floor, and organic components of mineral soil, as described in the national riverine model (Brinson et al. 1995). This function compares the amount of detrital biomass present at a site, relative to the reference standard detrital biomass stocks. This model assumes, as did the national riverine model, that detritus standing stocks are proportional to detritus turnover, and can, therefore, be used to substitute for turnover (Brinson et al. 1995). This function is assessed for the following regional wetland subclasses:

a. Riparian Depressions
b. Isolated Depressions
c. Slopes
d. Headwater Floodplains
e. Mainstem Floodplains
f. Fringing

Rationale for selecting the function

For this function, detritus is considered an indicator of the potential decomposition and nutrient cycling rates at a site. Decomposition is a process supplied by the available pool of detrital biomass. Dead wood present at a site is processed into fine particulate organic matter (FPOM) and then further processed and incorporated into organic matter (Bilby and Likens 1979, Jones and Smock 1991). At the same time, these pieces of dead wood provide habitat for numerous invertebrate and vertebrate species. The organic matter derived from vegetation becomes part of the soil matrix and serves two major roles. The first is to provide a substrate for microorganisms that further decompose vegetation and facilitate important nutrient cycling processes such as denitrification. Studies show that organic soil has much greater NO_3^- removal capacity than sandy soils (Davidsson and Stahl 2000). Further, NO_3^- consumption is positively correlated to SOM content (Davidsson and Stahl 2000). Second, organic matter in the soil acts as a growth medium, facilitating regeneration of trees, shrubs and herbaceous plants, which will eventually die and begin the decomposition cycle again. Detritus is important for the maintenance of wetland fertility via decomposition of plant material. Overall, detritus acts as a nutritional substrate, provides habitat for microorganisms, invertebrates, and vertebrates, is a nursery for tree seedlings, and serves as a long-term consistent source of organic material and nutrients (Harmon et al. 1986, Brown 1990, Taylor et al. 1990). For this function, we have focused on the amount of detrital biomass present in the forms of coarse woody debris, dead standing wood, leaf litter, and soil organic matter at a site.

Characteristics and processes that influence the function

Decomposition processes include leaching of soluble material, mechanical fragmentation, and biological decay (Taylor et al. 1990). Decomposition rates are a function of electron-acceptor availability, chemistry of the organic substrate, and the environment (pH, temp, nutrients) (Reddy and D'Angelo 1994). The rate of decomposition depends on soil moisture levels; optimum conditions for decomposition are aerobic with adequate moisture (Brinson et al. 1981, Taylor et al. 1990). Aerobic decomposition is faster and yields more energy than anaerobic decomposition (Brinson et al. 1981, Reddy and D'Angelo 1994). Bilby et al. (1999) found that wood decays at a faster rate when periodically wetted and dried, conditions typical of many wetlands, as compared to fully submerged or terrestrial conditions. Decomposition is generally faster in aquatic than terrestrial landscapes due to increased leaching, fragmentation and microbial activity (Shure et al. 1986). Large pieces of CWD are processed into fine particulate organic matter (FPOM) and then further processed and incorporated into organic matter (Bilby and Likens 1979, Jones and Smock 1991). Organic material may be transported to channels or respired as CO_2 at any stage of the decomposition process (Bilby and Likens 1979, Jones and Smock 1991). Model calculations by Morris and Bowden (1986) found that the greatest change in nutrients occurred in the top 2 cm of soil and observed data showed that organic matter decomposition was faster in the top 5 cm than deeper in the soil. To estimate the potential for nutrient cycling to occur at a site, presence of biomass in each of the variable categories was determined and then either compared to sites with low human alteration or conditions which support high levels of functioning, to determine conditions suitable for nutrient turnover.

General form of the assessment model

The model for assessing the maintenance of characteristic detrital biomass includes the following variables:

Riparian Depressions:

$V_{CWD\text{-}BA}$: estimate of area covered by CWD
$V_{CWD\text{-}SIZE}$: presence of CWD in each of three size classes
V_{FWD}: amount of fine woody debris present as fallen leaves and downed twigs <1 cm.
V_{SNAGS}: density of dead standing trees by diameter size class
V_{ORGMA}: amount of organic matter in the top 5cm of the soil

The general form of the assessment model is:

Riparian Depressions;

$FCI = [(V_{CWD\text{-}BA} + V_{CWD\text{-}SIZE}/2) + V_{FWD} + V_{SNAGS} + V_{ORGMA}]/4$

To evaluate this function, four metrics have been selected that indicate the present amounts of detrital biomass: coarse woody debris, dead standing wood, leaf litter, and soil organic matter. We believe that each of these variables represents a different level of decomposition present at the site. In this equation, CWD was split into two categories abundance and size. The arithmetic mean of these two components was taken since each contributes equally to the overall representation of CWD. This CWD expression was then averaged with V_{FWD}, V_{SNAGS}, and V_{ORGMA}. Due to the cyclic nature of decomposition and the fact that each of these variables represents a part of that cycle, each variable is considered equally and independently by calculating the arithmetic mean

Subclass rigor

This function was assessed for all HGM subclasses using the same function equation. Scoring of the variables was the same for all subclasses except for V_{CWD-BA} and V_{ORGMA}. V_{CWD-BA} is calibrated based on conditions at sites with the least amount of human alteration. V_{ORGMA} is calibrated based on subclass-specific as well as ecoregion specific methods discussed in detail in Function 5. It is important that a site be classified correctly, due to these differences in scoring.

The ability of a function to reflect disturbance is directly related to the ability of the individual variables to predict disturbance, which depends on the method of calibration used for each variable. The different methods of variable calibration and details on individual variable scores across ecoregions are discussed in Section II.B.3.b.2 (Hydrogeomorphic Model Building Process). Figures 17-19 show the relationship between the FCI and the degree of human alteration at the site, illustrating the how well the model responds to human disturbance for each ecoregion.

Figures 17-19. Relationship of Riparian Depression FCI and disturbance for sites in the Ridge and Valley, Allegheny Plateau and Glaciated Poconos (no data was available for the Piedmont). _ = Reference Standard Sites

Figure 17.

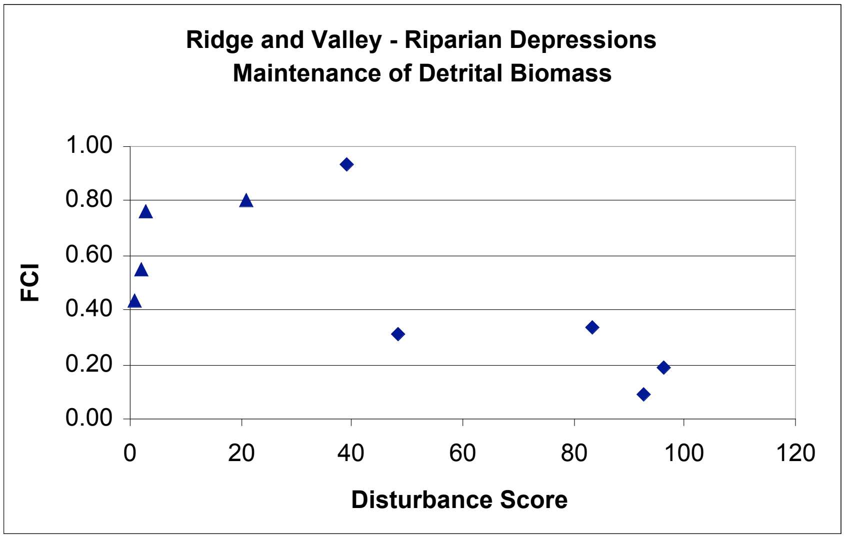


Figure 18.

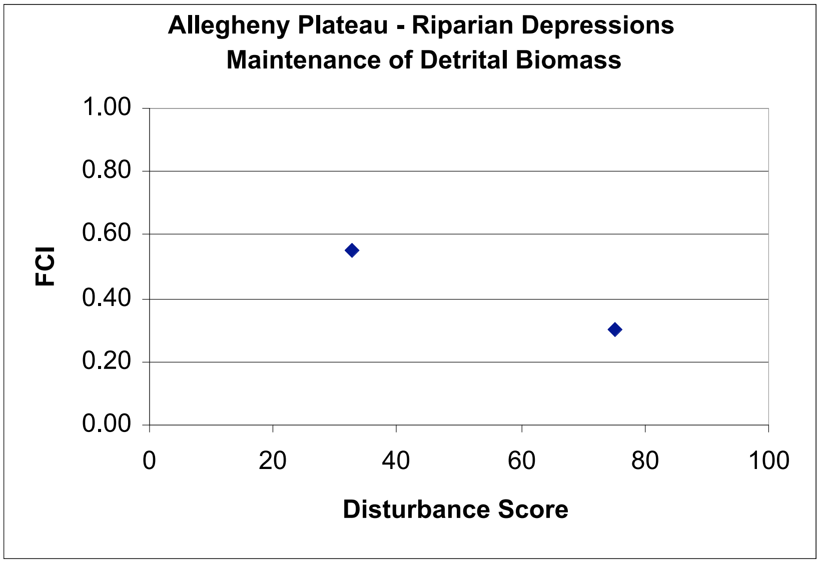
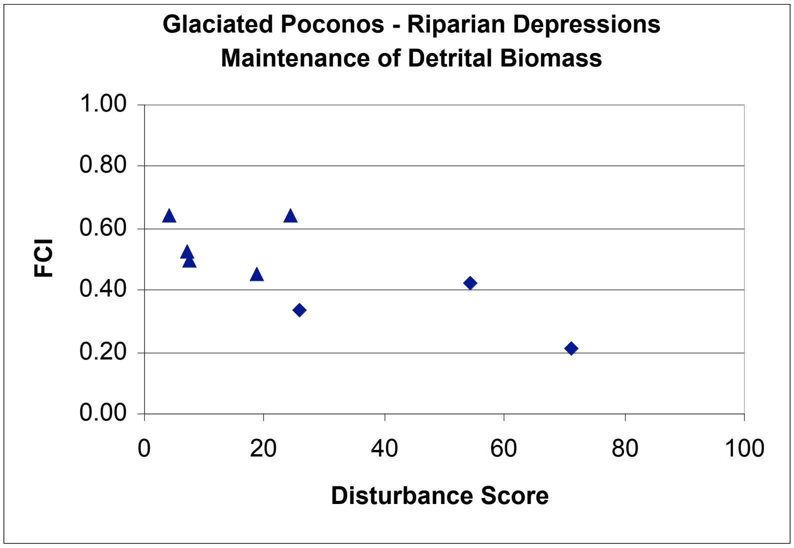


Figure 19.



Function 11. Vertebrate Community Structure and Composition

Definition and applicability

This function is assessed for the following regional wetland subclasses:

a. Riparian Depressions
b. Isolated Depressions
c. Slopes
d. Headwater Floodplains
e. Mainstem Floodplains
f. Fringing

Rationale for selecting the function

The provision of wildlife habitat is an often cited function of wetlands. Yet, we seldom have resources to census a diverse wildlife community. A commonly used alternative is to assess potential wildlife use with Habitat Suitability Index (HSI) models (USFWS 1980, Morrison et al. 1992, Anderson and Gutzwiller 1994). Thus, for this function, we adopted HSI models as a means to estimate the level of wetland functioning as wildlife habitat.

From the available pool of "blue book" models developed by the U.S. Fish and Wildlife Service (1980) and similar regional adaptations, such as the Pennsylvania Modified Habitat Procedures (Pennsylvania Game Commission 1982), we selected models for common species whose habitat preferences span both the vegetative and hydrologic gradients found in inland, freshwater wetlands typical of the northeastern U.S. We used a standard set of 10 wildlife species to construct a Wildlife Community Habitat Profile (WCHP), that included bullfrog *(Rana catesbeiana)*, muskrat *(Ondatra zibethicus)*, meadow vole *(Microtus pennsylvanicus)*, red-winged blackbird *(Agelaius phoeniceus)*, American woodcock *(Philohela minor)*, common yellowthroat *(Geothlypis thrichas)*, green-backed heron *(Butorides striatus)*, wood duck *(Aix sponsa)*, wood frog *(Rana sylvatica)*, and red-backed vole *(Clethriononmys gapperi)*.

The advantages of using the WCHP method include: 1) selection of species models no longer has to be tailored to each site; 2) comparisons among sites are consistent across the same set of species; 3) visual representation of the wildlife community is produced for each site, and 4) the vegetative diversity inherent in most wetlands is accounted for by using this diverse set of models.

Characteristics and processes that influence the function

Variables and calibration procedures for this function were conducted independently from the process used for the other functions. The process used is described in the HGM Model Building module (II.B.3.b.2). The actual models used, originally produced by Brooks and Prosser (1995), are presented as part of the section on sampling protocols (II.B.3.a.).

General form of the assessment model

The FCI scores for this function are calculated by using scores from Habitat Suitability Index (HSI) Models calculated for 10 common wetland species. (Brooks and Prosser 1995). FCI scores are based on HSI model scores at reference standard sites across subclasses and the amount of deviation from these reference standard conditions. The actual method for calculating FCI scores is discussed further in Section II.B.3.b.2 (Hydrogeomorphic Model Building Process).

Subclass rigor

HSI models and FCI scores are calculated identically across HGM subclasses. Therefore, this function is rigorous to misclassification issues.

Figures 20-22 show the relationship between the FCI and the degree of human alteration at the site, illustrating the how well the model responds to human disturbance for each ecoregion.

Figures 20-22. Relationship of Riparian Depression FCI and disturbance for sites in the Ridge and Valley, Allegheny Plateau and Glaciated Poconos (no data was available for the Piedmont). _ = Reference Standard Sites

Figure 20.

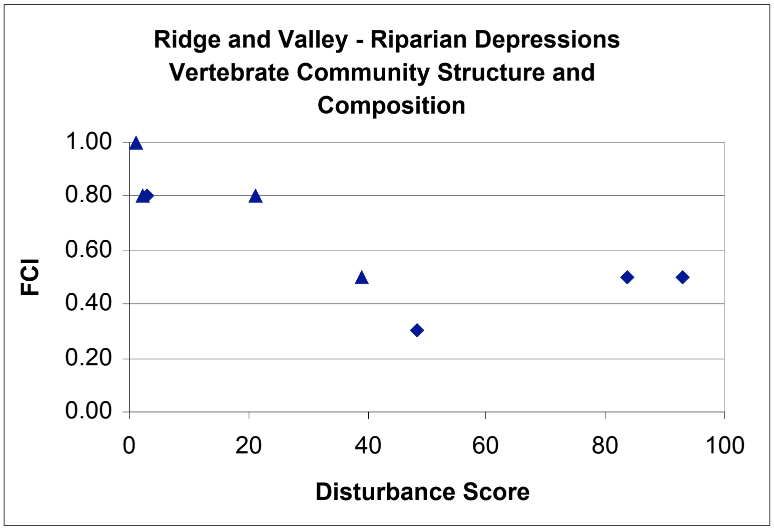


Figure 21.

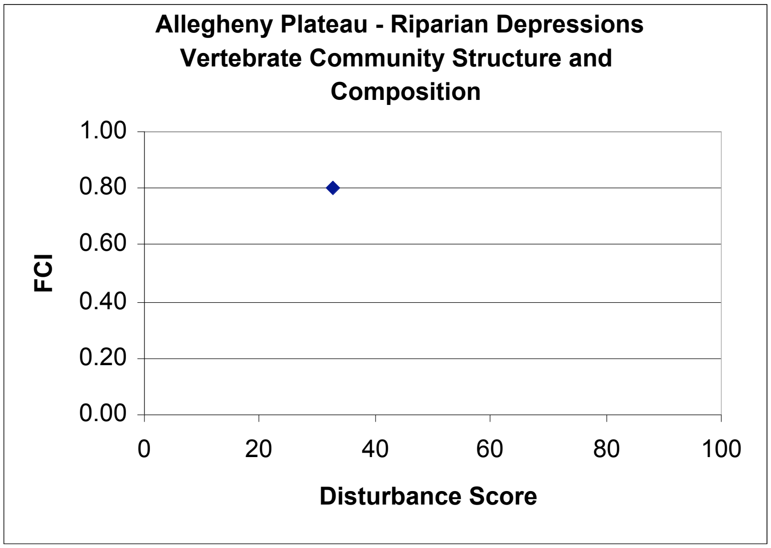
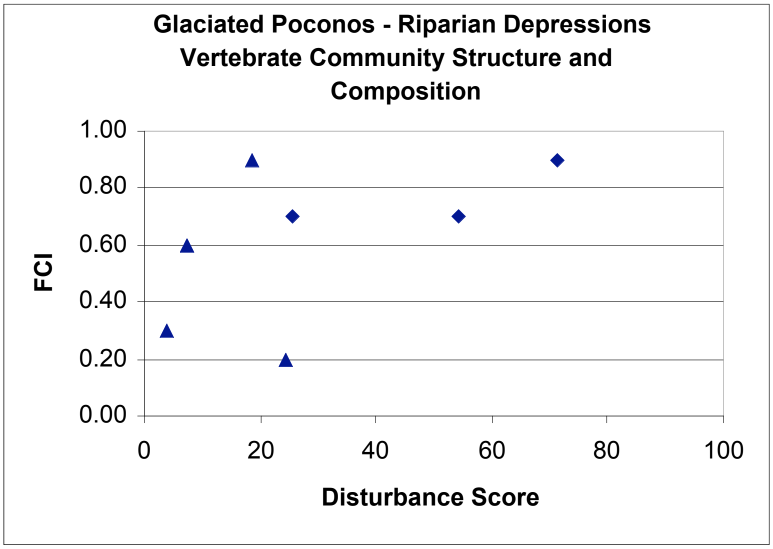


Figure 22.


Glaciated Poconos - Riparian Depressions Vertebrate Community Structure and Composition
1.00
0.80
0.60
0.40
0.20
0.00
FCI
0
20
40
60
80
100
Disturbance Score

Function 12. Maintain Landscape Scale Biodiversity

Definition and applicability

This function is assessed for the following regional wetland subclasses:

a. Riparian Depressions
b. Isolated Depressions
c. Slopes
d. Headwater Floodplains
e. Mainstem Floodplains
f. Fringing

Rationale for selecting the function

The strong influence of the surrounding landscape on a wetland's ability to perform a function has become increasingly evident (e.g., Gibbs 1993, Wardrop and Brooks 1998, O'Connell et al. 2000). To capture this factor, all variables for this function were based on measurements taken in a 1-km radius circle centered on each reference wetland. We have found that that distance incorporates stressors occurring in the landscape, but does not extend beyond the geomorphic setting for most wetland types in the ecoregions of Pennsylvania.

In the eastern U.S., we consider forested land cover to be the reference condition for most types of freshwater wetlands (e.g., Brooks et al. 2004). One way to characterize the extent of forest in the landscape matrix is by mean forest patch size (Forman 1995), so we used a variable derived from the forest patches within the circle. Connectivity among aquatic habitats has been shown to affect both faunal (e.g., Gibbs 1993) and floral communities, so we combined the best available synoptic data to construct that variable; 100-year floodplain, stream density, and nearest wetland. Similarly, urban development typically has negative impacts of aquatic communities (e.g., Karr and Chu). We represented that stressor with variables that characterize the proportion of urban land and road density. F12 integrates multiple stressors that potentially affect the way a wetland performs many of its functions.

General form of the assessment model

The model for assessing the maintenance of landscape scale biodiversity includes the following variables:

V_{AQCON}: degree of aquatic connectivity in a 1-km radius circle surrounding site. Composed of a combination of three indices: presence in 100-year floodplain, stream density index, and distance to nearest NWI wetland.

$V_{UNDEVEL}$: landscape variable made up of the average of two sub-variables:

- V_{RDDEN} – density of roads in 1-km radius circle
- V_{URB} - % of 1-km radius circle in urban development

V_{SDI}: natural log of the Shannon diversity index of eight landscape categories in the a 1-km radius circle around the site

V_{MFPS}: mean forested patch size within a 1-km radius circle

The general form of the assessment model for all HGM subclasses is:

$FCI = (V_{AQCON} + V_{UNDEVEL} + V_{SDI} + V_{MFPS})/4$

To evaluate this function, variables were chosen that represent the condition surrounding a wetland at a landscape scale. All indicators were based on measurements taken in a 1-km circle surrounding the site. Two of the variables, V_{AQCON} and $V_{UNDEVEL}$ were composites of other indicators in the 1-km radius circle. All variables were considered to contribute equally to the function and the arithmetic mean was taken. Although F12 is not identical to the human disturbance score generated for each wetland, it contains similar elements. For example, both values will score higher when the landscape circle contains more forest. Thus, when examining the figures, it is important to realize that we expect to see some correlation between the two scores because they represent different ways to express the condition of the landscape.

Subclass rigor

This function is assessed for all HGM subclasses using the same function equation. All variables were calibrated based on a linear relationship with disturbance. Each variable also has the same scoring criteria, regardless of HGM subclass. Therefore, this function is very robust to misclassification issues.

The ability of a function to reflect disturbance is directly related to the ability of the individual variables to predict disturbance, which depends on the method of calibration used for each variable. The different methods of variable calibration and details on individual variable scores across ecoregions are discussed in Section II.B.3.b.2 (Hydrogeomorphic Model Building Process). Figures 23-25 show the relationship between the FCI and the degree of human alteration at the site, illustrating the how well the model responds to human disturbance for each ecoregion.

Figures 23-25. Relationship of Riparian Depression FCI and disturbance for sites in the Ridge and Valley, Allegheny Plateau and Glaciated Poconos (no data was available for the Piedmont). _ = Reference Standard Sites

Figure 23.

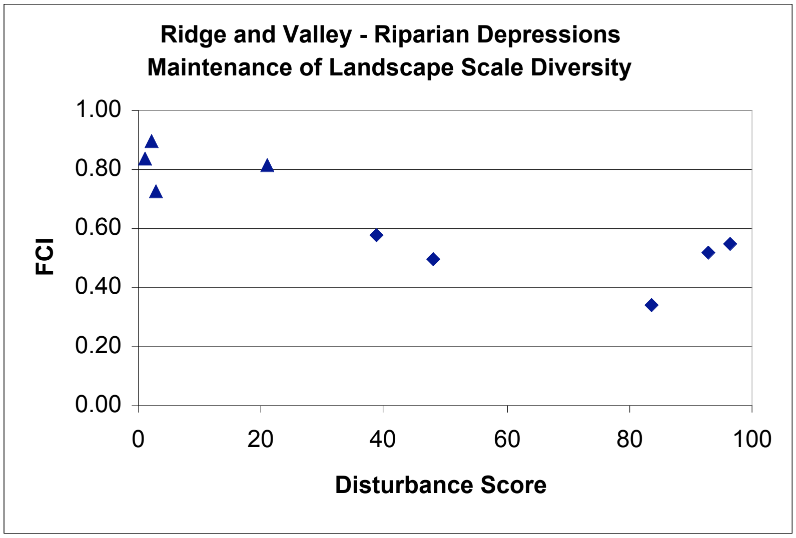


Figure 24.

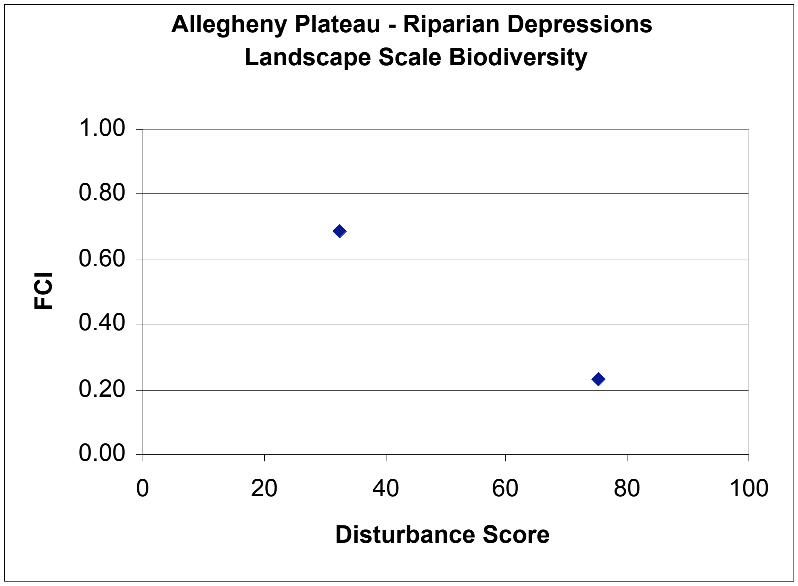
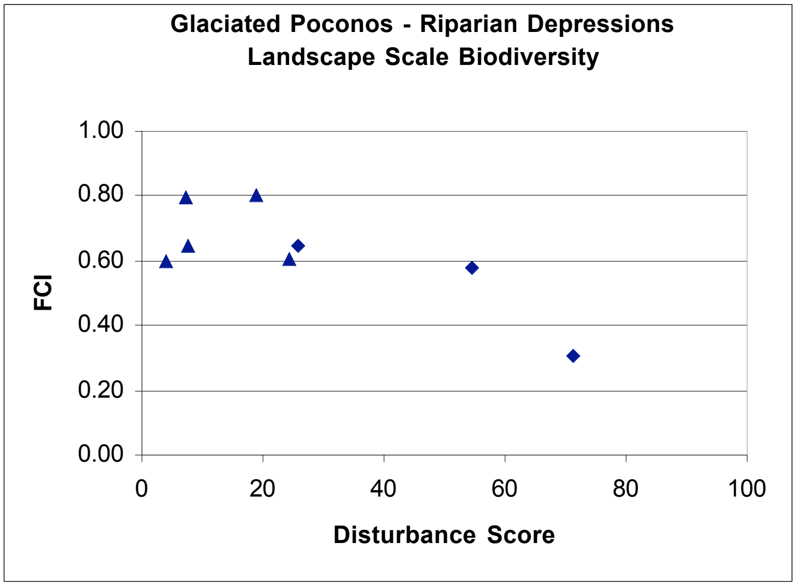


Figure 25.


Glaciated Poconos - Riparian Depressions
Landscape Scale Biodiversity
1.00
0.80
0.60
0.40
0.20
0.00
FCI
0
20
40
60
80
100
Disturbance Score

LITERATURE CITED

Ainslie, W. B., R. D. Smith, B. A. Pruitt, T. H. Roberts, E. J. Sparks, L. West, G. L. Godshalk, and M. V. Miller. 1999. A Regional Guidebook for Assessing the Functions of Low Gradient, Riverine Wetlands in Western Kentucky. WRP-DE-17, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Anderson, Stanley H., and Kevin. J. Gutzwiller. 1994. Habitat evaluation methods. Pages 592-606 *in* Research and management techniques for wildlife and habitats (Theodore A. Bookhout, Ed.). The Wildlife Society, Bethesda, MD.

Benke, A. C., and J. B. Wallace. 1990. Wood dynamics in coastal plain blackwater streams. Canadian Journal of Fisheries and Aquatic Sciences **47**:92-99.

Bilby, R. E., J. T. Heffner, B. R. Frasen, and J. W. Ward. 1999. Effects of immersion in water on deterioration of wood from five species of trees used for habitat enhancement prjects. Journal of North American Fisheries Management **19**:687-695.

Bilby, R. E., and G. E. Likens. 1979. Effect of hydrologic fluctuations on the transport of fine particulate organic carbon in a small stream. Limnology and Oceanography **24**:69-75.

Bowden, W. B. 1987. The biogeochemistry of nitrogen in freshwater wetlands. Biogeochemistry **4**:313-348.

Bridgham, S. D., C. A. Johnston, J. P. Schubauer-Berigan, and P. Weishampel. 2001. Phosphorus sorption dynamics in soils and coupling with surface and pore water in riverine wetlands. Soil Science Society of America Journal **65**:577-588.

Brinson, M. M. 1990. Riverine Forests. Pages 87-141 *in* S. Brown, editor. Forested Wetlands: Ecosystems of the World. Elsevier, Amsterdam.

Brinson, M. M. 1993. A hydrogeomorphic classification for wetlands. Technical Report WRP-DE-4, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Brinson, M. M., F. R. Hauer, L. C. Lee, W. L. Nutter, R. D. Rheinhardt, R. D. Smith, and D. Whigham. 1995. A guidebook for application of hydrogeomorphic assessments to riverine wetlands (Operational draft). Wetlands Research Program Technical Report WRP-DE-11, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Brinson, M. M., A. E. Lugo, and S. Brown. 1981. Primary productivity, decomposition and consumer activity in freshwater wetlands. Annual Review of Ecology and Systematics **12**:123-161.

Brooks, R. P., and D. J. Prosser. 1995. Wildlife Habitat Suitability Models. CWC Report 95-1, Penn State Cooperative Wetlands Center, University Park.

Brooks, Robert P., Denice Heller Wardrop, and Joseph A. Bishop. 2004. Assessing wetland condition on a watershed basis in the Mid-Atlantic region using synoptic land cover maps. Environmental Monitoring and Assessment. (in press)

Brown, R. G. 1985. Effects of an urban wetland on sediment and nutrient loads in runoff. Wetlands **4**:147-158.

Brown, R. G. 1988. Effects of wetland channelization on runoff and loading. Wetlands **8**:123-133.

Brown, S. L. 1990. Structure and dynamics of basin forested wetlands in North America. Pages 171-199 *in* S. Brown, editor. Forested Wetlands: Ecosystems of the World. Elsevier, Amsterdam.

Burke, M. J. W., and J. P. Grime. 1996. An experimental study of plant community invasibility. Ecology **77**:776-790.

Cole, C. A., R. P. Brooks, and D. H. Wardrop. 1997. Wetland hydrology as a function of hydrogeomorphic (HGM) subclass. Wetlands **17**:456-467.

Cooper, J. R., and J. W. Gilliam. 1987. Phosphorus redistribution from cultivated fields into riparian areas. Soil Science Society of America Journal **51**:1600-1604.

Cooper, J. R., J. W. Gilliam, and T. C. Jacobs. 1986. Riparian areas as a control of nonpoint pollutants. Pages 166-190 *in* D. L. Correll, editor. Watershed Research Perspectives. Smithsonian Institute Press, Washington DC.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. Laroe. 1979. Classification of wetlands and deepwater habitats of the United States. FWS/OBS-79/31, U.S. Fish and Wildlife Service, Washington D.C.

Cronan, C. S., J. T. Piampiano, and H. H. Patterson. 1999. Influence of land use and hydrology on exports of carbon and nitrogen in a Maine river basin. Journal of Environmental Quality **28**:953-961.

Crumpton, W. G., T. M. Isenhart, and S. W. Fisher. 1993. Fate of non-point source nitrate loads in freshwater wetlands: Results from experimental wetland mesocosms. Pages 283-291 *in* G. A. Moshiri, editor. Constructed Wetlands for Water Quality Improvement. Lewis Publishers, Ann Arbor, Michigan.

Cuffney, T. F. 1988. Input, movement and exchange of organic matter within a subtropical coastal blackwater river-floodplain system. Freshwater Biology **19**:305-320.

Dalva, M., and T. R. Moore. 1991. Sources and sinks of dissolved organic carbon in a forested swamp catchment. Biogeochemistry **15**:1-19.

Davidsson, T. E., and M. Stahl. 2000. The influence of organic carbon on nitrogen transformations in five wetland soils. Soil Science Society of America Journal **64**:1129-1136.

Detenbeck, N. E., C. A. Johnston, and G. J. Niemi. 1993. Wetland effects on lake water quality in the Minneapolis/St. Paul metropolitan area. Landscape Ecology **8**:39-61.

Dillon, P. J., and L. A. Molot. 1997. Effect of landscape form on export of dissolved organic carbon, iron, and phosphorus from forested stream catchments. Water Resources Research **33**:2591-2600.

Dosskey, M. G., and P. M. Bertsch. 1994. Forest sources and pathways of organic matter transport to a blackwater stream: A hydrologic approach. Biogeochemistry **24**:1-19.

Elton, C. S. 1958. The Ecology of Invasions by Animals and Plants. Methuen, London.

Forman, R. T. 1995. Land mosaics: the ecology of landscapes and regions. Cambridge University Press. 632pp.

Gambrell, R. P. 1994. Trace and toxic metals in wetlands-A review. Journal of Environmental Quality **23**:883-891.

Gibbs, J. P. 1993. Importance of small wetlands for the persistence of local populations of wetland-associated animals. Wetlands 13(1):25-31.

Gilliam, J. W. 1994. Riparian wetlands and water quality. Journal of Environmental Quality **23**:896-900.

Gregory, S. V., F. J. Swanson, W. A. McKee, and K. W. Cummins. 1991. An ecosystem perspective of riparian zones. BioScience **41**:540-551.

Groffman, P. M. 1994. Denitrification in freshwater wetlands. Current Topics in Wetland Biogeochemistry **1**:15-35.

Harmon, M. E., J. F. Franklin, F. J. Swanson, P. Sollins, S. V. Gregory, J. D. Lattin, N. H. Anderson, S. P. Cline, N. G. Aumen, J. R. Sedell, G. W. Lienkaemper, K. C. Jr., and K. W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research **15**:133-302.

Higgins, S. I., D. M. Richardson, and R. M. Cowling. 1996. Modeling invasive plant spread: The role of plant-environment interactions and model structure. Ecology **77**:2043-2054.

Hobbie, S. E. 1992. Effects of plant species on nutrient cycling. TREE **7**:336-339.

Huenneke, L. F., S. P. Hamburg, R. Koide, H. A. Mooney, and P. M. Vitousek. 1990. Effects of soil resources on plant invasion and community structure in Californian serpentine grassland. Ecology **71**:478-491.

Johnston, C. A., N. E. Detenbeck, and G. J. Niemi. 1990. The cumulative effect of wetlands on stream water quality and quantity. A landscape approach. Biogeochemistry **10**:105-141.

Jones, J. B., Jr., and L. A. Smock. 1991. Transport and retention of particulate organic matter in two low-gradient headwater streams. Journal of the North American Benthol. Society **10**:115-126.

Jordan, T. E., D. L. Correll, and D. E. Weller. 1993. Nutrient interception by a riparian forest receiving inputs from adjacent cropland. Journal of Environmental Quality **22**:467-473.

Kadlec, R. H. 2001. Phosphorus dynamics in event driven wetlands. Pages 365-391 *in* J. Vymazal, editor. Transformations of Nutrients in Natural and Constructed Wetlands. Backhuys Publishers, Leiden, The Netherlands.

Kadlec, R. H., and H. J. Alvord. 1989. Mechanisms of water quality improvement in wetland treatment systems. Pages 489-498 *in* D. W. Fisk, editor. Wetlands: Concerns and Successes. American Water Resources Association, MD.

Kadlec, R. H., and F. B. Bevis. 1990. Wetlands and wastewater: Kinross, Michigan. Wetlands **10**:77-92.

Marion, L., and L. Brient. 1998. Wetland effects on water quality: Input-output studies of suspended particulate matter, nitrogen (N) and phosphorus (P) in Grand-Lieu, a natural plain lake. Hydrobiologia **373-4**:217-235.

Mitsch, W. J., and J. G. Gosselink. 2000. Wetlands, 3 edition. John Wiley & Sons, Inc., New York, NY.

Mooney, H. A., and J. A. Drake. 1986. Ecology of Biological Invasions of North America and Hawaii. pringer-Verlag, New Yrok, NY.

Morris, J. T., and W. B. Bowden. 1986. A mechanistic, numerical model of sedimentation, mineralization, and decomposition for marsh sediments. Soil Science Society of America Journal **50**:96-105.

Morrison, M.L., B.G. Marcot, and R.W. Mannan. 1992. Wildlife-habitat relationships: Concepts and applications. University of Wisconsin Press. 343pp.

Mulholland, P. J. 1981. Organic carbon flow in a swamp-stream ecosystem. Ecological Monographs **51**:307-322.

Naiman, R. J. 1982. Characteristics of sediment and organic carbon export from pristine boreal forest watersheds. Canadian Journal of Fisheries and Aquatic Sciences **39**:1699-1718.

Norokorpi, Y. e. a. 1997. Stand structure, dynamics, and diversity of virgin forests on northern peatlands. *in* J. K. Jeglum, editor. Northern Forested Wetlands: Ecology and Management. Lewis Publishers, New York, NY.

Novitzki, R. P. 1989. Wetland Hydrology. Pages 47-64 *in* J. R. W. Tiner, editor. Wetlands Ecology and Conservation: Emphasis in Pennsylvania. The Pennsylvania Academy of Science, Easton, PA.

O'Connell, T. J., L. E. Jackson, R. P. Brooks. 2000. Bird guilds as indicators of ecological condition in the central Appalachians. Ecological Applications 10(6):1706-1721.

Peterjohn, W. T., and D. L. Correll. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. Ecology **65**:1466-1475.

Reddy, K. R., G. A. O. Connor, E. Flaig, and P. M. Gale. 1999. Phosphorus retention in streams and wetlands: A review. Critical Reviews in Environmental Science and Technology **29**:83-146.

Reddy, K. R., G. A. O. Connor, and P. M. Gale. 1998. Phosphorus sorption capacities of wetland soils and stream sediments impacted by dairy effluent. Journal of Environmental Quality **27**:438-447.

Reddy, K. R., and E. M. D'Angelo. 1994. Soil processes regulating water quality in wetlands. Pages 309-324 *in* W. J. Mitsch, editor. Global Wetlands: Old World and New. Elsevier, Amsterdam, The Netherlands.

Richardson, C. J., and C. B. Craft. 1993. Effective phosphorus retention in wetlands: Fact or fiction? Pages 271-282 *in* G. A. Moshiri, editor. Constructed Wetlands for Water Quality Improvement. Lewis Publishers, Ann Arbor, Michigan.

Schamberger, M., and A. Farmer. 1978. The habitat evaluation procedures: Their application in project and impact evaluation. N. Am. Wildl. and Nat. Resour. Conf. 43:274-283.

Schiff, S., R. Aravena, E. Mewhinney, R. Elgood, B. Warner, P. Dillon, and S. Trumbore. 1998. Precambrian shield wetlands: Hydrologic control of the sources and export of dissolved organic matter. Climate Change **40**:167-188.

Schiff, S. L., R. Aravena, S. E. Trumbore, and P. J. Dillon. 1990. Dissolved organic carbon cycling in forested watersheds: A carbon isotope approach. Water Resources Research **26**:2949-2957.

Schlesinger, W. H. 1997. Biogeochemistry: An Analysis of Global Change. Academic Press, New York, NY.

Scott, M. L., B. A. Kleiss, W. H. Patrick, and C. A. Segelquist. 1990. The effect of developmental activities on water quality functions of bottomland hardwood ecosystems: The report of the water quality workgroup. Pages 411-453 *in* T. A. Muir, editor. Ecological Processes and Cumulative Impacts: Illustrated by Bottomland Hardwood Wetland Ecosystems. Lewis Publishers, MI.

Seitzinger, S. P. 1994. Linkages between organic matter mineralization and denitrification in eight riparian wetlands. Biogeochemistry **25**:19-39.

Shure, D. J., M. R. Gottschalk, and K. A. Parsons. 1986. Litter decomposition processes in a floodplain forest. The American Midland Naturalist **115**:314-327.

Smock, L. A. 1990. Spatial and temporal variation in organic matter storage in low-gradient, headwater streams. Archives of Hydrobiolgy **118**:169-184.

Soranno, P. A., S. L. Hubler, and S. R. Carpenter. 1996. Phosphorus loads to surface waters: A simple model to account for spatial pattern of land use. Ecological Applications **6**:865-878.

Taylor, J. R., M. A. Cardamone, and W. J. Mitsch. 1990. Pages 13-86 *in* T. A. Muir, editor. Ecological Processes and Cumulative Impacts: Illustrated by Bottomland Hardwood Wetland Ecosystems. Lewis Publishers, MI.

U.S. Fish and Wildlife Service. 1980. Habitat evaluation procedures (HEP). Ecological Services Manual 101. USDI Fish and Wildlife Service, Washington, D.C.

Vought, L. B.-M., J. Dahl, C. L. Pedersen, and J. O. Lacoursiere. 1994. Nutrient retention in riparian ecotones. Ambio **23**:342-348.

Walbridge, M. R., and J. P. Struthers. 1993. Phosphorus retention in non-tidal palustrine forested wetlands of the mid-Atlantic region. Wetlands **13**:84-94.

Walker, L. R., and S. D. Smith. 1997. Community response to plant invasion. Pages 69-86 *in* J. W. Thieret, editor. Assessment and Management of Plant Invasions. Springer, New York, NY.

Wardrop, D. H., and R. P. Brooks. 1998. The occurrence and impact of sedimentation in central Pennsylvania wetlands. Environmental Monitoring and Assessment **51**:119-130.

Wedin, D. A., and D. Tilman. 1990. Species effects on nitrogen cycling: A test with perennial grasses. Oecologia **84**:433-441.

Woods, K. D. 1997. Community response to plant invasion. Pages 56-68 *in* J. W. Thieret, editor. Assessment and Management of Plant Invasions. Springer, New York, NY.