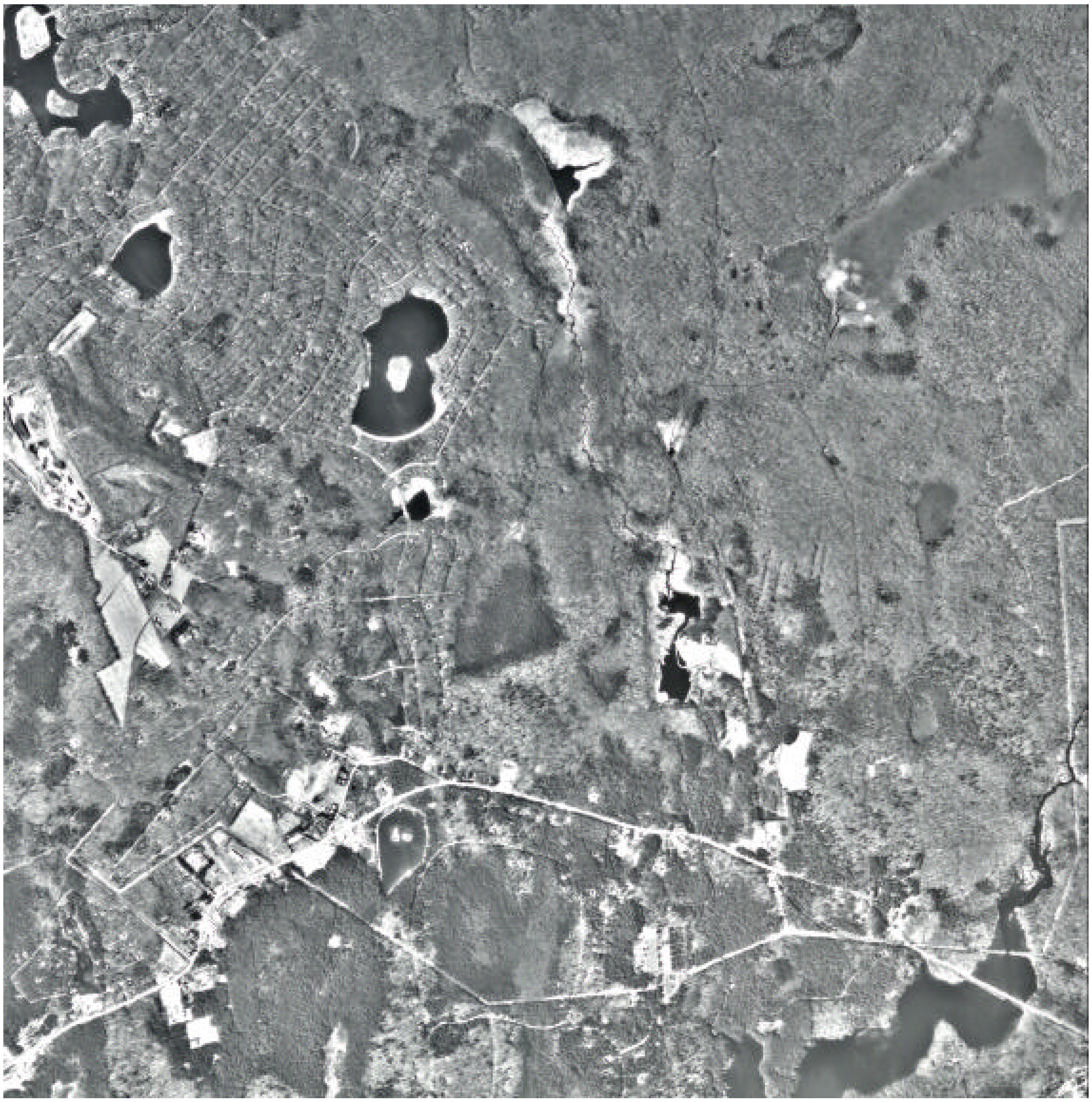
WATERSHED-BASED WETLAND PLANNING AND EVALUATION



Proceedings of a Symposium at the Wetland Millennium Event (an International Wetland Conference) Held August 6-12, 2000 in Quebec City, Quebec, Canada

Distributed by the Association of State Wetland Managers

WATERSHED-BASED WETLAND PLANNING AND EVALUATION

A Collection of Papers from the Wetland Millennium Event

August 6-12, 2000
Quebec City, Quebec, Canada

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Symposium Overview

This symposium presents examples of watershed-based approaches to wetland planning and evaluation. Much of the decision-making regarding the fate of individual wetlands has been and remains based on site-specific analysis and evaluation. This process tends to ignore the interrelationships and interdependencies among wetlands acting in combination as a functional unit. Today there is increasing interest in examination and management of the wetland resource from a watershed standpoint or landscape perspective. There is also widespread recognition that the health of wetlands is largely determined by various land use practices around wetlands and elsewhere in the watershed.

Wetlands perform many functions that are vital to maintaining a healthy watershed. They serve as flood storage basins, sinks for nutrients and sediments, stabilizers of shorelines, and habitat for many species of fish and wildlife. The quality of life for a society is, in many ways, determined by the abundance and condition of its natural resources. Recognizing the public values of wetland functions has led to the development of watershed-based approaches for characterizing wetlands and evaluating their performance.

This symposium brings together a number of watershed-based approaches that can serve as tools to aid resource managers in making decisions about wetlands. Such decisions would include permitted uses, acquisition, restoration, and other measures to strengthen protection for wetlands. These approaches also help educate non-wetland professionals and the general public about the relationships between wetland characteristics and functions and demonstrate that all wetlands do not necessarily perform all functions or functions at the same level of performance.

The Society of Wetland Scientists sponsored this special symposium at the Wetland Millennium Event, an international wetland conference, held August 6-12, 2000 in Quebec City, Quebec, Canada. This conference was a multi-organizational, international conference with participation from several wetland and peat-oriented scientific societies. The proceedings of this special symposium were compiled by the U.S. Fish and Wildlife Service, with subsequent distribution through the Association of State Wetland Managers, Inc. who gratiously agreed to post the proceedings on their web page and make hard copies and CD copies available to the public.

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An Approach to Geographic Prioritization of Wetland Management Given Limited Effort and Information

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Abstract

Setting priorities for protection and restoration efforts is necessary whenever resources are not sufficient to target all sites simultaneously. When this occurs priorities should be set to maximize the ecological benefit gained from available resources. Prioritizing protection and restoration efforts in this fashion is hampered, however, by a lack of information that can be used by resource managers. This paper describes an approach to prioritizing wetland restoration or protection that works around these constraints. A benefit/cost framework is used to deal with limited effort, and judgement indicators are employed to address information constraints. These two components have been incorporated into the Synoptic Approach developed by the U.S. Environmental Protection Agency. The Synoptic Approach was specifically designed as a prioritization approach that could incorporate best professional judgement in cases where information and resources are otherwise limited. The approach also makes use of a conceptual model to address problems introduced when indicator selection is driven by practical issues of data availability. The Synoptic Approach is an example of geographic prioritization, producing maps that comparatively rank counties, hydrologic units, or other suitable land units. Many of the concepts presented here are equally appropriate for ranking individual wetland sites.

Introduction

Much of the wetland management performed by government regulatory and subsidy agencies can be broadly classified into restoration or protection activities. For any organization or agency involved in more than one of these projects, prioritization of potential sites is likely to become an important issue. Prioritization becomes necessary if the number of candidate sites exceeds the number of sites that can be protected or managed with available effort. Given that resource constraints force us to select a subset of available sites, it is desirable to select a subset to maximize the ecological function that is restored or protected (Llewellyn et al. 1996; Hyman and Leibowitz 2000).

While resource limitations ultimately force prioritization of restoration or protection efforts, limitations in available information are a second constraint that strongly influences prioritization. Although it might be theoretically possible to identify a subset of sites that maximizes ecological function, the information this requires may not be available in practice (Hyman and Leibowitz 2000). It is therefore desirable to employ a prioritization approach that provides an optimal subset of sites based on information that is typically available to managers.

This paper describes an approach to prioritizing wetland restoration or protection that is designed to work within the constraints of limited effort and information. This approach is a synthesis of recent work published by the Landscape Function Project at the U.S. Environmental Protection Agency=s (EPA) National Health and Environmental Effects Research Laboratory - Western Ecology Division. A benefit/cost framework was introduced (Hyman and Leibowitz 2000) to deal with the issue of limited effort, and the concept of judgement indicators was developed (Leibowitz and Hyman 1999; Hyman and Leibowitz 2001) as a way of dealing with information constraints. These two components have been incorporated into the Synoptic Approach (Abbruzzese and Leibowitz 1997; McAllister et al. 2000), which has been under development at EPA for over a decade. The Synoptic Approach does not rank or prioritize individual sites. Rather, it is a regional approach to prioritization, providing maps that comparatively rank counties, hydrologic units, or other suitable land units. However, many of the concepts presented are equally appropriate for ranking individual sites.

Constrained Effort

Constraints in effort can include limits in the available amount of time, money, labor, and/or any other resource that can become a limiting factor. This is analogous to being in a candy store full of mouth-watering temptations with enough money for only one or two pieces of candy. The challenge is to maximize the amount of satisfaction that can be obtained with the limited amount of resources.

Although it may seem that resources would always be constrained, this may not always be the case. For example, more money could be available in the form of farm subsidies than there are farmers wishing to enroll in the particular program. In this case, the limiting factor is willing participants, not available money. Prioritization is not called for. The appropriate strategy would be to enlist every farmer that is willing to enroll in the program.

Based on economic theory, the prioritization criterion for comparatively ranking options is a benefit-cost ratio. Here, the benefit is ecological function, which could include general functions attributed to wetlands (e.g., flood retention, water quality improvement, and habitat) or specific functions, such as denitrification or bird habitat. The cost is defined with respect to the most limiting management resource, and could include money or hours of labor spent on capital outlays, operations, maintenance, and even opportunity cost. Specifically, the criterion is the marginal change in ecological function per management effort, dF/dE. The theoretical framework for this benefit/cost criterion is presented in Hyman and Leibowitz (2000). This prioritization criterion is valid under the following two conditions: First, total effort is constrained, as was previously discussed. And secondly, there is functional equivalence, meaning that a unit of function (e.g., a unit increase in biodiversity or a unit reduction in the concentration of a pollutant) at one location is equivalent to a unit at any other location. In other words, location has no effect on the benefit of a functional unit. This is necessary if ecological function is to act as a currency for comparison (although it should be noted that locational differences could be addressed through local weighting factors). The benefit-cost prioritization criterion can be applied to any situation where these two conditions are met. In the special case of geographic prioritization B where sites are not considered individually, but within the context of larger land units B there is the additional condition that there be a spatial trend to the variability in *dF/dE* (Hyman and Leibowitz 2000). Geographic factors that could influence *dF/dE* include geomorphology, ecoregion, and land use. Such spatial variability provides a basis for geographic prioritization. In practice, geographic prioritization may not be worthwhile if local variation dominates; i.e., variation in *dF/dE* within land units is much larger than variation between land units.

It should be noted that the prioritization criterion, *dF/dE*, is not total function (F) B either current, future, or past B nor is it an efficiency (F/E). Rather, it is a derivative (tangent) describing the instantaneous change in function given a change in effort. This prioritization criterion guarantees that the total level of function over all land units will be maximized after the management action. The use of this criterion may be somewhat counterintuitive. For example, we do not necessarily protect sites with the highest current function. Rather, we protect sites where the difference in function with and without protection would be highest. To illustrate this, consider two candidate sites for protection (Figure 1). Site 1 has the greatest current level of function (50 vs. 25), would have the larger function if protected (25 vs. 20), and has the greatest change in function between either current and protected (25 vs. 5) or current and developed (35 vs. 20). Selecting Site 1 for protection, based on any of these criteria, would result in $P_1 + D_2 = 25 + 5 = 30$ units of total function. However, the change in function between protection and development is greater for Site 2 (15 vs. 10). Selecting Site 2 would result in $D_1 + P_2 = 15 + 20 = 35$ units of function following protection. Selecting Site 2 for protection based on this prioritization criterion would therefore provide the greatest total function. Note that in this example we assume there is not complete protection, since permitting often allows development to proceed after plans are modified to minimize impacts.

The preceding example assumes that the amount of effort invested in protecting either site is equal. Thus we considered only the numerator of the benefit-cost criterion. But the denominator can also affect the outcome if costs are not equal. For example, consider a situation where restoring a site would provide a large change in function B 100 units B but at a high price tag of $25,000, due to property costs. If, for the same price, five lower cost sites could be restored that would each provide a 25 unit change, then total function would be maximized by restoring the five sites. Including the cost of restoration or protection in this manner is often omitted from consideration. Yet cost is often a major concern for program managers.

Constrained Information

In order to rank different locations using the prioritization criterion *dF/dE*, it is necessary to calculate this quantity for each of the land units to be compared. Unfortunately, wetland function is not easy to quantify, let alone the change in function. To calculate *dF/dE* directly would require a graph plotting function vs. effort for each land unit, so the slope could be determined. For geographic prioritization, this is further complicated because multiple wetlands must be considered in aggregate for each land unit. The situation facing managers is even more difficult, because typically there is only limited opportunity for obtaining new data. Thus the analysis must make use of existing information. The consequence of this information constraint is that there is rarely sufficient information to calculate *dF/dE* directly.

In cases where a quantity cannot be measured or calculated directly, it is possible to represent the quantity with indirect measurements of related variables, or *indicators*. With a *confirmed indicator*, the specific mathematical relationship between the indicator and the quantity of interest is known. Thus, the indicator can be used to numerically represent the quantity at some level of confidence. For example, if a regression equation was available relating wetland area to waterfowl abundance, then wetland area could be used as a confirmed indicator to quantitatively represent waterfowl abundance. In many cases, the mathematical relationship between an indicator and a quantity is not known. Such a related variable, which we refer to as a *judgement indicator*, does not allow for numerical estimation. However, it can be used to draw certain inferences if particular relational assumptions hold. If, for example, the statistical relationship between wetland area and waterfowl abundance was unknown, wetland area could still be used as a judgement indicator to make inferences about waterfowl. This is possible because years of observation show the two to be related. This could allow for relative comparisons, e.g., a wetland having twice as much area as a second wetland would have twice as many waterfowl as the other. Leibowitz and Hyman (1999) discusses the properties and use of judgement indicators.

Synoptic Approach

The Synoptic Approach was designed to use landscape scale indicators to prioritize wetland restoration or protection efforts. The use of this approach, or other assessment methods that utilize judgement indicators, is appropriate when (Abbruzzese and Leibowitz 1997): 1) quantitative, accurate information is not available, 2) the cost of obtaining or improving information is high, 3) the cost of a wrong answer is low, 4) there is a high demand for information, and 5) optimizing between multiple decisions (rather than a single decision) is desired.

Since its original development (Abbruzzese and Leibowitz 1997), the Synoptic Approach has been modified to incorporate the benefit-cost framework and make use of judgement indicators to deal with effort and information constraints (McAllister et al. 2000). Another modification is the use of a conceptual model to guide indicator selection. This was necessary because a number of problems are introduced when indicator selection is driven by practical issues of data availability (McAllister et al.

2000): 1) redundancies or correlations between variables are not identified, 2) there is no guidance on how to combine indicators, 3) variables may be included that are not ecologically relevant, and 4) it is difficult to determine whether important variables were omitted. The use of a conceptual model addresses these concerns. The model is based on our understanding of the relevant ecological processes that determine *dF/dE*. The purpose of the model is to formalize our ecological understanding, so as to guide indicator selection. The model is <u>not</u> developed for simulation, hypothesis testing, or direct analysis.

These modifications to the Synoptic Approach B the benefit-cost framework, conceptual models, and judgement indicators (Figure 2) B were utilized in two recent applications. McAllister et al. (2000) provide an assessment that prioritizes wetland restoration in the prairie pothole region to optimize regional flood attenuation. Schweiger et al. (2002) prioritize wetland protection efforts to minimize the risk of wetland species extirpation in EPA Region 7 (Iowa, Kansas, Missouri, and Nebraska). A third assessment is being finalized to prioritize restoration of headwater wetlands for reducing downstream sediment yields in EPA Region 4 (Alabama, Florida, Georgia, Kentucky, Mississippi, North Carolina, South Carolina, and Tennessee).

In the section AConstrained Effort,@ a number of conditions were discussed for the use of the benefit-cost criterion. This included the need for a spatial trend in the variability of *dF/dE* for the special case of geographic prioritization. Because the Synoptic Approach scores (ranks) all landscape units simultaneously, there is an additional condition for its use: the value of *dF/dE* for a landscape unit must be independent of the benefit that would be derived from protecting or restoring the other landscape units. For example, consider a basin which was ranked highly for floodplain restoration because of low water quality. If the source of pollution was an upstream basin, and if restoring wetlands in the upstream basin alleviated the water quality problem in the downstream basin, then the downstream basin should no longer receive a high ranking. In this case the value of *dF/dE* for the downstream basin is dependent on the benefit of restoring the upstream basin. In such cases the Synoptic Approach cannot be used. Instead, an approach that ranks units iteratively is needed (Hyman and Leibowitz 2000).

Conclusions

A benefit-cost approach for wetland prioritization could allow limited resources to be used to realize maximal functional gains. Because the information required for such an assessment is typically not available, geographic priorities must be approximated using judgement indicators. In interpreting results from any assessment B such as the Synoptic Approach B that uses judgement indicators, the following ABig Caveat@ should be kept in mind: Aresults should not be treated as empirical or field-tested findings. The conclusions of the assessment are based on judgement guided by scientific principles and a general understanding of the relevant ecological processes...Thus the results are somewhat akin to the conclusions of a scientist providing expert testimony at a trial@ (Schweiger et al. 2002). Such an assessment should not be treated as a final end product, but the results should be iteratively improved over time by testing assumptions, validating results, and substituting better data.

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Figure 1. Comparison of benefit criteria for two prospective protection sites. For Site 1, current function (C_1) is 50 units, function without protection from development (D_1) is 15 units, and function with partial protection (P_1) is 25 units. For Site 2, C_2, D_2, and P_2 are 25, 5, and 20 units, respectively.

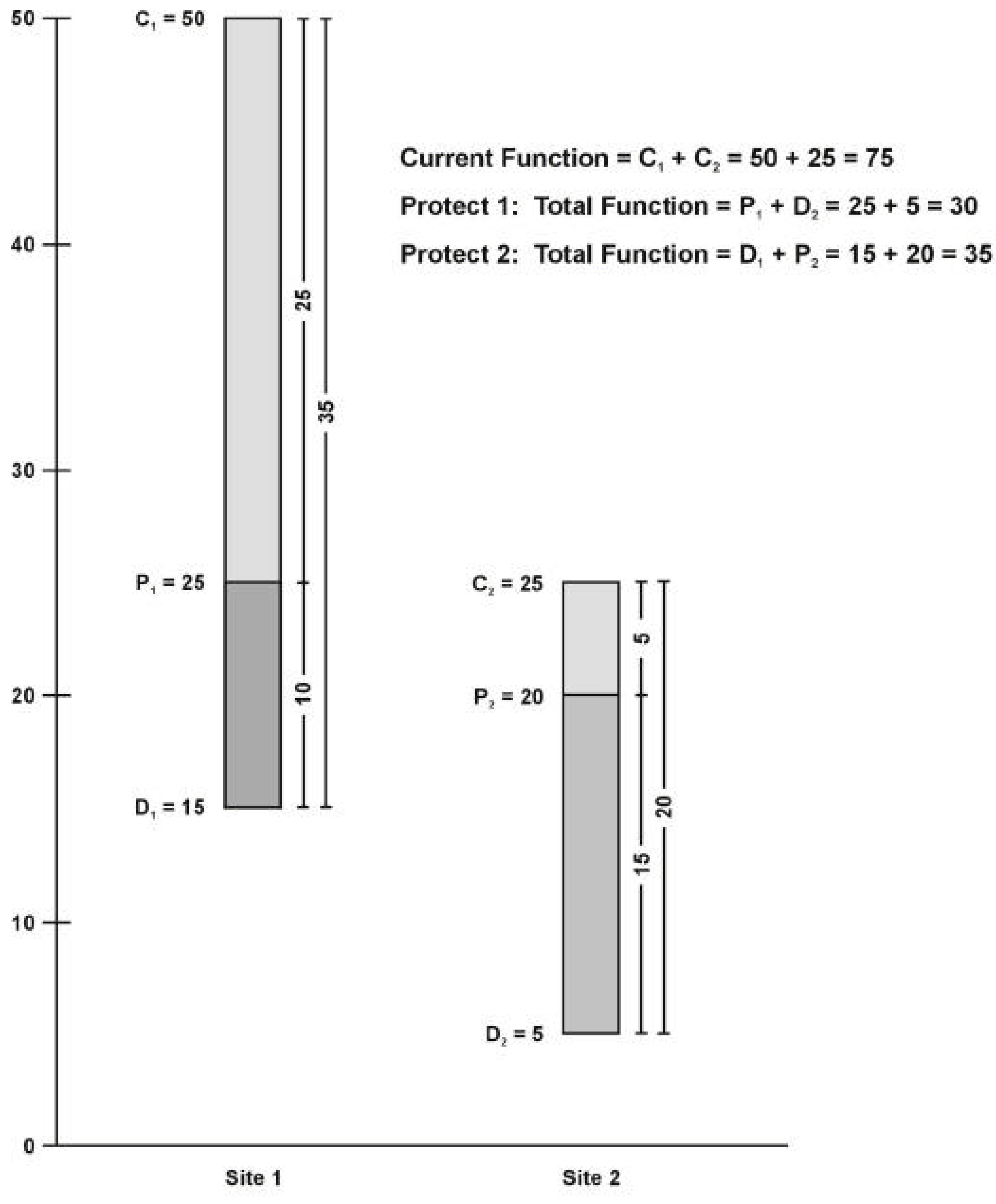
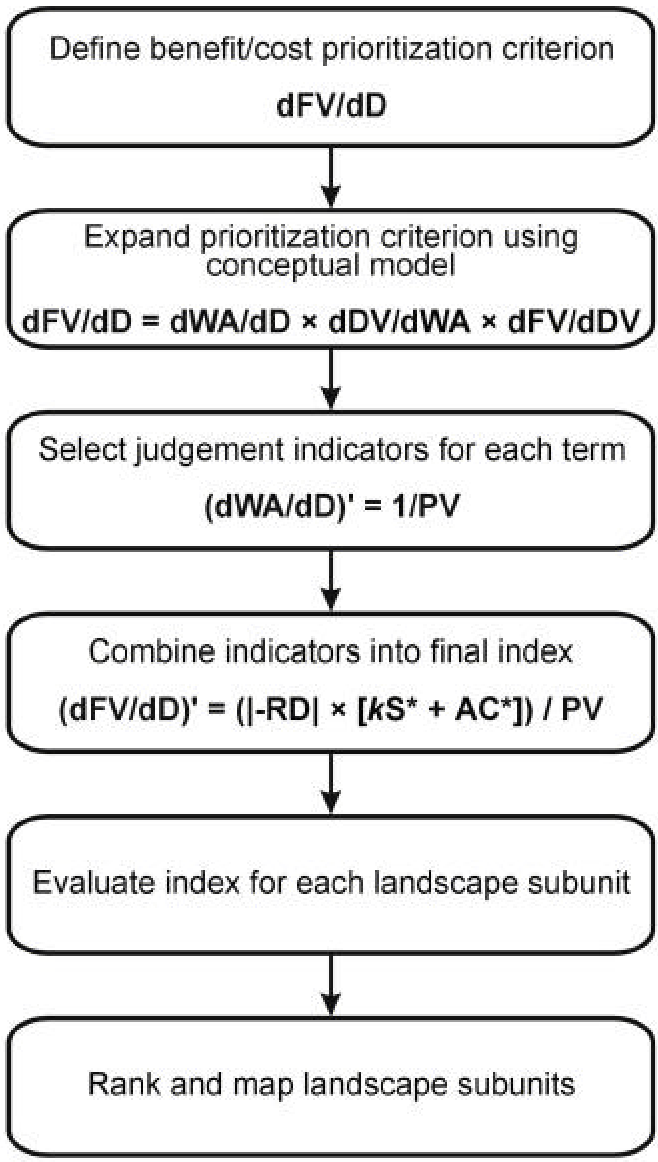


Figure 2. Steps in a synoptic assessment. dFV/dD = marginal decrease in total downstream flood volume per restoration dollar; dWA/dD = marginal increase in area of restored wetland per restoration dollar; dDV/dWA = marginal decrease in drainage volume per area of restored wetland; dFV/dDV = marginal decrease in total downstream flood volume per decrease in drainage volume; PV = farm property value; RD = total runoff depth; *k* = weighting and conversion coefficient; S = stream density; AC = artificial channel density. Indicators and standardized variables denoted by a prime and asterisk, respectively. Examples from McAllister et al. (2000).



Using Reference Wetlands for Integrating Wetland Inventory, Assessment, and Restoration for Watersheds

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Abstract

Watershed-based management of wetlands requires the integration of information obtained at various scales, from site-level assessments of wetland condition to landscape-level land use. We have developed such an approach that iteratively integrates information on the condition of individual wetlands and landscape factors resulting in an estimate of overall wetland health and restoration potential in the watershed. A key element of our approach is the use of reference wetlands that span a disturbance gradient from ecologically intact to severely impaired. This approach, with its associated assessment and management tools, can help resource management agencies and citizens make regulatory and non-regulatory decisions about watershed protection and restoration. We call this approach “Wetlands, Wildlife, and Watershed Assessment Techniques for Evaluation and Restoration” or W^{3}ATER - “the cube of water.”

Introduction

The Penn State Cooperative Wetlands Center has developed an approach to evaluate and protect wetlands on a watershed basis. The approach, called “Wetlands, Wildlife, and Watershed Assessment Techniques for Evaluation and Restoration” (W^{3}ATER) follows a standard planning process involving setting objectives, assessing the condition of the resource, prioritization, implementation, and evaluation. The W^{3}ATER approach also recognizes the need to acquire information for three separate, but integrated, tasks: inventory, condition assessment, and restoration. To accommodate differences in resource availability among various agencies and organizations, there are three levels of effort for each of the three tasks, with increasing confidence in decisions made based on effort. Our goal is to make this approach operational in the state of Pennsylvania, USA, during the next few years. A pilot application of the W^{3}ATER approach is being conducted in several watersheds to both begin the assessment process for those watersheds, and to train agency staff.

The Process

The W^3ATER approach begins with construction of a synoptic watershed map containing the best available wetlands inventory information. A synoptic map provides an overall visual representation of the watershed. We have modified the synoptic approach developed by Leibowitz et al. (1992) due to differences in the availability of remotely sensed data. We recommend that synoptic maps display at a minimum the most current land use and land cover data available. Although land use patterns do not completely describe disturbance levels, they are usually highly correlated (Brooks et al. 1996; Wardrop et al. 1998; O'Connell et al. 2000). A synoptic map provides a set of baseline conditions for comparing long-term changes, whether these changes involve degradation or restoration. The map can help identify potential landscape-level threats to parts of the watershed. Targeting of major projects, such as mitigation banks, can be facilitated.

Using a digital database for a synoptic map, a set of metrics for spatial analysis can be generated from GIS software programs to characterize the patterns of the landscape (e.g., proportional land cover and connectivity; Brooks et al. 1996; Miller et al. 1997). Recommended resources for developing synoptic maps include:

- current land use and land cover from Thematic Mapper (TM) satellite imagery
- stream network (digitized 1:24,000 blue line database)
- wetlands and waterbodies (National Wetlands Inventory digitized 1:24,000 base maps)
- road network (digitized 1:24,000 database)
- topography (Digital Line Graph [DLG] database)
- hydric and non-hydric soils (digitized county soil surveys as available, STATSGO)
- trends data (indicators of expected change, e.g., land use conversion rates, population growth rates, and intensity of landscape use).

Once the synoptic map is assembled, an assessment of wetland condition can occur using only this set of existing remotely-sensed data. The assessment conducted at Level 1 serves as a screening tool to focus on broad areas of concern within portions of the watershed, focusing primarily on proportions of land use around each designated wetland. If the Level 1 wetland inventory is insufficient or too outdated to conduct an assessment, we use landscape-based decision rules that identify areas of high probability for wetland occurrence in which to search. The latter requires ground reconnaissance to locate and classify individual wetlands, and results in an enhanced or Level 2 inventory. A Level 2 assessment combines the land use analysis from Level 1 with a characterization of the area adjacent to the wetland of interest and a checklist of stressors (e.g., sedimentation, eutrophication, see Adamus and Brandt 1990) observed during ground reconnaissance to determine the overall condition of the wetland.

Based on the results from a Level 1 or Level 2 assessment, an estimate of wetland condition becomes available for the target watershed. The estimate has wide confidence intervals. If this collective set of landscape and site indicators produces a problem or irregular disturbance "signal" within a specific area relative to an established reference condition, then a Level 3 assessment of hydrogeomorphic (HGM) functions and biological integrity (index of biological integrity [IBI]; Karr 1981; Karr and Chu 1999) can be used to diagnose specific stressors. Data collected during a Level 3 assessment are compared to existing set of reference wetlands of similar HGM type and condition. The data collection effort for a Level 3 assessment is substantial, and hence, is intended only for use in priority areas for protection and restoration.

Selection and Classification of Reference Wetlands

The use of reference wetlands is increasingly more common as ecologists and regulators search for a reasonable and scientific method to measure and describe the inherent variability in natural wetlands (Hughes et al. 1986; Kentula et al. 1992). Using reference wetlands from a wide variety of wetland types, disturbance regimes, and landscape positions allows for that characterization. Although reference sites often represent areas of minimal human disturbance, in some instances, it is more useful to represent a range of environmental conditions across a landscape. The primary reason for developing a set of reference sites is the need to compare impacted or degraded sites to a standard set of conditions. These baseline conditions can represent a starting point in time for trend analyses (e.g., long-term successional studies or impact analysis on a group of wetlands). Reference sites can also serve as alternatives to standard experimental controls that are seldom available. Reference sites provide the assessment criteria used for site evaluations. They can be used to set design standards for mitigation plans or to provide performance criteria to measure project success.

Sites within the reference set should span several gradients. They should include, at a minimum, the common types of wetlands found within a region, and range across the conditions found from relatively pristine (ecologically intact) to severely disturbed sites (degraded ecological integrity and functions). This will provide the data necessary to assess and rank the condition of other sites that are being assessed. If the measurement and establishment of baseline conditions are important for evaluating some anticipated impacts, then this could favor selection of sites either in degraded conditions facing further degradation, or sites with pristine conditions against which relatively minor impacts can be compared. The hydrogeomorphic (HGM) approach is based on characterization of reference wetlands across a wide range of conditions (Brinson 1993; Smith et al. 1995).

Given limited human and financial resources, creating a pool of reference wetlands that satisfies multiple objectives is desirable. Investigators must decide upon the acceptable level of analytical compromise they can tolerate versus the advantages of shared data and resources. Most studies will be able to benefit from some overlap among populations of reference sites. Once established, a set of reference wetlands can be used to set the standards by which wetland creation and restoration projects can be judged.

Since 1993, the Penn State Cooperative Wetlands Center (CWC) has established and studied more than 200 reference wetlands. Reference wetlands were chosen according to three criteria. First was long-term access, which suggested use of sites on public lands or on private lands with a written agreement from the landowner and an expectation of continued access if ownership changed. The CWC secured access agreements in all cases, with most sites being located on public lands. Second, the CWC emphasized wetland types and landscape settings that are most commonly impacted during the permitting process or prescribed under permit conditions. In general, these are HGM subclasses without significant amounts of open water. Third, sites were selected primarily at random. Randomized selection procedures should be followed during an assessment of wetlands for any given watershed. To ensure that all major HGM subclasses of wetlands were represented in our reference set, we selected individual wetlands from a pool of sites across a disturbance gradient. Also, part of our set of reference sites contained previously studied sites. Photographs and descriptions of many of our wetlands can be viewed on the web site at: www.wetlands.cas.psu.edu.

Based on the observed characteristics of our original set of 51 reference wetlands and preliminary information received during the evolution of the Corps' HGM program, we developed a regional HGM classification key for the inland freshwater wetlands of Pennsylvania, with further relevance to other Mid-Atlantic states. This dichotomous key is used to designate the HGM subclass based on examination of field characteristics (Brooks et al. 1996; Cole et al. 1997). Since classifying by HGM is not enough, one should link or modify regional HGM schemes to include wetland vegetation types (Cowardin et al. 1979) and disturbance levels. As with any classification system, there is overlap among subclasses, but if recognized, this aspect does not nullify the benefits of using W^3ATER. Professional judgment must be used to select the best possible match to a subclass. Most individual wetlands contain a mix of water sources and vegetation communities, and hence, will not result in perfect correspondence with reference subclasses. Usually a single HGM subclass will dominate a wetland site, but in some cases, two or more HGM subclasses will be present.

Recommended Steps for Establishing a Regional Set of Reference Wetlands

The following steps for establishing a regional set of reference wetlands are recommended. It is assumed that one of the primary uses of the reference set will be to classify wetlands and develop functional models using the HGM approach, but that other needs will be met by the same set.

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

- At some point, it will be necessary to determine whether all or only some HGM subclasses will be considered.
- Wetland types to be investigated can be prioritized by potential threats, relative abundance, or available expertise. We recommend that the assessment team identify a pool of wetlands at least 2-3 times the desired number of reference sites targeted for detailed characterization to account for access problems.

Further cautionary notes regarding selection of reference wetlands:

- Consider all needs for reference sites, not just for HGM functional assessment.
- One cannot always examine a statistically valid sample for each wetland type or HGM. subclass. Our rule of thumb is to use three wetlands as the absolute minimum per subclass, 30-50 is probably a maximum, and 8-12 begins to cover the variability in a subclass; Smith et al. (1995) suggest a minimum of 20.
- Sites can be chosen based on proportions of NWI types, or types of special concern.
- Sites should have long-term accessibility, which suggests public ownership, yet the reference set must cover site variability, including disturbance.
- A subset of the total reference set should met the requirement of long-term accessibility; this subset should consist of representative/typical wetlands.
- Once selected from the pool, secure written permission that acknowledges the probable sampling protocol and access procedures.

If implementing the HGM approach, begin to narrow the list of relevant functions (and their field measurements) for each subclass. This is a lengthy process involving significant discussions, field visits, and ultimately peer review before stakeholder acceptance will be forthcoming. When bounding reference wetlands, it will be necessary to truncate wetland complexes and mix types. The Penn State Cooperative Wetlands Center has selected sites in the range of about 0.25-3.0 ha, with most being about 0.4 ha.

Assessment Procedures for Wetlands

Numerous field sampling methods are available to assess the condition of wetlands being studied (e.g., Kentula et al. 1992; Gray et al. 1999). Our protocol for intensively sampling reference wetlands is also contained on the CWC web site (www.wetlands.cas.psu.edu) in the Adopt-a-Wetland section; contact the authors for minor changes in the sampling protocol. Intensive investigations are usually conducted to characterize the diversity of reference sites in a region. If resources are unavailable for intensive studies, then rapid assessment protocols (RAPs) applied by an experienced team for several wetlands in the vicinity may suffice. Once the data from reference sites are compiled and analyzed, then wetlands in target areas are assessed and compared to data from the reference sites.

Generally, RAPs require less intensive data collection than that collected for the reference sites. A RAP is selected with regard to how it can be used to identify the particular causes of problems observed in the watershed. Most problems can be categorized as being related to one or more stressors (Adamus and Brandt 1990). Knowing which stressors are impacting a wetland and its surroundings can help focus one's choice of an appropriate assessment tool. For example, the Wildlife Community Habitat Profile (WCHP, Brooks and Prosser 1995) was designed to be a

RAP that assesses potential wildlife habitat across a range of wetland types and conditions. The WCHP is strongly influenced by the structure of the vegetation, so any impact to those parameters will affect the rating or score. However, the WCHP would be a poor choice to assess water quality in riverine wetlands. The stream Habitat Assessment or Benthic Invertebrate Protocol would be more appropriate (Barbour et al. 1997). We anticipate that additional indices of biotic integrity or their equivalents will be forthcoming for a variety of taxa as efforts to develop ecological indicators continue.

The use of data from reference wetlands to develop and test HGM functional models should be viewed as an iterative process that continually improves over time as new data become available, and as the users become more experienced. As with the development of any tool, refinements should be both sought and anticipated.

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Enhancing Wetlands Inventory Data for Watershed-based Wetland Characterizations and Preliminary Assessments of Wetland Functions

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Abstract

Over the past 20 years, the U.S. Fish and Wildlife Service=s National Wetlands Inventory (NWI) Program has been mapping wetlands and building a geospatial wetland database for the country. Digital wetland data are now available for nearly half of the coterminous United States. This database is being expanded in selected areas to include hydrogeomorphic-type descriptors for mapped wetlands. These descriptors coupled with the existing NWI information make it possible to prepare wetland characterizations that include a preliminary assessment of wetland functions for watersheds and other geographic areas. The NWI Program has prepared a number of these characterizations over the past few years. The process of expanding the existing wetland inventory database and using it to prepare a preliminary assessment of wetlands is summarized. An example of a watershed-based wetland characterization is presented. This product provides valuable information for natural resource planners and will likely be in great demand to aid environmental planning efforts in the 21st century.

Introduction

Today there is great interest in managing wetland resources from a watershed standpoint or landscape perspective. Wetland managers need information on a variety of topics including the location and type of existing wetlands, wetland functions, potential wetland restoration sites, and the overall condition of natural habitat in the watershed. The U.S. Fish and Wildlife Service=s National Wetlands Inventory (NWI) Program has developed products that expand the use of its conventional maps and digital products to aid in resource management. In particular, the NWI has developed a procedure to improve and enhance existing NWI databases for providing additional characteristics for mapped wetlands that are important for assessing potential wetland functions.

The purpose of this paper is to describe the process used to enhance existing wetland inventory data, mainly NWI data, to generate information through geographic information system (GIS) technology for watershed-based wetland characterizations and preliminary assessments of wetland functions. An example for Maine=s Casco Bay watershed is given. This information will assist natural resource managers in wetland planning and evaluation at the watershed level.

Building a Comprehensive Geospatial Wetland Database

The foundation of this effort is construction of a fairly comprehensive, geospatial wetland database that allows use of GIS technology and integration with other geospatial datasets. Existing wetland digital data for most of the United States are NWI data. As of September 2001, 46 percent of the coterminous U.S. and 18 percent of Alaska have digital NWI data. Some states have more recent and detailed geospatial wetland data which can be used for the assessment, provided wetland classification is consistent with or can be converted to the federal wetland classification (Cowardin et al. 1979). Such data can also be used to update the NWI wetland database.

For wetland characterization and preliminary assessment of wetland functions, the NWI database is expanded to include hydrogeomorphic-type attributes for all mapped wetlands and waterbodies (Tiner 2000). The information contained within the database is then used to produce summary statistics, thematic maps, and a wetland characterization report for a given watershed. The report includes: 1) a summary of the extent and distribution of wetland types by NWI type and by hydrogeomorphic types and 2) a preliminary assessment of wetland functions for each watershed. More geospatial information can be added to the digital wetland database, including an inventory of potential wetland restoration sites, an assessment of the condition of wetland and waterbody buffers, an inventory of potential buffer restoration sites, and an evaluation of the extent of ditching.

Wetland Inventory and Classification

In preparing a watershed-based wetland characterization (including a preliminary assessment of wetland functions), the first step is to prepare a more complete and up-to-date wetland database.[1] At a minimum, a rapid-assessment revision of the NWI data is performed using a digital transfer scope (DTS). The DTS facilitates updating of existing digital data by allowing: 1) simultaneous viewing of digital wetland and hydric soil data with current-era aerial photography, and 2) editing of digital datasets based on this analysis. Utilizing hydric soils digital data helps expand the mapping of certain difficult-to-photointerpret wetlands, such as flatwood wetlands on the coastal plain, yet reliance on soils data may introduce commission errors (e.g., inclusion of upland forests in flatwood wetland polygons).

After compiling a more complete inventory of wetlands, hydrogeomorphic-type (HGM-type) information is added to each wetland in the NWI database. These attributes include landscape position, landform, water flow path, and other descriptors (Tiner 2000). To classify these HGM-types, NWI digital data are matched to on-line U.S. Geological Survey topographic maps (digital raster graphics

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[1] Many NWI maps contain information from the late 1970s and early 1980s and have not been updated.

obtained from www.gisdatadepot.com) and hydrologic data (digital line graphs from http://mcmcweb.er.usgs.gov/sdts/). Aerial photography is reviewed, where necessary.

HGM-type Attributes

Brinson (1993) developed a hydrogeomorphic wetland classification to aid in evaluating wetland functions. In his report, he refers to the classification as more of an approach rather than a strict classification system. Unfortunately, in developing this system, he used terms like lacustrine and riverine wetlands that are part of the federal wetland classification system (Cowardin et al. 1979) but defined them differently. This made it impossible to simply take his terms and apply them as additional descriptors for the Cowardin et al. system.

Recognizing the need to expand the Cowardin et al. (1979) system to include hydrogeomorphic properties reflected in Brinson (1993), Tiner (1995) prepared a set of hydrogeomorphic attributes that could be added to existing NWI data. These attributes have been revised after application in various study areas (Tiner 1997a, 2000) and will be modified in the future as necessary. During the past seven years, these descriptors have been added to NWI databases in selected areas (i.e., Massachusetts, Maine, Maryland, and New York) in the northeastern United States for watershed-based projects.

The HGM-type attributes of Tiner include three main categories: landscape position, landform, and water flow path. These features are used in conjunction with the Cowardin et al. properties (i.e., ecological system, subsystem, class [vegetation or substrate type], subclass, water regime [hydrology], water chemistry [pH and salinity/halinity], and special modifiers [e.g., beaver, diked/impounded, partly drained, and excavated]) to produce a more complete description of the characteristics associated with mapped wetlands and waterbodies (Tiner 1997b).

Landscape position defines the relationship between a wetland and an adjacent waterbody, if present. Five landscape positions are described: 1) marine (along the ocean and euhaline embayments), 2) estuarine (along brackish embayments and rivers), 3) lotic (along freshwater rivers and streams and their floodplains), 4) lentic (in lakes, reservoirs, and their basins), and 5) terrene (isolated, headwater, or fragments of former isolated or headwater wetlands that are now connected to downslope wetlands via drainage ditches). Lotic wetlands are further separated by river and stream gradients as high (e.g., shallow mountain streams on relatively steep slopes), middle (e.g., streams with moderate slopes), low (e.g., streams in relatively flat areas including mainstem rivers with considerable floodplain development), and tidal (i.e., under the influence of the tides). "Rivers" are separated from "streams" solely on the basis of channel width: watercourses mapped as linear (one-line) features on a 1:24,000 U.S. Geological Survey topographic map are designated as streams, whereas two-lined channels (polygonal features) are classified as rivers.

Landform is the physical form of a wetland or the predominant land mass on which it occurs (e.g., floodplain or interfluve). Seven types are recognized: basin, slope, interfluve, flat, floodplain, fringe, and island (see Table 1 for definitions). Wetlands on alluvial soils are considered to be floodplain wetlands.

Water flow path descriptors are assigned to indicate the type of directional flow of water associated with wetlands: throughflow, inflow, outflow, bidirectional flow, or isolated. Throughflow wetlands have either a watercourse or another type of wetland above and below it, so water flows through or over the subject wetland at times. All lotic wetlands are throughflow types, except for tidal ones (bidirectional flow). Inflow wetlands are sinks where no outlets exist, yet water is entering via a stream or river or an upslope wetland. Outflow wetlands have water leaving them and moving downstream via a watercourse or a slope wetland. Bidirectional flow pertains to situations where water levels fluctuate vertically due to tides or to changing lake levels. Isolated wetlands are essentially closed depressions or flats where water comes from surface water runoff and/or ground water discharge. Some isolated wetlands may have limited outflow during extremely wet conditions.

Other descriptors that are frequently applied include headwater, drainage-divide, and fragmented. *Headwater* wetlands are sources of streams or wetlands along first order (perennial) streams. They include wetlands connected to first order streams by ditching; these wetlands are also labeled with a ditched modifier. *Drainage-divide* wetlands are wetlands that occur in more than one watershed, straddling the defined watershed boundary line between the subject watershed and a neighboring one, or flowing into two different streams in the same watershed. An attempt at identifying *fragmented* wetlands is made. For this, wetlands separated by major highways (federal and state roads) and wetlands broken up by land development (e.g., farming) may be considered fragmented wetlands. In applying the fragmented descriptor, we attempt to cull out once larger wetlands that have been divided into smaller pieces. We do not apply the descriptor to wetlands that appeared to be simply reduced in size due to land use practices (e.g. drainage and conversion to farmland). The listing of fragmented wetlands is therefore conservative.

For open water habitats such as the ocean, estuaries, lakes, and ponds, we also apply additional descriptors following Tiner (2000). Different types of estuaries, lakes, ponds, rivers and streams may be identified. A dammed gradient was added to the river and stream types to include lock and dam situations, run-of-the-river dams, and other dams. Lake types include dammed valley, dammed, shallow, seasonal, and intermittent lakes. The ocean is separated into open ocean, reef-protected water, atoll lagoon, fjord, and semi-protected embayment. Estuary types include rocky headland bay, fjord, drowned river valley, barrier island back bay, coastal pond, hypersaline lagoon, barrier beach back bay, island-protected bay, and shoreline bay estuaries. Three tidal ranges are acknowledged: macrotidal (>4m), mesotidal (2-4m), and microtidal (<2m), and three hydrologic circulation patterns are recognized: salt-wedge, homogeneous, and partially mixed estuaries.

After expanding the classification of wetlands and deepwater habitats for a watershed, summary statistics and topical maps are prepared showing these different types. The digital database provides more information than the current NWI database and is especially useful for projecting functional capabilities for individual wetlands (see next subsection).

Table 1. Definitions and examples of landform types (Tiner 2000).

Landform	General Definition	Examples
Basin*	a depressional (concave) landform	lakefill bogs; wetlands in the saddle between two hills; wetlands in closed or open depressions, including narrow stream valleys; tidally restricted marshes
Slope	a landform extending uphill	seepage wetlands on hillside; wetlands along drainageways or mountain streams on slopes
Flat*	a relatively level landform, often on broad flat landscapes	wetlands on stream terraces; wetlands on hillside benches and toes of slopes
Floodplain	a broad, generally flat landform occurring on a landscape shaped by fluvial or riverine processes	wetlands on alluvium; bottomland swamps
Interfluve	a broad, level to imperceptibly depressional poorly drained landform on coastal or glacio-lacustrine plains occurring between two drainage systems (on interstream divides)	flatwood wetlands
Fringe	a landform occurring along a flowing or standing waterbody (lake, river, stream) and typically subject to permanent, semi-permanent flooding or frequent tidal flooding; including wetlands within stream or river channels**	buttonbush swamps; aquatic beds; nonpersistent emergent wetlands; salt and brackish tidal marshes; gravel bars; mudflats; beaches
Island	a landform completely surrounded by water (including deltas)	deltaic and insular wetlands; floating bogs

*May be applied as sub-landforms within the Interfluve and Floodplain landforms.
** Includes temporarily flooded cobble-gravel bars forming river and stream banks.

Preliminary Assessment of Wetland Functions

With the improved NWI digital database in-hand, several analyses are performed to produce a preliminary assessment of wetland functions for the watershed. Ten wetland functions may be evaluated, including: 1) surface water detention, 2) streamflow maintenance, 3) nutrient transformation, 4) sediment and other particulate retention, 5) coastal storm surge detention and shoreline stabilization, 6) inland shoreline stabilization, 7) fish and shellfish habitat, 8) waterfowl and waterbird habitat, 9) other wildlife habitat, and 10) biodiversity. The rationale for correlating wetland characteristics with wetland functions is described in each watershed report. After running the analyses, a series of maps for watershed is produced to highlight wetland types that may perform these functions at high or other significant levels. Statistics and topical maps for the study area are generated by ArcView software.

At the outset, it is important to emphasize that this functional assessment is a preliminary one based on wetland characteristics interpreted through remote sensing and using the best professional judgment of participating individuals. Wetlands believed to be providing potentially high or other significant levels of performance for a particular function are highlighted. As the focus of this analysis is on wetlands, an assessment of deepwater habitats (e.g., lakes, rivers, and estuaries) for providing the listed functions is not done (e.g., it is rather obvious that such areas provide significant functions like fish habitat). Also, no attempt is made to produce a more qualitative ranking for each function or for each wetland based on multiple functions as this would require more input from others and more data, well beyond the intent of this preliminary analysis.

Functional assessment of wetlands can involve many parameters. Typically such assessments have been performed in the field on a case-by-case basis, considering observed features relative to those required to perform certain functions or by actual measurement of performance. Our preliminary analysis does not seek to replace the need for such evaluations as they are the ultimate assessment of the functions for individual wetlands. Yet, for a watershed analysis, basinwide field-based assessments are not practical or cost-effective or even possible given access considerations. For watershed planning purposes, a more generalized assessment is worthwhile for targeting wetlands that may provide certain functions, especially for those functions dependent on landscape variables, vegetation lifeform, and hydrologic regimes. Subsequently, these results can be field-verified when it comes to actually evaluating particular wetlands for acquisition purposes, e.g., for conservation of biodiversity or for preserving its flood storage function. More up-to-date aerial photography may also be examined to aid in further evaluations (e.g., condition of wetland/stream buffers or adjacent land use) to supplement the preliminary assessment.

This analysis employs a watershed assessment approach that may be called "Watershed-based Preliminary Assessment of Wetland Functions" (W-PAWF). W-PAWF applies general knowledge about wetlands and their functions to develop a watershed overview that highlights wetlands predicted to perform various functions at significant levels. To accomplish this objective, the relationships between wetlands and various functions must be simplified into a set of practical criteria or observable characteristics. Such assessments may also be further expanded to consider the condition of the

associated waterbody and the neighboring upland or to evaluate the opportunity a wetland has to perform a particular function or service to society, for example.
W-PAWF usually does not account for the opportunity that a wetland has to provide a function resulting from a certain land-use practice upstream or the presence of certain structures or land-uses downstream. For example, two wetlands of equal size and like vegetation may be in the right landscape position to retain sediments. One, however, may be downstream of a land-clearing operation that has generated considerable suspended sediments in the water column, while the other is downstream from an undisturbed forest. The former should be actively performing sediment trapping in a major way, while the latter is not. Yet if land-use conditions in the latter subbasin change, the second wetland will likely trap sediments as well as the first wetland. The entire analysis typically tends to ignore opportunity since such opportunity may have occurred in the past or may occur in the future and the wetland is awaiting a call to perform this service at higher levels than presently. An exception would be for a wetland type that would not normally be considered significant for a particular function (e.g., sediment retention), but due to current land use of adjacent areas (e.g., tilled with ditches entering the wetland), it now receives substantial sediment input and thereby performs the function at a significant level.

W-PAWF also does not consider the condition of the adjacent upland (e.g., level of disturbance) or the actual water quality of the associated waterbody. These features are undoubtedly important metrics for assessing the health of individual wetlands.

It is further emphasized that the preliminary assessment does not obviate the need for more detailed assessments of the various functions. This assessment should be viewed as a starting point for more rigorous assessments, as it attempts to cull out wetlands that may likely provide significant functions based on generally accepted principles and the source information used for the analysis. This type of assessment is most useful for regional or watershed planning purposes. For site-specific evaluations, additional work will be required, especially field verification and collection of site-specific data for potential functions (e.g., following the HGM assessment approach as described by Brinson 1993 and other onsite evaluation procedures). This is particularly true for assessments of fish and wildlife habitats and biodiversity. Other sources of data may exist to help refine some of the findings of this report (e.g., natural heritage program data for biodiversity). Additional modeling could be done, for example, to identify habitats of likely significance to individual species of animals (based on their specific life history requirements).

Rationale for Preliminary Functional Assessments

To date, ten functions are evaluated, as appropriate, for study watersheds: 1) surface water detention, 2) streamflow maintenance, 3) nutrient transformation and recycling, 4) sediment and other particulate retention, 5) coastal storm surge detention and shoreline stabilization (for coastal watersheds), 6) inland shoreline stabilization, 7) provision of fish and shellfish habitat (coastal and inland), 8) provision of waterfowl and waterbird habitat, 9) provision of other wildlife habitat, and 10) conservation of biodiversity. The criteria used for identifying these functions for querying the digital wetland database may vary from place to place for some of the functions, yet the criteria for a few functions have virtual universal application. Examples of the rationale used for coastal Maryland are outlined in Table 2 (for more detailed information see the subject report: "Watershed-based Wetlands Characterization for Maryland=s Nanticoke and Coastal Bays Watersheds: A Preliminary Assessment Report" [Tiner et al. 2000b], on the web at: wetlands.fws.gov.). The criteria were developed by the senior author of the report based on previous work and were reviewed and modified for the subject watersheds based on comments from U.S. Fish and Wildlife Service field personnel and specialists from the Maryland Department of Natural Resources.

In developing a protocol for designating wetlands of potential significance, wetland size is generally disregarded from the criteria, with some possible exceptions (e.g., surface water detention, other wildlife habitat, and biodiversity functions). This approach is followed because it was felt that state agencies and others using the digital database and charged with setting priorities should make the decision on appropriate size criteria as a means of limiting the number of priority wetlands, as necessary. Our intent is to present a more expansive characterization of wetlands and their likely functions and not to develop a rapid assessment method for ranking wetlands for acquisition, protection, or other purposes.

Table 2. Examples of correlations between wetland characteristics and wetland functions for wetlands in the Nanticoke watershed of Maryland. Predicted potential (e.g., High, Moderate-High, Some, and Local) is also noted.

Function	**Wetland Characteristics for Significant Performance**
Surface Water Detention	High: Lotic floodplain and basin wetlands Moderate-High: Terrene wetlands (≥50 acres) Some: Lotic flat wetlands; nonditched terrene wetlands (20-50 acres)
Streamflow Maintenance	High: Terrene headwater wetlands Moderate-High: Lotic headwater and lotic floodplain wetlands
Nutrient Transformation and Recycling	High: Lotic wetlands on organic-rich soils or having a seasonally flooded or wetter water regime; estuarine fringe wetlands (vegetated) Some: Lotic flat wetlands; terrene outflow wetlands surrounded by cropland (>50% of perimeter)
Sediment and Other Inorganic Particulate Retention	High: Lotic floodplain, fringe, and basin wetlands; estuarine fringe and island wetlands (vegetated and nonvegetated) Some: Lotic flat wetlands; terrene outflow wetlands surrounded by cropland (>50% of perimeter) Local: Isolated ponds
Coastal Storm Surge Detention/Shoreline Stabilization	High: Estuarine vegetated wetlands; seasonally flooded-tidal palustrine vegetated wetlands Moderate-High: Palustrine nontidal wetlands bordering the above types Some: Estuarine nonvegetated wetlands
Inland Shoreline Stabilization	High: Lotic vegetated wetlands (except island wetlands)
Coastal Fish and Shellfish Habitat	High: Estuarine submerged aquatic beds, tidal flats, and emergent wetlands

Freshwater Fish and Shellfish Habitat	High: Palustrine and riverine tidal emergent wetlands and tidal flats; Palustrine nontidal semipermanently flooded wetlands and aquatic beds Also Important: Lotic stream wetlands that were forested Some: Ponds and shallow marsh-open water zone of impoundments
Waterfowl and Waterbird Habitat	High:Estuarine and riverine emergent wetlands; estuarine mixed emergent/shrub wetlands; estuarine and riverine tidal flats; palustrine tidal emergent wetlands; palustrine semipermanently flooded wetlands; palustrine and lacustrine mixed open water-emergent wetlands;aquatic beds Important for Wood Ducks: Seasonally flooded lotic wetlands (forested or mixed forested/shrub types); palustrine tidal deciduous forested wetlands (seasonally flooded-tidal and semipermanently flooded-tidal types with mixtures of other vegetative life forms) Some: Ponds
Other Wildlife Habitat	Notables: Large wetlands (≥20 acres); smaller diverse wetlands (10-20 acres with multiple cover types)
Biodiversity	Notables: Uncommon types in watershed; riverine tidal and estuarine oligohaline wetlands; wetlands within a contiguous 7,410 acre region of forest; estuarine aquatic beds; selected large wetlands

An Example of a Watershed-based Wetland Characterization

The U.S. Fish and Wildlife Service has published watershed-based wetland characterizations for several watersheds in the northeastern United States (e.g., Casco Bay watershed in Maine; small watersheds in New York; Nanticoke and Coastal Bays watersheds in Maryland; Nanticoke watershed in Delaware). Similar work is in progress for numerous small watersheds in New York (including the New York City water supply system) and for Pennsylvania=s coastal zone region. The following example comes from the Casco Bay watershed report (Tiner et al. 1998), the first such report published and gives readers a good look at the kinds of information that can be generated from this type of work. For a more recent example, see the Maryland watersheds report posted on the web at: wetlands.fws.gov (go to publications, then to the report on Maryland=s Nanticoke and Coastal Bays watersheds, for pdf files of report and separate files for maps).

Wetland Inventory and Classification for the Casco Bay Watershed

The initial step in preparing a watershed-based wetland characterization is to conduct a wetland inventory for the study watershed. For the Casco Bay project, wetlands were previously classified according to the U.S. Fish and Wildlife Service's official wetland classification system (Cowardin et al. 1979) by the Service=s National Wetlands Inventory (NWI) Program. Maps at a scale of 1:24,000 showing NWI wetlands were prepared and digitized to create a digital wetland database in the late 1980s/early 1990s. This database was the foundation for the Casco Bay watershed analysis. Classification of each vegetated wetland was expanded to include hydrogeomorphic-type attributes (e.g., landscape position, landform, and water flow path; HGM types) following Tiner (1997a). Data are summarized below (see Tiner et al. 1998 for details).

Wetlands by NWI Types

According to the NWI=s 1980s-era wetland inventory, the Casco Bay watershed had nearly 9,500 wetlands totaling 46,681 acres (Table 3). Palustrine wetlands were the most abundant types with over 35,500 acres inventoried. Freshwater swamps, bogs, marshes, and ponds represented about 76 percent of the watershed's wetlands. Estuarine wetlands accounted for only 14 percent of the wetlands (about 6,500 acres), while marine wetlands represented about 10 percent (about 4,600 acres). Only 13.5 acres of lacustrine wetlands (unconsolidated shore) were inventoried. Aquatic beds and nonpersistent emergent wetlands that may be associated with some lakes were not detected due to use of spring aerial photos for NWI mapping (high water and no visible leaf cover).

Forested wetlands were the predominant palustrine type in the watershed accounting for about 56 percent of the palustrine wetlands (excluding dead forested wetlands that were mainly shallow water wetlands). Scrub-shrub wetlands were next in abundance, representing about 26 percent of the palustrine wetlands. Emergent wetlands (including shrub/emergent mixtures) made up nearly 13 percent. The remaining palustrine wetlands were ponds (unconsolidated shores).

Estuarine wetlands were dominated by tidal flats (unconsolidated shores) which comprised about 74 percent of these wetlands. Salt marshes (emergent) represented about 23 percent of the estuarine wetlands. The remainder were either aquatic beds (mostly rocky shores vegetated by fucoid algae; 3.3%) or nonvegetated rocky shores (0.3%).

Marine wetlands were mostly tidal flats (57 percent) and aquatic beds (34%; mostly vegetated rocky shores). Nonvegetated rocky shores accounted for about 9 percent of the marine wetlands. Mussel reefs comprised about 0.2 percent of the marine wetlands in Casco Bay.

Table 3. Wetlands in the Casco Bay watershed, southern Maine classified by NWI wetland type to the class level (Cowardin et al. 1979). Other modifiers (e.g., beaver, diked/impounded, partly drained) have been deleted from NWI types for this compilation. (Source: Tiner et al. 1998)

NWI Wetland Type	**Acreage**
Marine Wetlands	
Aquatic Bed	1550.4
Reef	9.4
Rocky Shore	417.1
Unconsolidated Shore	2625.7
--------------------------	--------
Subtotal	4602.6
Estuarine Wetlands	
Aquatic Bed	215.7
Emergent	1491.7
Rocky Shore	18.7
Unconsolidated Shore	4799.2
--------------------------	--------
Subtotal	6525.3
Lacustrine Wetlands	
Unconsolidated Shore	13.5
-------------------------	--------
Subtotal	13.5
Palustrine Wetlands	
Aquatic Bed	8.3
Emergent (Nontidal)	3260.7
Emergent (Tidal)	64.6
Emergent/Scrub-Shrub (Nontidal)	1101.5
Emergent/Scrub-Shrub (Tidal)	49.7

Broad-leaved Deciduous Forested (Nontidal)	6944.1
Broad-leaved Deciduous Forested (Tidal)	17.6
Needle-leaved Deciduous Forested	3.4
Needle-leaved Evergreen Forested (Nontidal)	6632.4
Needle-leaved Evergreen Forested (Tidal)	75.3
Mixed Forested (Nontidal)	5494.6
Mixed Forested (Tidal)	2.4
Forested/Emergent	120.4
Evergreen Forested/Scrub-Shrub	432.7
Deciduous Forested/Scrub-Shrub	107.6
Dead Forested	154.7
Deciduous Scrub-Shrub (Nontidal)	6736.8
Deciduous Scrub-Shrub (Tidal)	79.2
Broad-leaved Evergreen Scrub-Shrub	370.3
Needle-leaved Evergreen Scrub-Shrub (Nontidal)	419.2
Needle-leaved Evergreen Scrub-Shrub (Tidal)	5.6
Evergreen Scrub-Shrub (unspecified/Nontidal)	155.9
Mixed Scrub-Shrub (Nontidal)	1292.3
Mixed Scrub-Shrub (Tidal)	8.7
Unconsolidated Bottom (Nontidal)	1986.5
Unconsolidated Bottom (Tidal)	14.8
Subtotal	35,539.3
GRAND TOTAL (all wetlands)	46,680.7

Hydrogeomorphic-Type Wetlands

Most of the wetlands in the Casco Bay watershed were terrene wetlands, principally headwater and isolated types. Terrene wetlands accounted for 66 percent of the wetlands by number, yet only 38 percent of the wetland acreage classified to hydrogeomorphic (HGM) type (Table 4). This contrast means that many of the terrene wetlands were rather small. Lotic wetlands ranked second in abundance (2,105 wetlands; 26 percent of the number of wetlands), but first in acreage totaling 19,364 acres, accounting for 52 percent of the wetland acreage (that was classified by HGM-type descriptors) in the watershed. This suggests that lotic wetlands were, on average, much larger in size than the terrene wetlands. Estuarine wetlands (essentially vegetated types) represented almost 5 percent of the watershed's wetlands by number and by acreage. Wetlands associated with lakes -- lentic wetlands -- comprised nearly 4 percent by number and about 5 percent by acreage.

From the landform perspective, basin wetlands were most abundant, accounting for 84 percent of the wetlands by number (6,826) and about 74 percent of the total acreage (27,354 acres). Due to the fact that most estuarine vegetated wetlands are fringe types, the fringe wetlands were second-ranked in regard to number (484 wetlands or 6% of the wetlands), yet fourth-ranked in acreage (2,227 acres or 6%). Floodplain wetlands were second in acreage (with 2,814 acres or 8%) and fourth in number (244 or 3%). Slope wetlands were third-ranked in both categories (438 wetlands or 5%; 2,702 acres or 6%). Fifth-ranked in both categories were flat wetlands (108 or 1%; 2,072 acres or 6%). Island wetlands were poorly represented -- 26 wetlands (9 estuarine; 3 lotic river; 14 lentic) for a total of 21 acres.

Considering water flow path for freshwater wetlands, five types were recognized: 1) inflow, 2) outflow, 3) throughflow, 4) bidirectional flow (associated with lakes), and 5) isolated. Isolated wetlands were most numerous (4,255 wetlands), representing 55 percent of the freshwater wetlands. These wetlands, however, occupied only 6,171 acres or 17 percent of the acreage. Most of the freshwater wetland acreage (19,716 acres; 56%) was composed of throughflow wetlands, mainly associated with rivers and streams. These wetlands accounted for about 28 percent of the number of freshwater wetlands in the Casco Bay watershed (2,141 wetlands). Outflow wetlands made up about 22 percent of the freshwater wetland acreage (7,620) and almost 13 percent of the wetlands by number (978). Bidirectional flow wetlands associated with lakes comprised about 4 percent by number (312) and nearly 5 percent by acreage (1,688 acres). Inflow wetlands were scarce representing almost 1 percent by number (67) and about 0.5 percent by acreage (189 acres).

Characterization Maps for the Casco Bay Watershed

A series of 15 maps were produced at 1:130,000 to profile the Casco Bay watershed=s wetlands. These maps were distributed to the Maine State Planning Office. The 15 maps addressed the following themes: 1) NWI types, 2) wetlands by landscape position, 3) wetlands by landform, 4) estuarine and marine wetlands by landscape position and landform, 5) inland wetlands by landscape position and landform, 6) wetlands and surface water detention, 7) wetlands and streamflow maintenance, 8) wetlands and nutrient cycling, 9) wetlands and sediment/particulate retention, 10) coastal wetlands and storm surge detention/shoreline stabilization, 11) wetlands and inland shoreline stabilization, 12) wetlands and fish habitat, 13) wetlands and waterfowl/waterbird habitat, 14) wetlands and other wildlife habitat, and 15) wetlands and biodiversity. A few of the maps are presented in this paper for illustration purposes (see Figures 1-4 at end of report).

Table 4. Estuarine and freshwater wetlands in the Casco Bay watershed, southern Maine classified by landscape position, landform, and water flow path (Tiner 1997a). Note: Most nonvegetated estuarine wetlands were not classified by these descriptors as the emphasis on this characterization was largely based on vegetated types, especially in the marine and estuarine systems. The nonvegetated estuarine wetlands, namely intertidal flats and rocky shores, are "fringe" wetlands.

Landscape Position	**Landform**	**Water Flow**	**# of Wetlands**	**Acreage**
Terrene			5336	14281.4
	Slope		224	1602.1
		Inflow	10	52.6
		Isolated	84	391.2
		Outflow	114	1055.7
		Throughflow	16	102.6
	Basin		5104	12473.6
		Inflow	57	136.5
		Isolated	4171	5779.6
		Outflow	856	6358.2
		Throughflow	20	199.3
	Flat	Outflow	8	205.7
Lentic			312	1688.3
	Basin	Bidirectional	199	1285.9
	Fringe	Bidirectional	99	390.8
	Island	Bidirectional	14	11.6
Lotic River			324	3582.8
	Basin	Throughflow	91	1817.4
	Flat	Throughflow	11	169.4
	Floodplain	Throughflow	217	1589.8
	Fringe	Throughflow	2	1.0
	Island	Throughflow	3	5.2
Lotic Stream			1781	15831.1
	Basin	Throughflow	1408	11639.1
	Flat	Throughflow	89	1697.1
	Floodplain	Throughflow	27	1224.3
	Fringe	Throughflow	43	171.1
	Slope	Throughflow	214	1099.5
Estuarine			373	1805.9
	Basin	Bidirectional	24	137.6
	Fringe	Bidirectional	340	1664.3

Island	Bidirectional	9	4.0

Preliminary Functional Assessment Results for the Casco Bay Watershed

The results of the preliminary functional assessment for the Casco Bay watershed are shown in Table 5. Wetlands performing various functions at notable levels ranged from 92 percent of the watershed=s wetlands (for sediment and other particulate retention) to 4 percent (for coastal storm surge detention and shoreline stabilization). Wetlands predicted as important for habitat included: 59 percent of the watershed=s wetlands for fish habitat, 74 percent for waterfowl and waterbirds, 51 percent for other wildlife, and 22 percent for biodiversity.

Table 5. Summary of wetlands predicted to perform certain functions at notable levels for the Casco Bay watershed, Maine.

Function	**Predicted Level of Significance**	**% of Wetland Acreage**
Surface water detention	high	53
	local	12
Streamflow maintenance	high	25
Nutrient transformation	high	51
Sediment/particulate retention	high	49
	moderate	27
	local	16
Coastal storm surge detention/shoreline stabilization	high	4
Inland shoreline stabilization	high	56
Fish habitat	high (coastal)	26
	some (coastal)	3
	high (inland)	1
	some (inland)	29
Waterfowl and waterbird habitat	mod-high (coastal)	24
	high (inland)	2
	some	48

Other wildlife habitat	high	51
Conservation of biodiversity	high	22

Use of Characterization Reports

The watershed-based wetland characterization report provides a basic portrayal of wetlands in a watershed including a preliminary assessment of wetland functions. The results are a first-cut or initial screening of a watershed's wetlands to designate wetlands that may have a significant potential to perform various functions. The targeted wetlands have been identified as being predicted to perform a given function at a significant level presumably important to a watershed's ability to provide that function. "Significance" is a relative term and is used in this analysis to identify wetlands that are likely to perform a given function at a level above that of wetlands not designated. The report is useful for general natural resource planning, as an initial screening for considering prioritization of wetlands (for acquisition, restoration, or strengthened protection), as an educational tool (e.g., helping the public and nonwetland specialists better understand the functions of wetlands and the relationships between wetland characteristics and performance of individual functions), and for characterizing the differences among wetlands in terms of both form and function within a watershed.

While the results are useful for gaining an overall perspective of the watershed's wetlands and their relative importance in performing certain functions, the report does not identify differences among wetlands of similar type and function. The latter information is often critical for making decisions about wetland acquisition and designating certain wetlands as more important for preservation versus others with the same classification. Additional information may be gained through consulting with agencies having specific expertise in a subject area and by conducting field investigations to verify or refine the preliminary assessments as needed. When it comes to actually acquiring wetlands for preservation, other factors must be considered. Such factors may include: 1) the condition of the surrounding area, 2) the ownership of the surrounding area and the wetland itself, 3) site-specific assessment of wetland characteristics and functions, 4) more detailed comparison with similar wetlands based on field investigations, 5) proximity to existing public lands, and 6) advice from other agencies (federal, state, and local) with special expertise on priority resources (e.g., for wildlife habitat, contact appropriate federal and state biologists). The latter agencies may have site-specific information or field-based assessment methods that can aid in further narrowing the choices to help insure that the best wetlands are acquired for the desired purpose.

The value of watershed-based wetland characterization is recognized by several states that have provided funding for this work. Massachusetts uses the HGM-type descriptors to help predict the likely functions of potential wetland restoration sites in various watersheds. Maine is using the Casco Bay characterization report to develop a strategy for improving wetland protection and conservation in that high priority watershed. The report and accompanying maps are also used to help educate the public on the values of wetlands and to help build a constituency for strengthening wetland protection. In his book AEcologically Based Municipal Land Use Planning,@ Honachefsky (2000) specifically referenced the Casco Bay wetland characterization project as an example of the type of information that is valuable to land planners in compiling an accurate ecologically based municipal master plan. New York City Department of Environmental Protection is using this type of information to aid in protecting the City=s water supply. The Maryland Department of Natural Resources (MDDNR) is using the wetland characterization report for two watersheds (Coastal Bays and Nanticoke) to add to their GIS tools for their Green Infrastructure Assessment which is used by resource and land management decision-makers within MDDNR and outside of the agency for natural resource planning. In particular, the characterization brings wetland functional information into their GIS framework which greatly increases its capability. The State of Delaware has funded a similar effort for its portion of the Nanticoke watershed to help them with watershed assessment and management. The U.S. Environmental Protection Agency (Southeast Region) has initiated work to expand the existing NWI digital database to include HGM-type attributes. This will help them identify priority wetland resources across the Southeast. The use of GIS to identify wetland functions is a new form of wetland assessment that is gaining more support (see North Carolina Department of Environment, Health, and Natural Resources 1996; Cedfeldt et al. 2000; Sutter 2001). Watershed-based wetland characterizations are new GIS-based tools for natural resource managers and should facilitate the development of strategies to improve wetland conservation and management across the country and around the globe.

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Figure 1. Watershed map showing wetlands by NWI type for the Casco Bay watershed.

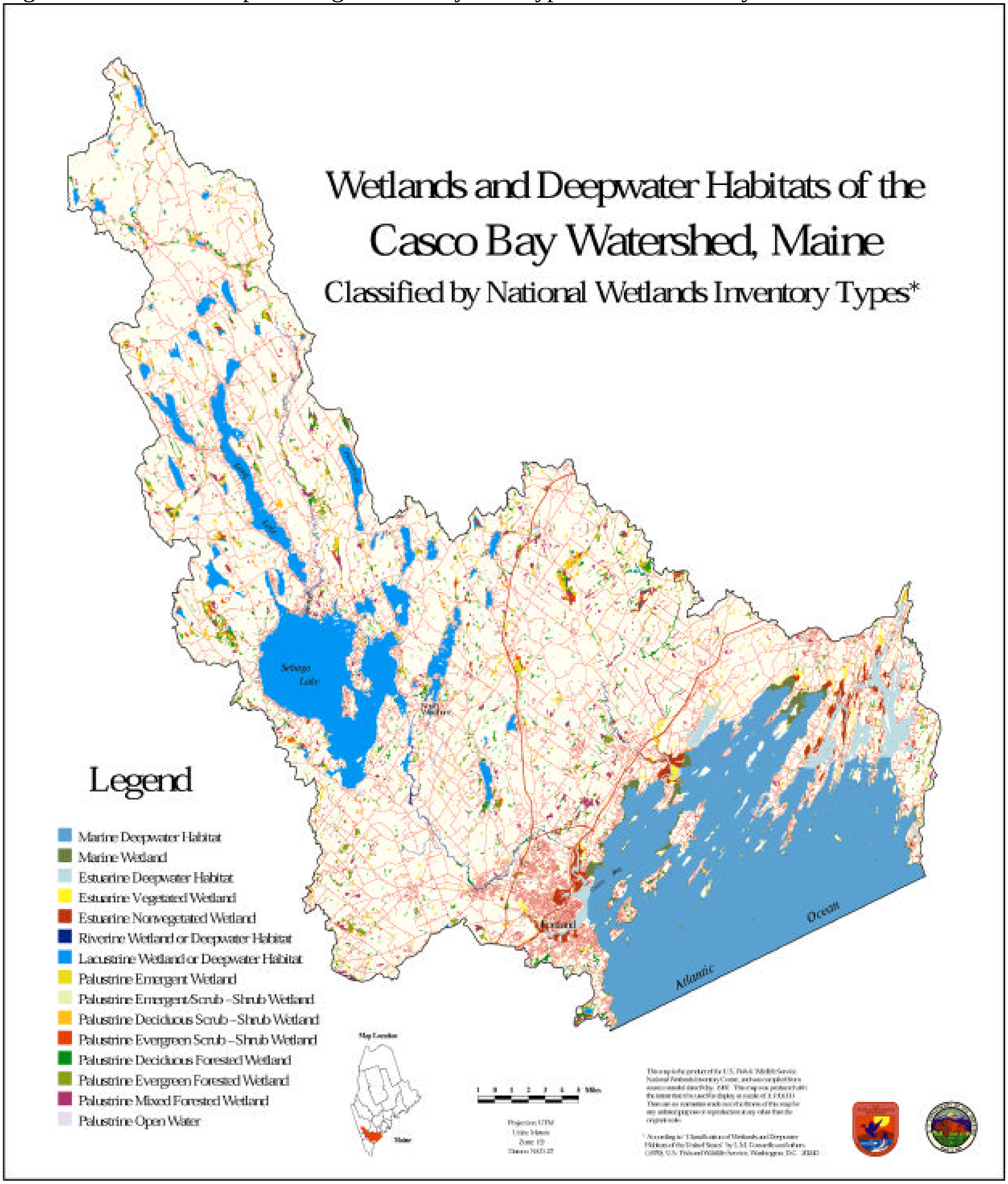


Figure 2. Watershed map showing wetlands by landscape position for the Casco Bay watershed.

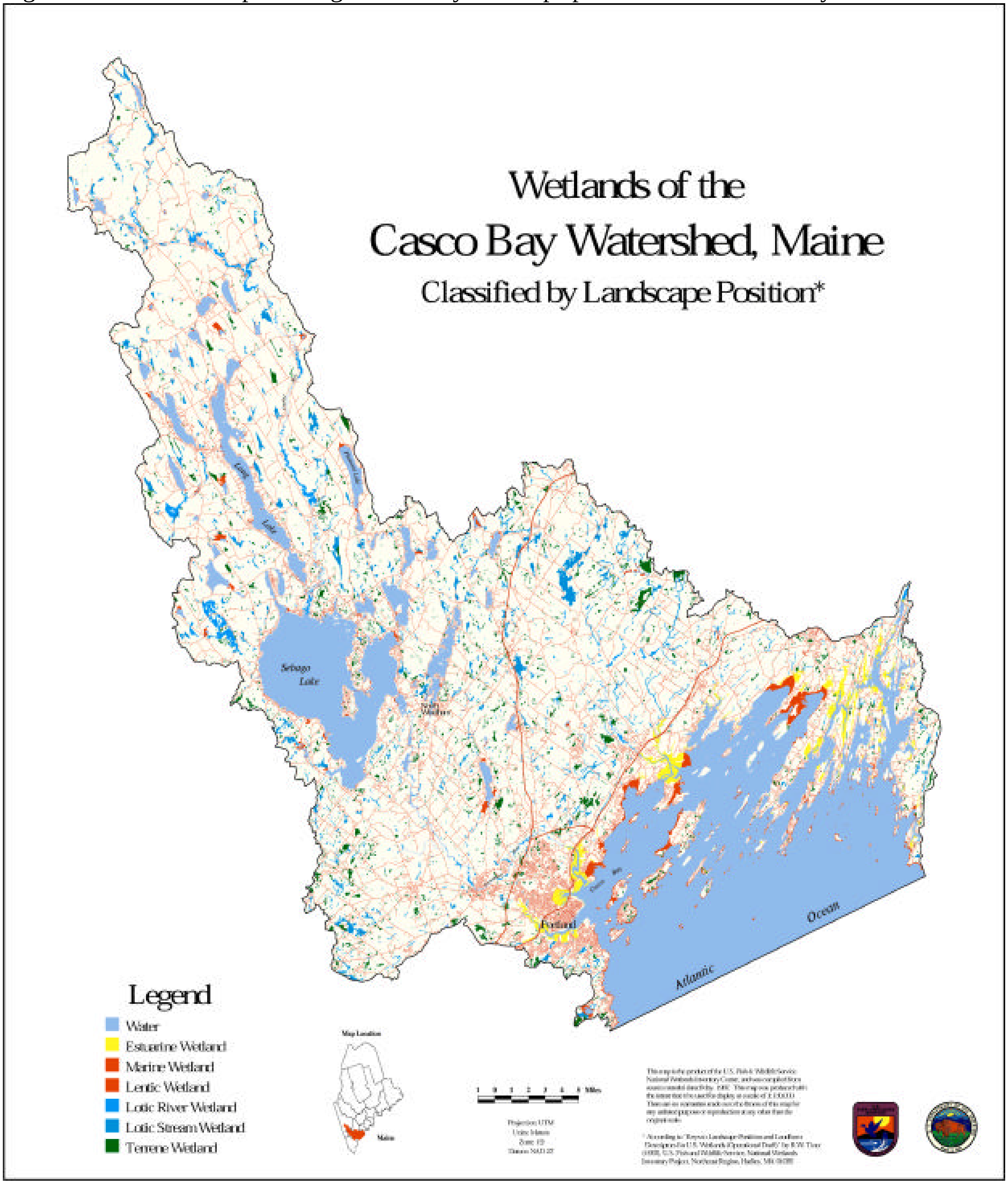


Figure 3. Watershed map showing wetlands by landform type for the Casco Bay watershed.

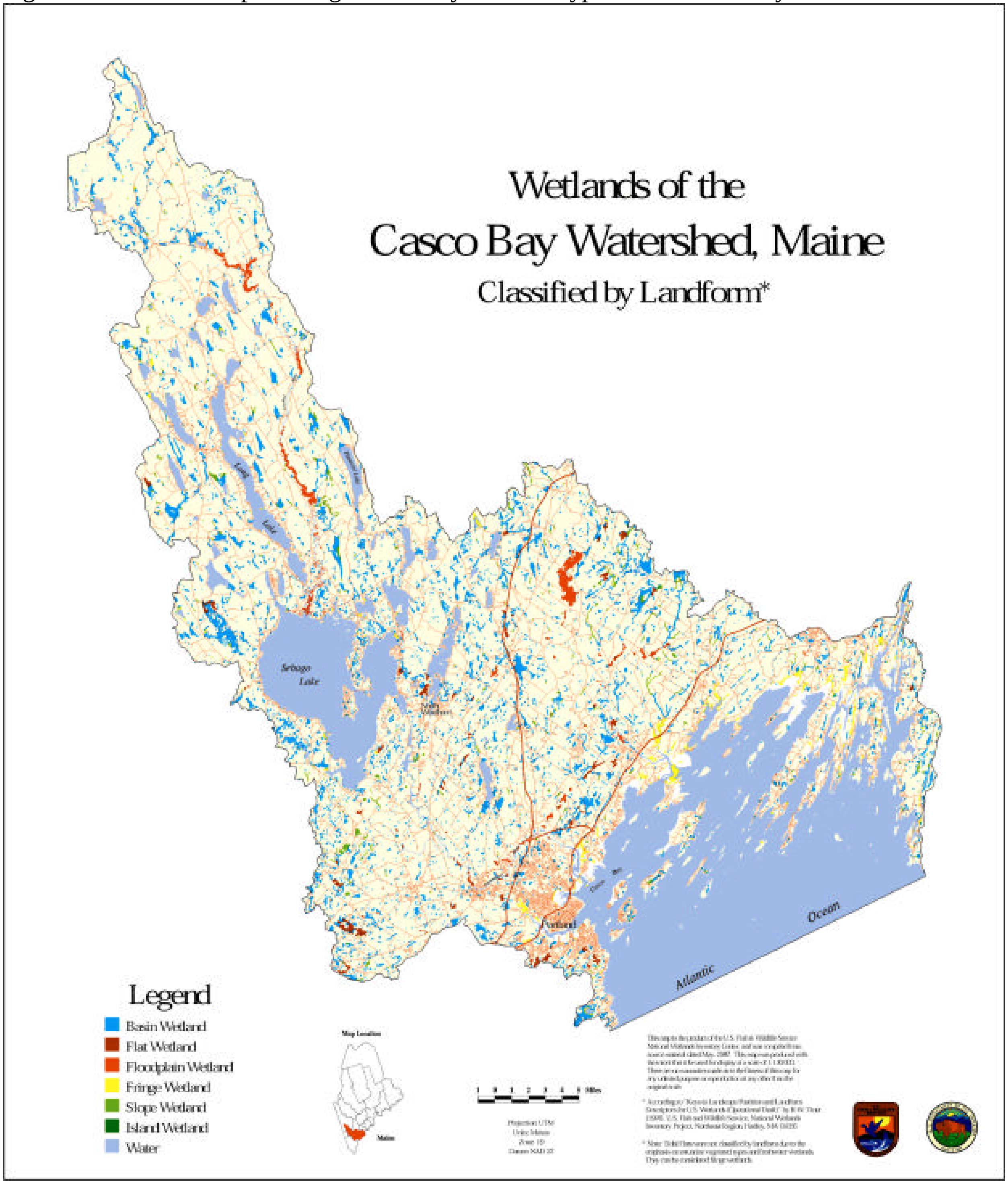
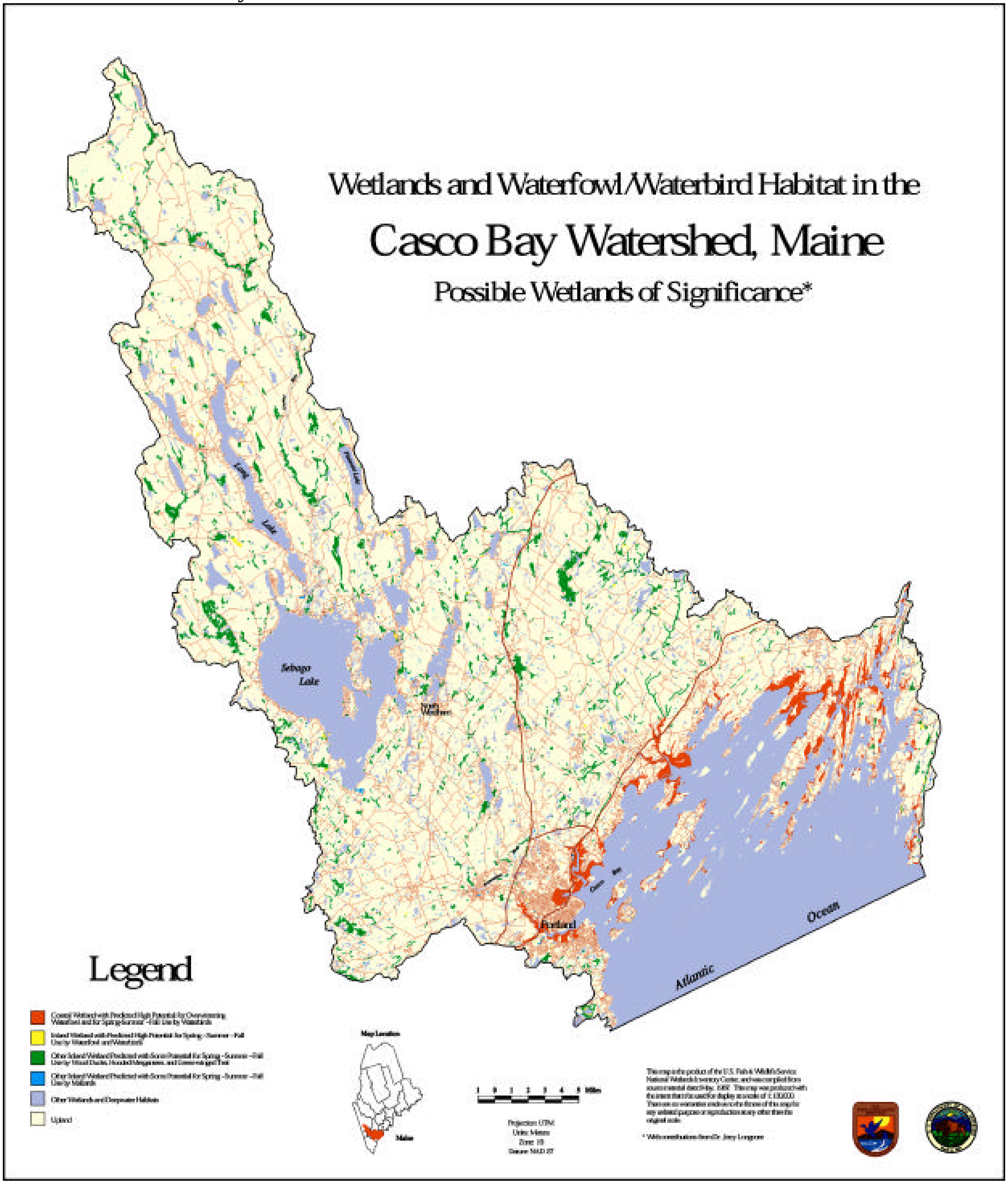


Figure 4. Watershed map showing wetlands of potential significance for waterfowl and waterbird habitat for the Casco Bay watershed.



A Landscape Level Approach to Wetland Functional Assessment for the New York City Water Supply Watersheds

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Abstract

The New York City Department of Environmental Protection (DEP) is developing a wetland functional assessment program for the watersheds that provide unfiltered drinking water to nearly half the population of New York State. The program was implemented in 1998 as a pilot study in two reservoir sub-basins of the New York City water supply watershed. The U. S. Fish and Wildlife Service (USFWS) applied hydrogeomorphic-type modifiers to each National Wetlands Inventory (NWI) wetland in its digital wetland database to provide information for a landscape-level preliminary assessment of eight wetland functions for the study area. In 1999, DEP initiated a reference wetland monitoring program in the pilot sub-basins in order to field-check the USFWS wetland hydrogeomorphic classifications and functional assessments. Studies are also being conducted at the reference wetlands to determine whether the USFWS hydrogeomorphic classification system can provide a framework for a variety of DEP regulatory and non-regulatory wetland programs.

Introduction

The New York City water supply watershed consists of 19 reservoirs and 3 controlled lakes in a 2,000 square-mile watershed located in upstate New York. This watershed provides approximately 1.3 billion gallons of unfiltered drinking water daily to nearly eight million residents of New York City and to an additional one million upstate residents (NYCDEP 1999). Approximately 10 percent of the water supply is derived from the Croton system, located east of the Hudson River in Westchester, Putnam, and Dutchess Counties. The Catskill and Delaware systems, located west of the Hudson River in Delaware, Greene, Scoharie, Sullivan, and Ulster Counties, provide approximately 90 percent of New York City's water needs. The entire watershed contains 27,968 acres of wetlands (Tiner et al. 1996).

In 1989, the U.S. Environmental Protection Agency (EPA) granted a Filtration Avoidance Determination for the Catskill and Delaware portions of the New York City Water Supply Watershed. The DEP has developed a comprehensive long-term watershed protection program that provides the basis for a series of waivers from the filtration requirements of the Surface Water Treatment Rule. This watershed program includes a Wetlands Protection Strategy with the goal to develop and implement a program that will preserve the critical water quality protection functions provided by natural wetland systems located within the New York City watersheds. A key non-regulatory component of the Wetlands Protection Strategy is the development of a Wetland Functional Assessment Program.

In 1997, EPA awarded a Wetland Program Development Grant to the DEP to partially fund a pilot functional assessment program in portions of the Croton watershed that receive water from the Delaware system via an aqueduct. The pilot program combines a GIS approach with a reference wetland monitoring program. For the GIS component, the DEP contracted the USFWS to implement their recently developed Wetland Characterization and Watershed-based Preliminary Assessment of Wetland Functions (W-PAWF) in the West Branch and Boyd Corners sub-basins of the Croton watershed. For the W-PAWF, the USFWS attached hydrogeomorphic-type modifiers to the digital database for each NWI wetland in order to permit a watershed-scale preliminary assessment of wetland functions (Tiner et al. 1999).

In 1999, DEP initiated vegetation, soils, water table, and water quality monitoring at reference wetlands located throughout the pilot study area. Besides determining baseline characteristics of watershed wetlands, the goals of the reference wetland monitoring program included verifying the USFWS NWI maps, hydrogeomorphic modifiers, and preliminary functional assessments. Studies are also currently being conducted at reference wetlands to determine whether community and biogeochemical characteristics vary signficantly with hydrogeomorphic classifications. Such relationships would enable DEP to use the USFWS hydrogeomorphic classification system as a framework for additional wetland programs. Potential future applications of the hydrogeomorphic-enhanced NWI database include the identification of wetlands that meet criteria for increased protection through state and local regulations and the prioritization of wetlands for acquisition and other non-regulatory programs. The enhanced NWI database will also have important applications in DEP modeling programs if loads of various water quality constituents are found to vary significantly with hydrogeomorphic classification.

This paper presents an overview of the preliminary assessment of wetland functions for the two pilot sub-basins and the reference wetland monitoring program.

Study Area

The West Branch and Boyd Corners Reservoir sub-basins are located in the headwaters of the Croton watershed in Putnam and Dutchess Counties, NY (Figure 1). The Boyd Corners and West Branch sub-basin occupy 14,317 acres (22.4 square miles) and 12,736 acres (19.9 square miles), respectively. Combined, this 27,053 acre (42.3 square mile) study area includes approximately 1,858 acres of palustrine wetlands and 1,870 acres of deepwater habitat. Thus, palustrine wetlands and deepwater habitats each occupy approximately 7 percent of the study area. All of the deepwater habitats are human-impounded lakes and reservoirs. According to the NWI, 72 percent of the palustrine wetland acreage is forested, followed by unconsolidated bottom (14.3%), emergent (9.4%), scrub-shrub (3.0%), and aquatic bed (1.2%) cover types. (Tiner et al. 1999).

Typical forested wetland communities include red maple swamps and hemlock-hardwood swamps on sapric histosols (saprists). Acer rubrum (red maple) swamps often include *Betula alleghaniensis*, *Fraxinus pennsylvanica*, and *Ulmus rubra*. *Tsuga canadensis* (hemlock) swamps are codominated by *Acer rubrum*, and also include *Betula alleghaniensis* and *Nyssa sylvatica*. Typical shrub communities include shrub swamps on mineral soils and medium fens and blueberry bog thickets on sapric to hemic histosols. Common shrub species include *Ilex verticillata*, *Rhododendron viscosum*, *Vaccinium corymbosum*, *Viburnum dentatum*, *Alnus serrulata*, *Cornus amomum*, and *Clethra alnifolia*. Typical herbaceous dominants in forested and scrub-shrub wetlands include *Carex stricta*, *Osmunda cinnamomea*, *Symplocarpus foetidus*, *Onoclea sensibilis*, *Thelypteris palustris*, along with *Boehmeria cylindrica*, and *Impatiens* and *Polygonum* species. Most emergent wetland types are associated with human and beaver impoundments and are commonly dominated by *Lythrum salicaria*, *Scirpus cyperinus*, *Sparganium androcladum* and *Typha* species (Reshke 1990; Tiner 1996).

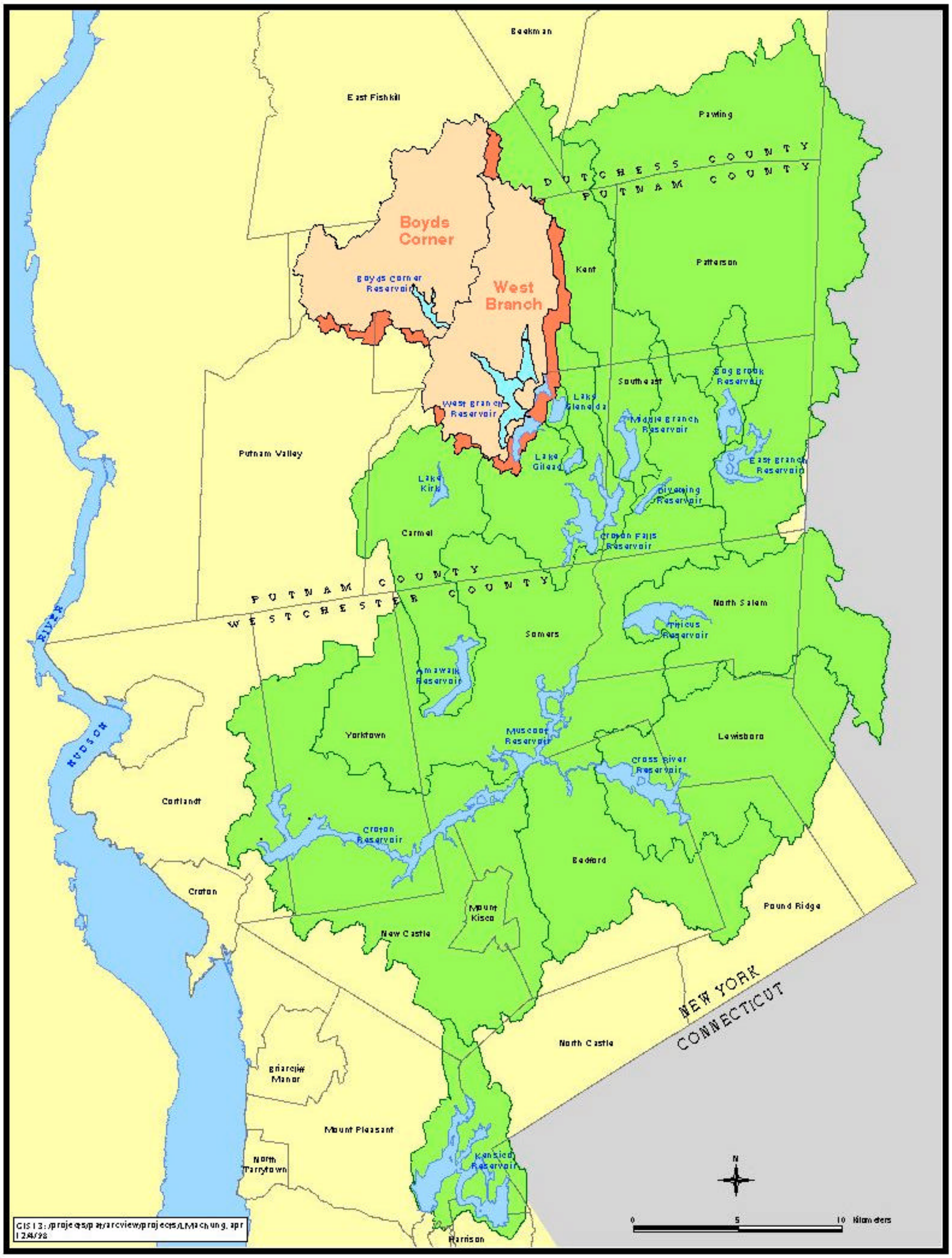

Figure 1. The location of the study areas (Boyd Corners and West Branch sub-basins) in the Croton watershed.

Methods

Watershed-based Preliminary Assessment of Wetland Functions (W-PAWF)

In 1998, the USFWS applied hydrogeomorphic-type modifiers to the digital database for each NWI wetland in the study area. Modifiers were assigned according to methods set forth in "Keys to Landscape Position and Landform Descriptors for U.S. Wetlands" (Tiner 1997). The modifiers, depicting wetland landscape position, landform, and water flow path, were interpreted from NWI and topographic maps and aerial photography (Table 1). Additional modifiers were added, when appropriate, to further describe wetland characteristics. These included headwater, drainage-divide, fragmented, and human impacted.

The USFWS prepared a set of draft wetland classification maps for DEP review. DEP field-checked nearly all accessible wetlands, and reviewed topographic, soil survey, NWI maps, and digital data. After receipt of DEP comments, USFWS revised the draft classifications added them to the NWI digital database using ARC/INFO software (Environmental Systems Research Institute, Redlands, CA).

Next, the USFWS analyzed the enhanced digital database to identify wetlands with significant potential to perform eight functions: surface water detention, streamflow maintenance, retention of sediments and other inorganic particulates, nutrient cycling, shoreline stabilization, provision of fish habitat, provision of waterbird habitat, and provision of other wildlife habitat. Wetlands with predicted high potential to perform these functions were identified through a series of protocols developed by the USFWS with technical support from regional wetland specialists. The protocols are based on the scientific literature and relate wetland features, identified by the NWI and hydrogeomorphic-type classifications, to specific functions. Since this paper focuses on wetland functions related to water quality, assessment protocols for these functions are shown in Table 2. It should be noted that USFWS established additional criteria to identify wetlands that may be performing these functions at significant, but not the highest, levels: these wetlands were designated to be of "other" significant potential. Finally, USFWS produced a series of thematic maps, designating significant wetlands for each function, and generated summary statistics using ARC/INFO (Tiner et al. 1999).

Reference Wetland Monitoring

DEP selected five reference wetlands in 1998 (Table 3). Reference sites were located throughout the West Branch and Boyd Corners sub-basins on State-, County-, or DEP-owned lands. Selection criteria included wetlands contiguous with watercourses (Terrene outflow and Lotic types), representative of common wetland vegetation classes in the study area (forested and scrub-shrub), minimally disturbed in their catchment areas, and accessible to routine sampling. Boundaries of 4 of the 5 sites were delineated using Global Positioning System (GPS). The boundary of the fifth site (Putnam County Park) was not entirely accessible at the time of field work.

Vegetation sampling was conducted at all sites during the 1999 growing season. To achieve a 10 percent cruise, 10m^2 plots were located at 56m intervals along a randomized grid throughout each wetland. At each plot, percent cover was recorded for all herbaceous species present in a 1m^2 area, number of stems for each shrub species was recorded for a 5m^2 area, and the species and diameter at breast height for all trees within the 10 m^2 plot. Importance values were calculated as the sum of Relative Frequency, Relative Density, and Relative Dominance for tree species, of Relative Frequency and Relative Density for shrub species, and of Relative Frequency and Relative Dominance for herbaceous species. Importance values were then compared between lotic and terrene wetlands using non-parametric one way analysis of variance with SAS software (SAS Institute Inc., Cary, NC).

Water table monitoring wells, constructed of 5 cm diameter PVC screen and a vented point, were installed in October 1999 at a subset of the sampling plots in each wetland. Wells were installed in each community type present in a single wetland. A total of 12 well plots were established throughout the 5 study sites. Water table monitoring was initiated in the 2000 growing season and is ongoing, during the non-frozen portions of the year. Soil cores from the well auger holes were collected and described according to standard methods and analyzed for organic matter content by loss on ignition (Soil Survey Staff 1993). In addition, one 20 cm^2 plexiglass sediment disk was installed in October 1999 at each well plot, flush with the soil surface. In October 2000, accumulated sediments were collected, oven-dried and weighed.

In April 2000, gages were installed the outflow streams of the terrene reference wetlands, and at the outflow and inflow streams of lotic reference wetlands and a routine water quality sampling program was initiated. Routine water quality monitoring will be conducted for a minimum of one year. Analytes include color, total phosphorus, total dissolved phosphorus, total suspended solids, total organic carbon, and dissolved organic carbon (Eaton et al. 1995). In fall 2000, automated stage loggers were installed in all inflow and outflow streams. Automated discharge loggers were installed at the lotic stream sites (Unidata America, Lake Oswego, OR).

Table 1. Hydrogeomorphic-type descriptors relevant to the study area (Tiner et al. 1999).

Landscape Position - defines relationship between wetland and adjacent waterbody, if present
Lotic - along rivers and streams Lentic - along lakes and reservoirs Terrene – isolated or headwater outflow or throughflow wetlands with no channelized flow
Landform - shape or physical form of the wetland
Basin - a depressional landform Slope - a landform extending uphill Flat - a relatively level landform Fringe - a landform occuring along or within a flowing or standing waterbody
Water Flow Path - describes direction of water flow in the wetland
Inflow - water enters via an upslope wetland or waterbody, no surface water outlets exist Outflow - lack a wetland or waterbody above them, discharge to wetland or waterbody below Throughflow - have a wetland or waterbody above and below them, so water passes through Isolated – closed depression, lack channelized surface water inflow and outflow

Table 2. Protocols used to identify wetlands with predicted high potential for water quality-related functions (Tiner et al. 1999). C = seasonally flooded water regime.

Function	**Hydrogeomorphic and/or NWI Classes of predicted high potential**	**Rationale**
Surface water detention	Lotic basin and flat Terrene, basin, throughflow	Identifies wetlands contiguous with surface water
Streamflow maintenance	Terrene, outflow Lotic, basin and slope, headwater Lentic basin and fringe	Identifies headwater and impounded wetlands that discharge surface water at times of low flow
Nutrient cycling	Lotic, with C water regime or wetter Lentic, with C water regime or wetter Terrene, basin and slope, throughflow	Landscape positions indicate wetlands contiguous with surface water. C water regime (seasonally flooded) indicates wetter moisture regimes remove nutrients from flood waters and are conducive to organic matter buildup and associated microbial transformations.
Retention of particulates	Lotic, fringe and basin Terrene, basin, throughflow Lentic, basin, throughflow	Landscape positions and landforms identify wetlands that intercept and lower the energy of floodwater, allowing particulates to settle out.
Shoreline stabilization	Lotic, vegetated subclasses Lentic, vegetated subclasses	Vegetation along waterbodies help to stabilize the shoreline.

Table 3. Classifications and community characteristics of the reference wetlands.

Site	NWI Classification	Hydrogeomorphic Classification	Ecological Communities**
Fahnestock	PFO4/PFO1E, PFO1E, PFO4E	Terrene, basin, throughflow, headwater	Highbush blueberry bog, red maple swamp, hemlock swamp
Big Buck	PSS1C	Terrene, basin, outflow, headwater	Medium fen, red maple swamp
Yale Corners	PFO1C	Terrene, basin, outflow*, headwater	Red maple swamp
Ninham	PFO1E, PSS1/EM1E	Lotic stream, basin, throughflow	Shrub swamp, Emergent marsh
Putnam County Park	PFO1/SS1E	Lotic stream, basin, throughflow, headwater*	Shrub swamp, Emergent marsh

*These classifications were included in the digital database, and later determined to be incorrect. Yale Corners is a throughflow wetland, and Putnam County Park was erroneously designated as headwater.
**Based on Reschke (1990)

Results and Discussion

Wetland Characterization

Landscape position, landform, and water flow path classifications for the NWI wetlands in the study area are shown in Table 4. Approximately 47 percent of watershed NWI wetlands were mapped in lotic landscape positions, 45 percent in terrene positions, and 8 percent in lentic positions. Basins were the most abundant landform, accounting for 97.5 percent (1812 acres) of the wetland acreage in the study area (1858 acres), followed by slope (1.7%, 32.1 acres), flat (0.5%, 9.1 acres), and fringe (0.3%, 5.1 acres) landforms. Approximately 68 percent (1270.4 acres) of the wetlands were classified as throughflow, 26 percent as outflow (485.9 acres), 5 percent (89.8 acres) as isolated, and 1 percent (12.3 acres) as inflow. All lotic wetlands (by definition) and lentic wetlands were classified as throughflow. For terrene wetlands, 58 percent (485.9 acres) were classified as outflow, 30 percent (250.1 acres) as throughflow, 11 percent (89.8 acres) as isolated, and 1 percent (12.3 acres) as inflow (Tiner et al. 1999).

Table 4. Hydrogeomorphic characterization for the West Branch and Boyd Corner Reservoir sub-basins combined. All lotic and lentic wetlands are designated as throughflow (Tiner et al. 1999).

Landscape Position	Landform	Water Flow Path	Acreage
Terrene	Slope	Outflow	26.0
	Basin	Isolated	88.8
		Inflow	12.3
		Outflow	459.9
		Throughflow	250.1
	Flat	Isolated	1.0
	Terrene Subtotal		838.1
Lentic	Basin		141.0
	Fringe		5.1
	Lentic Subtotal		146.1
Lotic Stream	Basin		860.0
	Flat		8.1
	Slope		6.1
	Lotic Subtotal		874.2
Grand Total			1858.4

The abundance of terrene outflow is consistent with the location of the West Branch and Boyd Corners reservoir sub-basins in the upper reaches of the Croton watershed. Nearly all of the lotic wetlands (98%) occupy basin landforms, which also reflects the headwater location of the study area. No lotic wetlands were designated as floodplain landforms, indicating that alluvial processes are not dominant along these small, headwater streams.

Based on DEP's review of the draft maps, 11 percent may be an overestimate of the coverage of isolated terrene wetlands. Through field-checks and review of additional map and digital data sources, it was determined that nearly 80 of the total 479 wetlands in the study area had been incorrectly mapped as isolated. These wetlands were connected to other wetlands and surface waters via small or intermittent streams. These small tributaries were not depicted on USGS topographic maps, but were often shown on soil survey maps that had been digitized into a DEP digital stream coverage.

Despite the significant acreage of lakes and reservoirs, lentic wetlands only occupy 0.5 percent of the study area, indicating that the human impounded lakes and reservoirs in the study area create minimal shoreline for wetland establishment. In fact, only 3 percent of the lentic wetlands were mapped in fringe positions along shorelines, the remainder was classified as lentic basins. Lentic basins were determined through field-checking to be naturally occurring wetlands in depressions along an impounded stream. They are located just upstream of, and border, the lacustrine habitat. Given this landscape position, these lentic wetlands were designated with a throughflow water flow path modifier as opposed to a bidirectional modifier typical of lentic fringe systems to indicate a lakeside wetland with a stream running through it.

Watershed-based Preliminary Assessment of Wetland Functions

While eight functions were assessed, results presented in this paper are limited to those related to water quality (Table 2). Based on the assessment protocols, 70 percent (1,294 acres) of the wetlands in the study area was predicted have high potential for streamflow maintenance, 68 percent (1,264 acres) for surface water detention, 67 percent (1,242 acres) for sediment retention, 58 percent (1,075 acres) for nutrient cycling, 40 percent (741 acres) for stream shoreline stablization, and 8 percent (146 acres) for lake shoreline stabilization (Tiner et al. 1999).

The abundance of wetlands with predicted high potential for streamflow maintenance is expected, given the location of the study area in the upper reaches of the Croton watershed. A high percentage of the study area wetlands were also predicted to have high potential for both surface water detention and sediment retention. This is due to the predominance of basin wetlands in the study area. These depressional areas store water, which, in turn, facilitates particle deposition. Thus, there is close agreement between wetlands predicted for the two functions. As previously discussed, there is a lack of fringing wetlands in lentic landscape positions due to the nature of the impoundments in the study area. Hence, there is little wetland area available to stabilize lake shorelines.

An additional 14 percent (264 acres) of the study area's wetlands were predicted to be of "other" significant potential for streamflow maintenance. These included large lotic complexes with little upstream acreage. For surface water detention, an additional 30 percent (557 acres) of the study area's wetlands were predicted to be of "other" significant potential. These included terrene basins with outflow or inflow that may be locally important detention basins. An additional 27 percent (494 acres) of the wetlands, including lotic stream flat, lentic fringe, terrene outflow basin, and larger terrene inflow and isolated basin wetlands were predicted to be of "other significant potential" for sediment retention. For nutrient cycling, an additional 21 percent (391 acres) were predicted to be of possible local importance. These include terrene outflow basin and slope wetlands (Tiner et al. 1999).

Reference Wetland Monitoring

<u>NWI and Hydrogeomorphic Classifications</u>

The first objective of the reference wetlands monitoring program was to ground-truth the draft NWI maps and initial hydrogeomorphic classifications since they provide the basis for the W-PAWF. The W-PAWF predicts the total area of wetlands potentially significant for individual functions and it is therefore important to gauge the accuracy of the sizes of wetlands mapped in the NWI. According to delineations conducted using GPS, the four reference wetlands were roughly 15 percent to 64 percent larger than shown in the NWI (Table 5). More work is needed to gain an accurate estimate of the wetland size error rate, and to determine whether it varies significantly with wetland type. Limitations of aerial photointerpretation for wetland mapping are widely recognized, particularly for evergreen forested wetlands and drier-end wetlands (Tiner 1999). A constant error rate would be more desirable, as size-related errors specific to wetland types could bias the W-PAWF results.

Table 5. Acreage comparisons and predicted levels of water quality-related functions for the reference wetlands in the study area. Functional predictions are from the W-PAWF. Sites designated as "High" were predicted to perform the function at a level significant to the watershed, while sites designated as "Other" were predicted to be of some, but not the highest, potential for individual functions. Sites designated as "Not Listed" were not predicted to have significant potential on a watershed scale for performing an individual function (Tiner et al. 1999).

Site	NWI acres	GPS acres	Sediment Retention	Water Detention	Streamflow Maintenance	Nutrient Cycling	Shoreline Stabilization
Fahnestock	14.3	16.8	High	High	High	High	Not Listed
Big Buck	3.9	6.7	Other	Other	High	Other	Not Listed
Yale Corners	2.4	6.6	Other	Other	High	Other	Not Listed
Ninham	7.9	11.4	High	High	Not Listed	High	High
Putnam County Park	34.7	*	High	High	High	High	High

*The boundaries of this wetland were not accessible for GPS delineation

There was good agreement between NWI vegetation classes and communities observed during site visits and quantitative vegetation sampling (Table 3). Putnam County Park is one exception. This predominantly emergent wetland is mapped as scrub-shrub. This inconsistency is likely due to successional changes caused by impoundment of this site by a beaver dam.

Landscape position and landform classifications were correct for all of the reference wetlands. Water flow path modifiers were more problematic for the terrene systems. Two of the three terrene reference wetlands (Fahnestock and Big Buck) were originally mapped as isolated and later determined to be outflow, headwater wetlands. These changes were incorporated into the final maps. The remaining terrene reference site (Yale Corners) was mapped as terrene outflow, but is a terrene throughflow system. Unfortunately, this determination was not made until after the Wetland Characterization and W-PAWF was completed for the study area and this incorrect classification was used for the W-PAWF. Criteria to accurately establish water regime modifiers for terrene wetlands and the cutoff between terrene throughflow and lotic wetlands in a consistent, repeatable fashion are needed for future projects. Additional field work prior to applying W-PAWF is recommended. The Putnam County Park reference wetland was also flagged with a headwater modifier, even though it is along a higher order stream, and all wetlands upstream of it are not classified as headwater. This incorrect classification was also incorporated into the W-PAWF.

The W-PAWF uses NWI water regime modifiers as part of the criteria to select wetlands important for the nutrient cycling function (Table 2). All of the reference sites had C (seasonally flooded) or E (seasonally flooded/saturated) water regimes and were predicted to have high potential for the nutrient cycling function (Tables 3 and 5). Water table data collected over the 2000 and future growing seasons will be to analyzed to determine the amount of time that the 5 sites are flooded or saturated in the growing season, and whether this agrees with the NWI modifiers.

Preliminary Functional Assessments

Indicators of specific functions were included as parameters in the reference wetland program. Therefore, in addition to verifying various NWI and hydrogeomorphic modifiers, DEP's monitoring program will provide a means to assess the functional predictions of the W-PAWF.

Soil organic matter is typically measured as an indicator of nutrient cycling since its oxidation by microbial populations supports nutrient and elemental transformations. Soil organic matter contents of reference sites with wetter water regimes and predicted high nutrient cycling potential will be compared to sites with drier-end water regimes that lack high potential designations. However, the reference wetland population will have to expanded to achieve this, since all current reference were of predicted high potential (Table 5). The routine water quality monitoring program will also enable DEP to assess the nutrient cycling function, specifically for phosphorus. Comparisons of inflow and outflow loads of total and total dissolved phosphorus will indicate the effects of different wetland types on stream-water phosphorus concentrations.

Disks were installed to measure the amount and composition of sediments accumulated in the reference sites over time. One of the terrene (Fahnestock) and both lotic reference wetlands were predicted to have high potential for sediment retention in the W-PAWF (Table 5). After one year, disks in lotic wetlands had negligible sediment accumulations and were left in place to be collected after another one year period. Of the terrene wetlands, Fahnestock had accumulated an average of .04 +/- .005 g/cm^2, Yale 0.02 +/- .02 g/cm^2, and Big Buck 0.01 +/- .002 g/cm^2. Laboratory results are not yet available for the terrene wetland disks, but field observations revealed that these depositions are primarily organic litter. Future years of sedimentation and water table data will be analyzed to determine the impacts of hydrologic regime on the type and nature of sediments accumulated in various wetland types. Lotic wetlands may accumulate less organic sediment than terrene systems due to less standing water in the growing season or to scouring by flood events. Additional years of data are also required to determine the extent that lotic wetlands retain mineral particulates, and to determine whether terrene throughflow wetlands perform significant sediment retention, as predicted, or accumulate primarily organic litter like their outflow counterparts.

Data from the automated stage and discharge loggers will be analyzed to assess the streamflow maintenance and surface water detention functions of the five reference sites. All three terrene and one of the lotic (Putnam County Park) reference sites were predicted to have high potential for streamflow maintenance. The Putnam County Park wetland was erroneously predicted to be of high potential for streamflow maintenance due to its incorrect classification as a headwater wetland. A comparison of the volumes of water discharged during baseflow per acre of wetland will be compared among the terrrene sites, which may enable DEP to rank these terrene wetlands with regard to this function. The timing and magnitude of flood peaks in relation to storm events will be examined to assess the storm water detention function for the five reference sites. One terrene (Fahnestock) and both lotic reference wetlands were predicted to have high potential for surface water detention. Fahenstock is included in this category because it is a terrene throughflow wetland. Yale Corners should be ranked as high potential as well, but was not, due to its incorrect classification as outflow (Table 5).

Future Applications of the W-PAWF

If significant relationships are found between reference wetland data (water and soil chemistry and vegetation communities) and hydrogeomorphic classes (landscape position, landform, water flow, or other NWI modifiers), then the enhanced NWI data base may benefit additional DEP programs. For example, if exported loads of any of the water quality parameters (color, carbon, suspended solids, and phosphorus) are found to vary with wetland hydrogeomorphic class, then the enhanced database could be applied to DEP's terrestrial modeling program. With this approach, specific values could be assigned by wetland type, as opposed to one constant for all wetlands.

If plant community compositions are found to vary significantly with hydrogeomorphic class, then this classification system could provide the framework for developing a wetland biological assessment program. The digital database could also be used to identify potential locations of specific wetland community types. This could benefit DEPs effort to identify wetlands that meet criteria for increased protection through state and local regulations, and to prioritize wetlands for acquisition and other non-regulatory programs.

Preliminary analysis of vegetation data was conducted to examine relationships between landscape position and species importance values in the five reference wetlands. In the canopy, *Ulmus rubra* had a significantly higher importance value in lotic wetlands (Median IV = 13.61) than terrene wetlands (Median IV = 2.74) ($p = .04$) while *Acer rubrum* was distributed evenly among the lotic (IV = 50.96) and terrrene (IV = 51.41) wetlands ($p = 0.7$). A few species, including *Betula alleghaniensis*, *Nyssa sylvatica*, and *Tsuga canadensis* were measured in terrene plots only. *Betula alleghaniensis* was observed in lotic wetlands, but occurred at a frequency too low to appear in study plots. In the shrub layer, *Vaccinium corymbosum* and *Clethra alnifolia* were measured only in terrene plots, while *Rosa palustris* was measured only in lotic plots. Of the species that occurred in both terrene and lotic wetlands, *Cornus amomum*, *Lindera benzoin*, and *Viburnum lentago* had higher median importance values in lotic wetlands and *Ilex verticillata* and *Rhododendron viscosum* had higher median importance values in terrene wetlands, although these differences were not significant (Table 6). In the herbaceous layer, arums and sedges exhibited similar importance values in lotic and terrene wetlands. The median importance value the Arum family collectively was 7.9 in lotic and 7.3 in terrene wetlands ($p = 0.7$), median importance values for Cyperaceae were 6.2 in lotic and 6.6 in terrene wetlands ($p = 0.5$). This can be explained by the relatively even distribution of *Symplocarpus foetidus* and *Carex stricta* throughout wetlands in the study area. *Osmunda cinnamomea* occurred in all three terrene and one of the lotic wetlands, and had higher importance values in the terrene wetlands (IV = 24.8, 9.7, and 14.0 at Fahnestock, Big Buck, and Yale, respectively) than in the lotic site in which it was measured (IV = 1.41 at Putnam County Park). The importance value of the Urticaceae family was significantly higher in lotic (IV = 8.4) than in terrene (IV = 0.9) wetlands (p = 0.02). Given the small size of the pilot study (n = 3 terrene and n = 2 lotic wetlands), more reference wetlands are needed to determine whether several of these trends are significant. Also, the vegetation data will be analyzed along with well and water quality data to examine relationships between wetland hydrogeomorphic setting, hydroperiod, water chemistry, and species composition.

Table 6. Median importance values of various shrub species at lotic and terrene wetlands.

Species	Lotic	Terrene
Rosa palustris	5.4	*
Cornus amomum	16.9	2.3
Viburnum lentago	10.3	2.3
Ilex verticillata	13.5	32.7
Rhododendron viscosum	6.5	28.5
Vaccinium corymbosum	*	20.5
Clethra alnifolia	*	10.3

Conclusions

The USFWS attached modifiers to the digital database for NWI wetlands in a pilot study area in the Croton System of the New York City Water SupplyWatershed. These modifiers characterize wetlands in terms of their hydrogeomorphic characteristics, which permits a preliminary, watershed-scale assessment of wetland functions. DEP has coupled this GIS-based project with a reference wetland monitoring program in order to verify the wetland hydrogeomorphic and NWI classifcations, to test some of the preliminary functional assessments, and to expand the applications of the GIS-based methodolgy.

Landscape position, landform, and NWI vegetation class modifiers were, in general, accurately ascribed to wetlands by the USFWS. The water flow path modifier was more problematic, particularly for terrene wetlands with intermittent outflows. This problem will be addressed in future projects by overlaying streams digitized from the soil survey. Well data will be analyzed to assess NWI water regime modifiers, as these modifiers are included in the criteria for a number of functional assessments.

A number of parameters are currently being monitored at reference wetlands in order to test some of the preliminary functional assessments, and to rank wetlands within the high predicted potential categories. Measuring the amount of sediments accumulated over time in the reference sites will enable DEP to assess the prediction that one of the terrene and both lotic reference sites are of high potential for sediment retention. Soil organic matter is an indicator of nutrient cycling and will be compared among the reference sites, all of which are predicted to be of high potential for this function. Routine phosphorus monitoring at the reference wetland streams will provide an indicator of effects of different wetland types on the concentration of this nutrient in stream water. Automated stage and discharge data is being collected to determine the surface water detention and baseflow maintenance of the different wetland types.

A future goal is increase the number of reference wetlands in order to examine relationships among wetland vegetation and biogeochemical characteristics and their hydrogeomorphic classifications. This combination of field and digital data should provide DEP with a management tool with applications in water quality modeling projects and in the development of regulatory and non-regulatory wetland protection programs.

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A GIS-Based Model for Evaluating Wetland Significance

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Abstract

The National Oceanic and Atmospheric Administration's (NOAA) Coastal Services Center has been developing a GIS-based model to help wetland managers examine wetland habitats within a watershed. This model, called the "Spatial Wetland Assessment for Management & Planning" or "SWAMP," currently consists of two modules, tidal and riverine, that evaluate a wetland's contribution to water quality, hydrology, and habitat. The model considers site-specific characteristics obtained from soil and vegetative data, as well as landscape characteristics obtained from GIS analyses. SWAMP uses ArcView® Spatial Analyst® with an interface that walks the user through alternatives for prioritizing wetland habitat. SWAMP was developed originally for the Ashepoo-Combahee-Edisto River Basin of South Carolina but should be transferable to basins in other regions in the near future. Current efforts are underway to develop SWAMP to address wetland restoration issues.

Introduction

Wetlands along the coastal plain are of great ecological importance because they occupy much of the landscape, are significant components of virtually all coastal ecosystems, and are factors influencing water quality, estuarine productivity, wildlife habitat, and the overall character of the coastal area. Despite significant reduction in wetland losses to agricultural conversion (the main cause of wetland loss in the past), wetlands continue to be drained or filled for development (Hefner et al. 1994). The loss of vast acreages of wetlands causes concern because the services these ecosystems provide are lost with them. Conflicts between economic development and wetlands protection continue to be a major issue, with many coastal communities considering wetlands protection to be a major barrier to desired economic development. Since wetlands are a dominant part of the coastal landscape and are vitally important to many aspects of an area's ecology (National Research Council 1995), their management and protection are main goals of coastal managers charged with protecting the integrity of a wetland ecosystem while managing it for human use.

Borrowing from a model developed by the North Carolina Division of Coastal Management—the North Carolina Coastal Region Evaluation of Wetland Significance (NC-CREWS) (Sutter et al. 1999), the NOAA Coastal Services Center has developed a modeling approach to guide managers in decision making. This approach - the Spatial Wetland Assessment for Management and Planning (SWAMP) model - uses basic ecological principles to evaluate the significance of wetlands within a watershed, while allowing the wetland professional the opportunity to assign the relative importance of each parameter. This paper presents only an introduction to the SWAMP approach.

Highlights of SWAMP

Several wetland features are emphasized in the SWAMP model, including hydrogeomorphology, vegetation, and associated landscape characteristics. The hydrogeomorphic (HGM) classification system for wetlands (Brinson 1993) classifies wetlands into categories based on landscape position (geomorphic setting), water sources, and hydrodynamics (direction of water flow and strength of water movement). HGM classification focuses on the abiotic features of wetlands rather than on the species composition of wetland vegetation which is the basis for more traditional wetland classification schemes. The result is to aggregate wetlands with similar functions into appropriate classes based on geomorphic, physical, and chemical properties of wetlands. The HGM class of a wetland, in itself, indicates much about the ecosystem functions of the wetland. The HGM approach also forces consideration of factors external to the wetland site, such as water source. This helps relate the wetland to the larger landscape and places consideration of the wetland's functions into a landscape and watershed context. To date, tidal estuarine and freshwater riverine modules have been developed for SWAMP, and each module is a distinct HGM class.

In addition to HGM classes, wetland types identified by dominant vegetation and landscape position are used at several points in the assessment. This reflects a recognition that the biological properties of a wetland considered together with its hydrogeomorphic properties can provide a more detailed indication of wetland function than either property taken alone. Wetland types are used in SWAMP as indicators of functional characteristics. The correlation between wetland type and wetland functions was determined from fieldwork and best professional judgment.

Unlike assessment procedures that depend solely on information collected within a wetland, this procedure relies heavily on factors external to the wetland site itself. Relationships between a wetland and the landscape within which it exists are integral considerations in determining wetland functional significance (National Research Council 1995; Vivian-Smith 2001). Characteristics of the landscape surrounding a wetland are often more important determinants of its functional significance than the characteristics of the wetland itself. While the emphasis on a wetland's landscape context is a more ecologically-sound approach to functional assessment than site-specific methods, this emphasis requires a great deal more information than could be collected within the wetland itself. The procedure is conducted in a geographic information system (GIS) using geospatial data and analyses because this technology provides the most practical way to analyze the spatial relationships of landscape elements and their properties.

This geospatial data structure also makes it possible to consider specific wetland functions individually. For example, in a watershed targeted for nonpoint source pollution reduction, a management objective may be to provide the highest level of protection to wetlands most important for maintaining water quality. GIS makes it possible to examine each wetland's significance for a suite of functions, persistently storing model outputs.

Individual ratings in the GIS also can be used to improve planning, impact assessment, and mitigation for development projects that impact wetlands. If alternative sites are available (such as optional corridors for a highway), the alternative with the least impact on the wetland function considered most important in the watershed can be identified. Rather than simply minimizing the number of acres of wetlands impacted, the objective would be to reduce impacts to the most important wetland functions. Environmental assessment of wetland impacts can identify specific functions that would be lost. Mitigation can be improved by giving high priority to sites with the greatest potential for performing the same functions. SWAMP allows managers to evaluate the role that tidal and riverine wetlands play in their watersheds and identifies those wetlands that make the greatest contribution to water quality, hydrology, and habitat.

The SWAMP approach can be transferred to other regions, initiated with a workshop that brings together local experts to describe the system ecology and establish parameters and thresholds. Some GIS programming may be required to modify the features important for particular watersheds, but the interface to evaluate those features is dynamic and should be useful within any geographic area. SWAMP also includes a tool for loading data from other areas and establishing rules for those data.

The SWAMP approach was developed initially for tidal and riverine wetlands within the Ashepoo Basin of the larger Ashepoo-Combahee-Edisto (ACE) Basin, South Carolina (Figure 1). Residential and urban land use in the ACE Basin is steadily increasing. As a result, the area is contending with issues such as habitat loss, resource depletion, nonpoint source pollution, and nutrient loadings to estuaries and coastal waters (South Carolina Department of Natural Resources and NOAA Coastal Services Center 2000). Within the ACE Basin, SWAMP was tested within the Ashepoo Basin. It consists of 10 smaller watersheds (14-digit hydrologic units, as defined by the United Stated Geological Survey). Based on the wetland and land cover data created in a joint effort in 1994, the Ashepoo consists of 13,590 hectares (33,580 acres) of tidal wetlands, 20,791 hectares (51,375 acres) of riverine wetlands, and 1,558 hectares (3,850 acres) of other wetland types. A listing of the land cover in the Ashepoo Basin can be found in Table 1.

Figure 1. Location of the ACE Basin in South Carolina is shown in darker shades. The darkest watershed in the center is the Ashepoo Basin where SWAMP testing occurred.

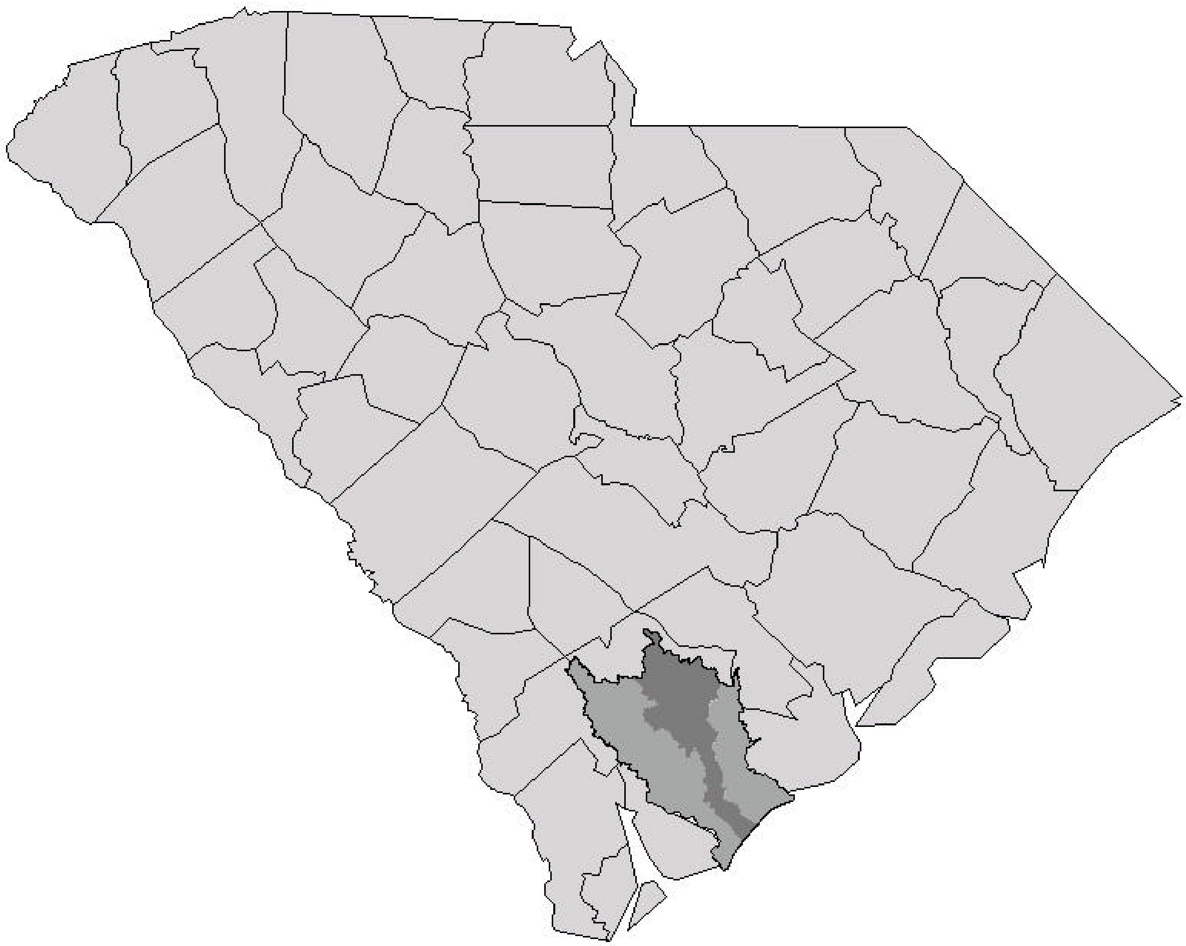


Table 1. Approximate land cover in the Ashepoo Basin, South Carolina, as identified in the 1994 National Wetland Inventory. These numbers are a slight underrepresentation of the actual land cover in the basin because data were not available for the very tip of the watershed. These data (and the associated metadata) are available for download at *www.dnr.state.sc.us/gisdata/*.

Land Cover	Hectares
UPLAND PLANTED PINE	27361.22
FORESTED WETLAND	19609.45
NON-FORESTED WETLAND	16198.63
CROPLAND/PASTURE	11473.16
MIXED UPLAND FOREST	6064.84
OPEN WATER	5514.83
EVERGREEN UPLAND FOREST	4516.47
BAY/ESTUARY	3373.66
RESIDENTIAL	3310.10
TRANSPORTATION/UTILITIES	499.23
COMMERCIAL/SERVICES	479.14
SANDY AREA	345.90
DECIDUOUS UPLAND FOREST	223.83
INDUSTRIAL	149.62
BEACHES	81.07
MINES/QUARRIES/PITS	65.16
OTHER URBAN	39.16
INDUSTRIAL/COMMERCIAL COMPLEX	19.46
MIXED URBAN	18.98
ORCHARD/GROVE/VINEYARD	17.77
OTHER	1.15
HERBACEOUS RANGELAND	0.43
SHRUB/BRUSH RANGELAND	0.21

Parameter Development

In November 1998, researchers and managers knowledgeable about the ACE Basin met to discuss the measurable parameters that might be included in an analytical tool. The experts focused on the water quality and habitat components of tidal estuarine wetlands, keeping in mind geospatial data availability and GIS capabilities. Once the parameters were established, some thresholds were developed. For thresholds not determined during the workshop, the author of the model reviewed the literature and requested feedback from participants and other interested parties. Riverine wetland parameters and thresholds were extracted from the NC-CREWS model and modified as necessary.

Model Structure

The procedure uses the Spatial Analyst® extension of ArcView® and Avenue® programming to perform the geospatial analyses on wetlands. The decision interface employs Microsoft® Visual Basic®. Since the assessment procedure was designed for GIS analyses, the choice and expression of individual parameters have been shaped to a large extent by the available GIS data and the capabilities and limitations of ArcView.

GIS data layers used in the procedure include: 1) land cover, including wetland boundaries and types, 2) detailed soils, 3) hydrography, and 4) watershed boundaries. Data were obtained and converted to raster format as necessary, using ArcView. For sample runs of this model, a 10-meter cell size has been employed. To ensure that cells maintained a consistent origin throughout and among all layers, a common extent was selected and applied to each grid.

SWAMP requires a continuous surface of land cover within a watershed, indicating the type of habitat appearing on the ground. Wetland types were derived from National Wetlands Inventory digital data (classes from Cowardin et al. 1979) and collapsed into fewer groups. These wetland attributes are critical and form the starting point for the assessment. Wetlands also were labeled to HGM classes manually, based on the proximity to tides, salinity, and rivers (including streams and tributaries).

The soils data layer consists of digitized detailed county soils maps produced by the Natural Resources Conservation Service (NRCS, formerly Soil Conservation Service). The soil series associated with a wetland is identified from the soils data layer, and the properties of the series are used to determine soil capacity for facilitating the wetland's capability to perform various functions.

The basic hydrography data layer consists of 1:24,000-scale United States Geological Survey (USGS) digital line graphs (DLGs) converted to ArcInfo® coverages. For the purposes of the draft development in the ACE Basin, 1:100,000 hydrography from the United States Environmental Protection Agency's REACH file was substituted for stream order only. Model users will need to ensure that data with Strahler stream order are provided. The procedure uses stream order as an indicator of watershed position, and stream order classified according to the Strahler system was determined to be the most appropriate classification scheme for this application. Stream order was determined manually and added to the attribute files.

The watersheds used in the procedure are relatively small hydrologic units (14-digit units) delineated by the Natural Resources Conservation Service (NRCS).

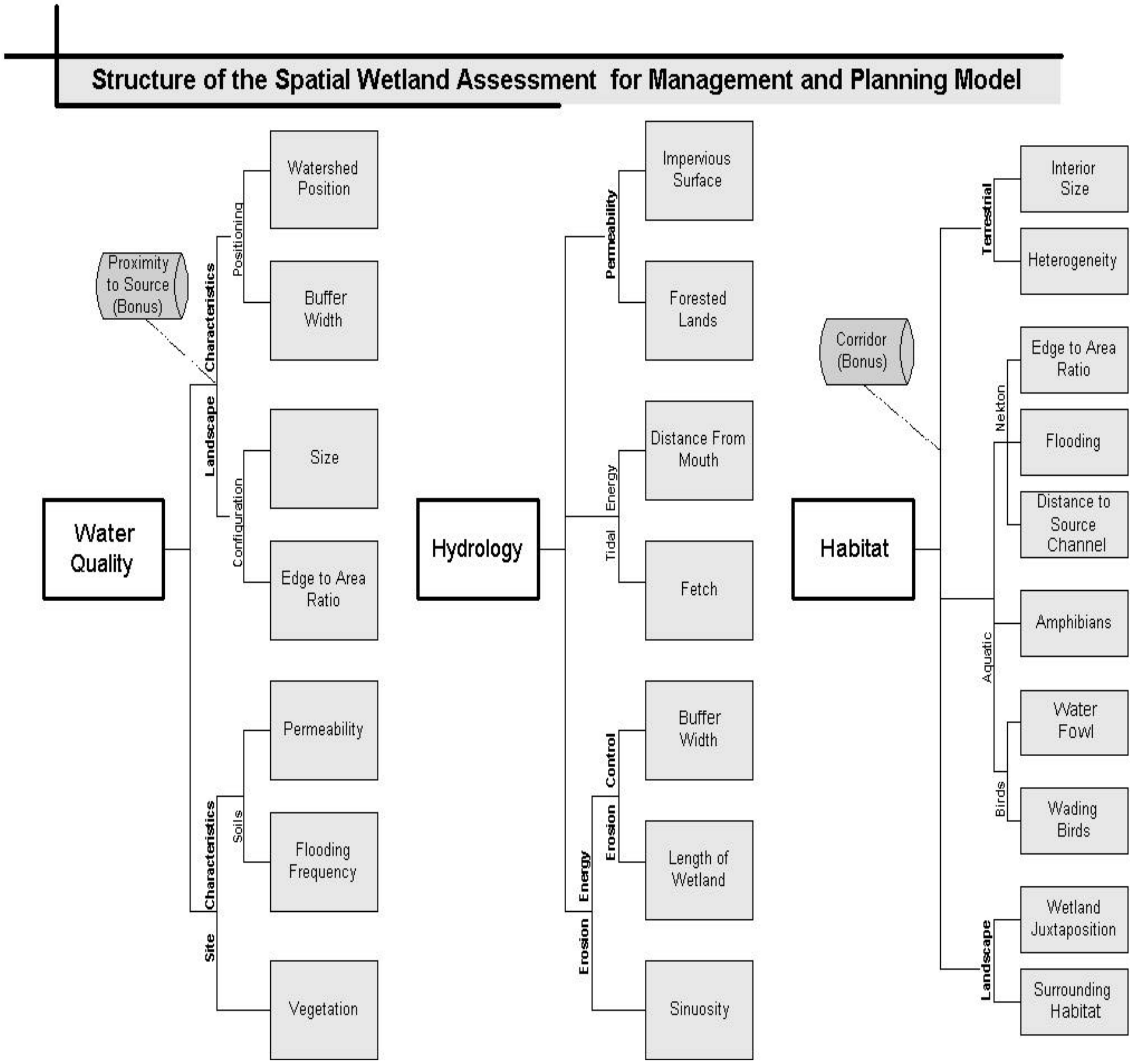
In addition to wetland type and HGM class, several other parameters are used as indicators of the existence or level of specific wetland functions. These include both site-specific parameters, such as wetland size and soil characteristics, derived from the data above, and landscape properties, such as watershed position, water sources, land uses, and landscape patterns. Values for these landscape parameters are determined by GIS analyses based on the data layers previously referenced.

SWAMP uses a hierarchical structure in which individual parameters are rated and successively combined until the significance is determined for each primary function (water quality, hydrology, and habitat) (Figure 2). The functional significance of a wetland is determined by the degree to which it performs, or has the capacity to perform, each function. SWAMP results in a rating of each individual wetland's ecological significance to its watershed, depending on the priorities established by the user.

An evaluation of specific parameters is performed to derive significance ratings for wetland functions. In all cases, parameter values are determined by GIS analyses based on the available data layers. Some parameters, such as wetland type, are surrogates or indicators of other wetland properties that actually determine the wetland's functional capacity. The use of indicator parameters is necessitated by the limitations of GIS data and techniques. Parameter values, in turn, are combined to produce ratings for the various functions.

The broadest grouping of wetland function used in SWAMP includes water quality, hydrology, and habitat. The highest hierarchical level, or end result of applying the procedure, is the wetland's contribution to its watershed's water quality, hydrology, and habitat function. The decision rules are left to the user to determine and allow parameters that vary in importance to be ranked differently, depending on the context of the particular model run. Default rules are provided and recommended, unless specific regional knowledge suggests an alternate evaluation.

Figure 2. SWAMP model structure for tidal estuarine wetlands.



Model Evaluation

Significance for each of the functions that SWAMP evaluates is divided into three broad classes: 1) exceptional significance, 2) substantial significance, and 3) beneficial significance. This approach is used, as opposed to a numerical scoring approach, due to current limitations in our understanding of wetland functions. Attempting to assign a specific value along a numeric continuum of functional significance potentially exaggerates the accuracy upon which current knowledge can realistically be applied (i.e., gives a false sense of accuracy). The three significance classes used in SWAMP provide the information necessary to meet the procedure's objectives without going beyond the realm of reasonable scientific validity. As scientific understanding and data availability increase, it is likely that continuous, numerical scoring will be incorporated into SWAMP.

The basic evaluation of a function is performed at the parameter level. A value is assigned to each parameter as it relates to the performance of the wetland function being considered. For example, if the wetland soils have properties that are highly conducive to the function being considered, the soil characteristics parameter is rated exceptional (E); if soil properties are less conducive to performing the function, the parameter is rated substantial (S) ; and if soil properties are not at all conducive to the function, the parameter is rated beneficial (B). The individual parameter rating is then combined with ratings of other parameters to give an "E", "S", or "B" rating for the specific function.

The evaluation of individual parameters is based on ecological principles describing how wetlands and landscapes function and can be modified somewhat easily for other regions. The process of successively combining ratings up the structural hierarchy is the most complex aspect of the SWAMP procedure. Since the ecological processes themselves interact in complex ways, combining ratings is much more complicated than a simple summation of individual ratings. Users of SWAMP should understand these complexities when designing decision rules. Because coastal wetland managers requested the opportunity to alter the decision rules, an interface was developed to allow users to change the defaults and combine the parameters to meet their needs. The user has the responsibility of creating and documenting the relationships used to determine the ratings.

SWAMP maintains all of the individual parameter ratings and combinations within the ArcView project. Since the combining process can be complex, it may not be obvious why a wetland receives an exceptional, substantial, or beneficial rating for any function. ArcView makes it possible to trace through the parameters and couplings that result in a wetland's final rating.

Capacity versus Opportunity

For a wetland to actually perform a given function, it must have both the opportunity and the capacity for that function. Considering nonpoint source pollution, for example, there must be a source of potentially polluted runoff entering the wetland to provide an opportunity, and the wetland must have the internal capacity to hold the runoff and remove the pollutants before releasing the water. Opportunity to perform a function is usually determined by factors external to the wetland, while capacity to perform the function is determined by properties of the wetland itself, along with its landscape position.

Since SWAMP is a landscape-scale procedure that evaluates the functions a wetland performs in relation to its surroundings, opportunity parameters are included where appropriate. A functional assessment that is too heavily dependent on opportunity parameters, however, is static and will rapidly become invalid as land use changes. A wetland that is bordered by natural forest today can be bordered by a young pine plantation or a subdivision tomorrow. The fact that a wetland does not have the opportunity to perform certain functions today does not mean that it will not have the opportunity in the future. If an assessment of wetland significance is to remain valid over time in a landscape subject to change, opportunity parameters alone cannot be determinative. Lack of present opportunity alone should never result in a lower level of significance for a function. Opportunity is treated essentially as a "bonus" consideration that can result in a higher evaluation for a wetland than its capacity alone would indicate; the user has the choice of whether to include the "bonus" in the evaluation.

Model Process

Analyses are performed in ArcView Spatial Analyst. A raster environment is used for quick processing speed and concurrent layer evaluation. All vector data used in this evaluation must be converted to raster format (ARCGRID® grid) using a consistent cell size and constant grid extent. Each parameter in this model produces an output grid written to a directory specified by the user.

SWAMP is supplied as an ArcView extension. Once loaded into ArcView, the user must establish the data. Then, the user is asked to select a watershed for the next series of evaluations. Using a pull-down menu, the user must select which module (HGM class of wetland) and which component (water quality, hydrology, or habitat) to evaluate. The user may evaluate any combination of functions within the watershed. Once the parameters are established for the area of interest, the user may select one of three methods for establishing the decision rules: 1) linear weighting, 2) coupled parameters, or 3) a full matrix of decisions. Examples of these three options are shown in Figures 3 to 5. Results of the user's decisions are shown graphically in ArcView at each opportunity the user has to make a decision, and the frequency of that decision varies with the method selected.

Figure 3. The simplest option for evaluating parameters within SWAMP is applying weights to each parameter and selecting the cutoff points for the exceptional and beneficial categories.

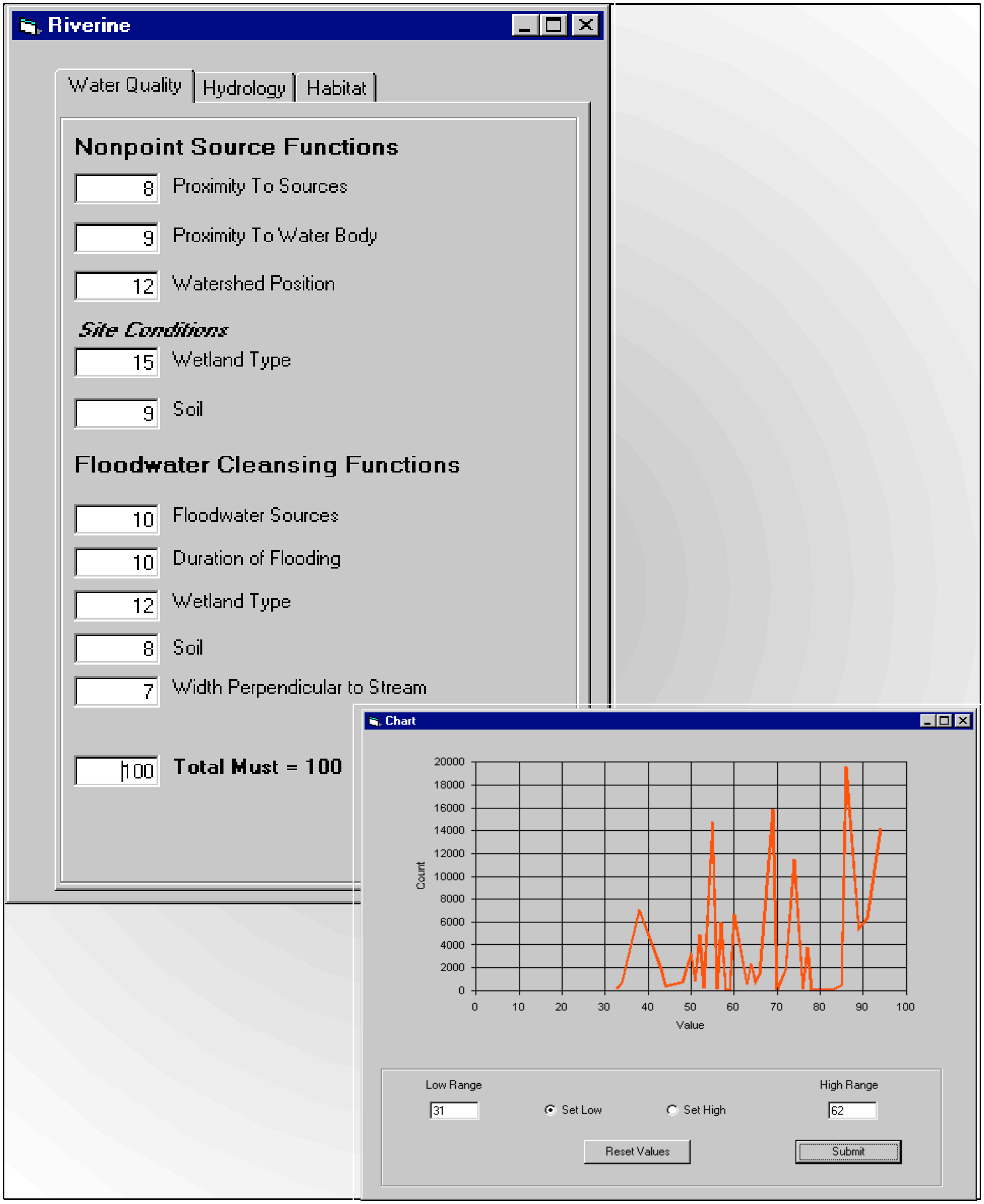
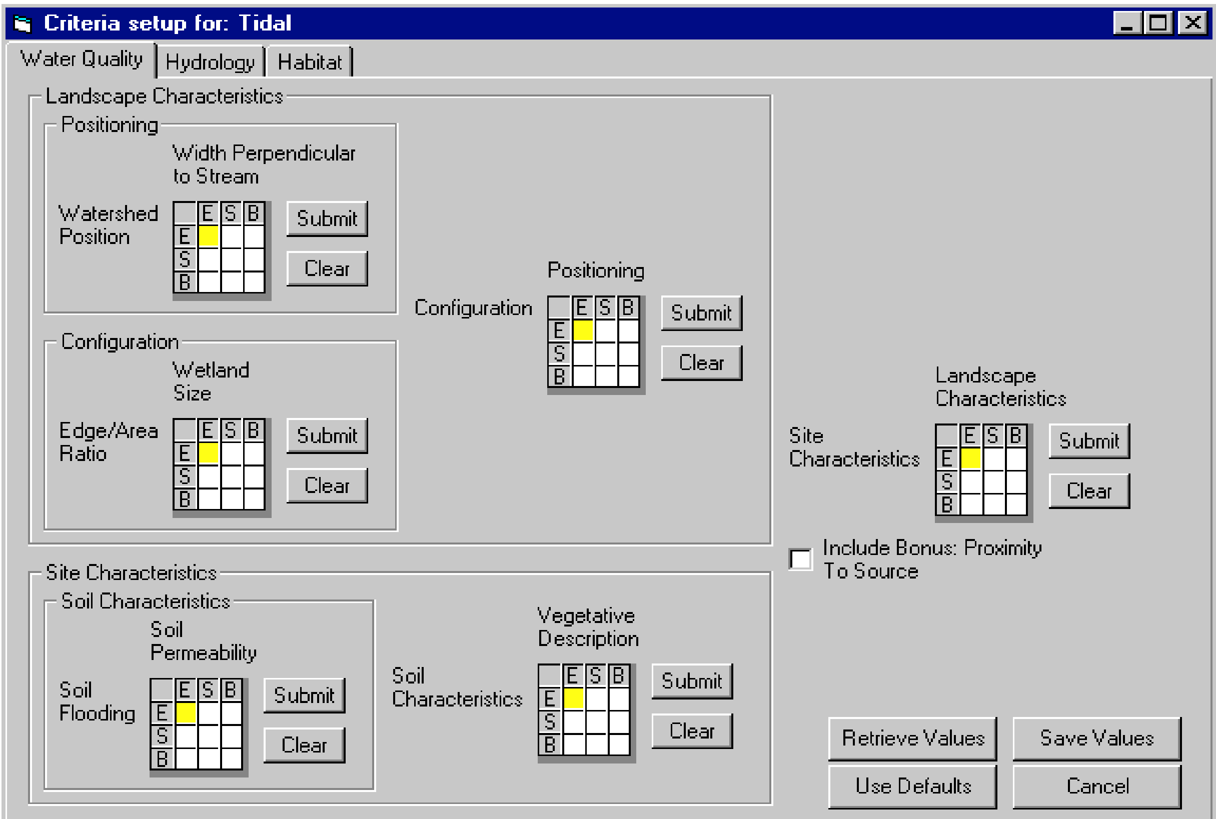


Figure 4. Another option for combining parameters in SWAMP is selecting combinations of coupled parameters, as seen in this example.

Figure 5. The third option, allowing the greatest flexibility in combining SWAMP parameters, is the matrix shown here. Users select the combination of the parameters shown on the edges of the matrices to determine what combination of results leads to a rating of exceptional, substantial or beneficial.



Application Considerations

SWAMP allows a repeatable method of evaluating wetlands. SWAMP can be a meaningful tool in evaluating wetland sites for prioritization and restoration when qualified wetland professionals consider the context and rules for the decision rules they develop. It is a useful tool for organizing, assembling, and visualizing wetland data for the purpose of examining wetland function and developing relative priorities within a watershed.

Data loading and parameter analyses take a minimum amount of time with current computer technology, and the display of results for decision rules is almost instantaneous. The speed and acceptance of this model allows planners to develop scenarios for alternative development patterns. The results of such scenarios can be used to choose alternatives that have the least impact on wetlands that contribute the most to a watershed's water quality, hydrology, and habitat. SWAMP also can be used to prioritize wetland sites for acquisition by identifying the sites that are most likely to maintain the watershed's ecological integrity. Finally, SWAMP can help to support sites identified as potential wetland enhancement or restoration sites by suggesting which function on the landscape they would most likely support if restored. As with any model, the results of SWAMP are no better than the data and information provided by the user.

The work completed on SWAMP to date has been a proof-of-concept to ensure that the analyses required could be performed in an ArcView environment. Model developers will be investigating opportunities to explore more scientifically rigorous parameter thresholds and integration with other numerical and predictive models. The NOAA Coastal Services Center also is investigating the application of this modeling approach to the West Coast of the United States.

When transferring the technology to another geographic region, it is important to employ a wetland professional (or a team of experts) to verify the accuracy of the source data and to determine which parameters should be used. Thresholds categorizing the data into various levels of significance must also be developed. Once the parameters and thresholds are finalized and written into the ArcView project, the decision rules used by managers should be verified with the wetland expert(s). This step is necessary to ensure that the complex rule assignments being made make ecological sense.

To increase the utility of this modeling approach, it would be beneficial to research the links that can be made to other numerical and predictive models. These models could provide the data to support one or more parameters and more effectively separate the thresholds for the different levels of significance.

Acknowledgments

The author would like to acknowledge the work of the North Carolina Department of Environment and Natural Resources Division of Coastal Management in the development of the original model concept. SWAMP would never be complete were it not for Jeff Cowen, the skilled programmer at the NOAA Coastal Services Center who made all my thoughts actually work within a desktop environment. Sincere gratitude also is extended to the participants of the ACE Basin workshop held in November 1998, and especially to Pace Wilber for encouraging the development of this model and interface.

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A GIS-based Integrated Wetland Assessment and Restoration Targeting Tool

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Abstract

For the past five years, Maryland Department of Natural Resources (DNR) has been developing GIS-based watershed/landscape assessment and targeting tools for evaluating resource condition and targeting areas for restoration or protection. The DNR recently merged these efforts to create an integrated decision-support framework that is now driving funding and staffing allocation decisions. This framework, called the Integrated Natural Resource Assessment (INRA) contains several components, three of which are relevant to the management of wetlands. They are: 1) a multi-scale watershed assessment and prioritization tool (the Integrated Watershed Analysis and Management System or IWAMS), 2) the Green Infrastructure Assessment (GIA), and 3) a wetland creation/restoration targeting tool (WCRTT). The IWAMS allows users to select indicators and combine and weight them according to their needs. The GIA uses multiple-data layers to determine the most ecologically significant portions of the landscape, and identifies existing and potential corridor connections. The WCRTT utilizes watershed indicators and several data layers to assess physical characteristics to identify and rank potential creation/restoration sites. Functional assessment capability is now being enhanced through the addition of digital elevation model data and enhanced geospatial data from the National Wetlands Inventory. These analytical components are integrated in an Arcview®-based framework to support resource decision-making (e.g., how to manage and where to create wetlands; where to purchase lands and focus restoration areas to protect or enhance landscape function; where to target grant monies for monitoring and restoration activities; and where, strategically, should we focus staff and money).

Introduction

Like most State natural resource agencies, the Maryland Department of Natural Resources (DNR) had evolved over time such that its programs and staff were rather narrowly focused on species or habitat-specific management. Coincidentally, the proportion of DNR's budget that was State general funds has diminished over the last 10 years, resulting in greater dependence on federal and "special" funds to maintain existing programs and services. Given these circumstances, DNR's management team concluded that a shift to a more holistic, ecosystem-based management philosophy would result in greater benefit to the natural resources of Maryland by integrating programs and focusing staff and funds where they were needed most as determined by a more science-based approach to decision-making. In 1995, the DNR began to investigate how to make its management and assessment programs more integrated and ecosystem-based. The goal was to make data and analytical tools more accessible to resource managers so that management decisions would be better based in science and result in greater benefits to the larger ecosystem, instead of a single species or habitat type.

Methods

As part of a departmental re-organization, the Watershed Management and Analysis Division was created to develop integrated assessment methodologies and analytical tools that would take the vast amounts of existing data and allow it to be easily used by line staff and managers. Work began on what has become the Integrated Natural Resource Assessment (INRA). INRA is a statewide assessment that will provide an ecosystem based framework for public and private decisions affecting natural resource management in Maryland. The assessment focuses on three major themes: 1) conducting, and then combining, watershed and landscape assessments, 2) developing an integrated resource management approach that aligns and focuses DNR's collective priorities and opportunities for land conservation and resource management based on ecosystem principles, and 3) promoting major, long-term regional economic opportunities for resource-based industries that support ecosystem integrity. The assessment framework revolves around three questions:

- What resources or features do we value as important?
- What stresses exist which currently or potentially impact "valued" resources?, and
- What is our programmatic response capability to influence decisions relating to land and resource management?

The three components of the INRA that are most relevant to the topic of wetland assessment and restoration are: 1) a multi-scale watershed assessment and prioritization tool (the Integrated Watershed Analysis and Management System or IWAMS), 2) the Green Infrastructure Assessment (GIA), and 3) a wetland creation/restoration targeting tool (WCRTT).

The Integrated Watershed Analysis and Management System (IWAMS)

IWAMS is a multi-purpose natural resources analysis and targeting tool that integrates databases, digital geographic data layers, and analytical tools for water quality, watershed and ecosystem characterization, and targeting for restoration and protection activities. IWAMS facilitates use of quantitative information in land and resource management decision-making at multiple scales using a geographic information system (GIS) interface (ArcView® software) as its integrating framework.

One of the IWAMS modules used in determining wetland creation or restoration opportunities is the "Watershed Prioritization" module. This module provides the analytical tool for conducting INRA comparative watershed assessments and prioritizations. Specifically, it houses the watershed data upon which watershed comparisons are based. It also includes the programs/scripts for running prioritization scenarios, depicting the results graphically (maps, spreadsheets, and charts), and conducting statistical analyses.

IWAMS uses a series of landscape indicators depicting resource conditions, stresses and agency programmatic response capability at watershed scales. Currently, the Watershed Prioritization module of IWAMS is set up to depict, combine and weight indicators at two scales: the Maryland state 8 digit watershed scale (i.e. 138 watershed analysis units statewide averaging 75 square miles in size), which are comparable to federal 11 digit HUCs; and the state 12 digit watershed scale (i.e. 1162 watersheds statewide averaging 8 square miles in size), which are comparable to the federal 14 digit HUCs. Indicators have been developed using GIS data from a variety of sources including DNR data bases, Chesapeake Bay Program and other federal data sets, and other state agencies.

The application of the watershed prioritization module requires the selection and weighting of indicators appropriate to address specific landscape or watershed targeting initiatives. INRA/IWAMS does not preselect indicators for any given targeting application, nor does it assign weights to indicate different "levels of importance" for different indicators. It is up to the user to define the indicators and weights that best address the targeting question at hand.

The INRA framework for assessing and prioritizing watersheds for conservation and restoration involves the steps in Figure 1. The first three steps in the process represented by the flow chart involve the establishment of prioritization goals and the development of indicators to support these goals. Goal setting is typically addressed outside of the IWAMS environment. Indicators may be developed outside of IWAMS and then be added to the IWAMS indicator structure through the addition of tables with a watershed identifier field. The last three steps in the flow chart

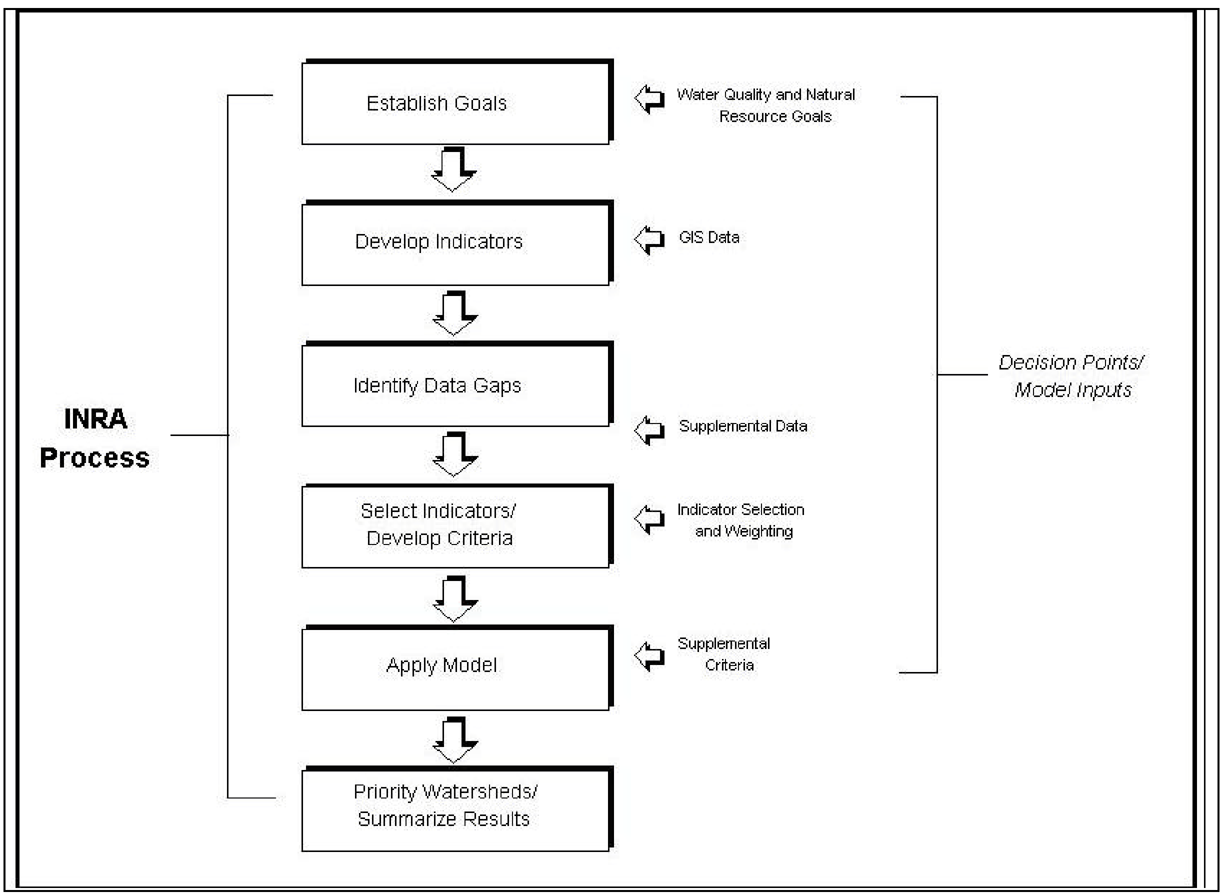


Figure 1 INRA Process for Establishing Priority Watersheds

collectively represent the IWAMS watershed prioritization process.

1. Establish Ecosystem Health, Clean Water or other Natural Resources Goals and Supporting Parameters. This step focuses on establishing the underlying purpose for applying the INRA model in qualitative terms. It provides the context for the analytical process using ecological condition, landscape stress, and programmatic response parameters.

2. Develop Environmental Indicators. Environmental indicators have been developed that are capable of quantifying the parameters of interest geographically. These indicators were calculated at watershed scales and cataloged within IWAMS in a series of related data bases. Indicators selected for priority setting should be relevant to the goals or objectives established in Step 1 (see Figure 2 at end of paper).

3. Identify Data Gaps and Limitations. Where insufficient indicator data exists, surrogate data sets can be used to (1) achieve statewide geographic coverage or (2) account for an ecological feature or process not directly represented by existing, available indicators.

4. Select IWAMS Model Inputs and Develop Criteria for Selecting Priority Watersheds. The most critical step in the INRA process is the development of "decision rules" for applying the IWAMS model -- specifically in terms of (a) selecting the indicators to include, (b) developing the "scale" for scoring individual indicators, and (c) the assignment of weights for combining indicators for use in the composite watershed scoring phase.

5. Apply IWAMS Model. Once indicators and weights are selected, a composite watershed conservation or restoration score is calculated.

6. Develop Summary Report. Finally, indicators used in the analysis can be presented both individually and collectively through the composite conservation/restoration ranking. This process provides a method of hierarchically assessing watersheds based on a variety of indicators, allowing the resource manager to then focus dollars and staff where the need is greatest.

The Green Infrastructure Assessment (GIA)

The Green Infrastructure is a landscape assessment tool being developed to help identify and prioritize areas in Maryland for conservation and restoration (see Figure 3). The goal is to target those areas of greatest statewide ecological importance, as well as those at greatest risk of loss to development. The green infrastructure assessment methodology was developed, in part, to provide a consistent, scientifically defensible approach for evaluating ecosystem conservation and restoration initiatives in Maryland. It specifically attempts to recognize: 1) the role of a given place as part of a larger interconnected ecological system, 2) the value of integrating multiple resource interests into a single framework, 3) the importance of considering natural resource/ecosystem integrity in the context of existing and potential human impacts to the landscape, 4) the importance of regional (i.e., inter-jurisdictional) coordination of local planning, and 5) the need for a regional element to a biodiversity conservation strategy.

The Green Infrastructure land network is based on principles of landscape ecology and conservation biology, and consists of "hubs", "corridors", and "nodes". Hubs are defined as contiguous areas of major ecological importance, at least 100 acres in size. These include:

- Sensitive Species Project Review Areas (SSPRA), which primarily contain rare, threatened, and endangered species in Maryland. Polygons also generally encompass, but do not delineate, such regulated areas as Habitat Protection Areas, Nontidal Wetlands of Special State Concern (WSSC), Natural Heritage Areas (NHA), Colonial Waterbird Sites, and Waterfowl Concentration and Staging Areas.

- Large blocks of contiguous interior forest (at least 250 contiguous acres, plus a 300 ft transition zone)

- Large wetland complexes, with at least 250 acres of unmodified wetlands

- Freshwater aquatic core areas, including the upper Potomac river, Youghiogheny wild river section, brook trout streams, blackwater streams, Maryland Biological Stream Survey (MBSS) complementary streams, stream reaches containing aquatic species of concern, and streams within watersheds with high anadromous fish scores. These were combined and buffered to the flood plain (to maximum of 1000 feet) or 550 feet, whichever was greater.

- Existing protected lands.

Developed areas and major roads were excluded, while adjacent forest and wetland were added, and the edges were smoothed. Hubs that were separated by major roads and/or intervening human land uses, were ranked within their physiographic province using a linear nonparametric combination of 29 ecological variables. Parameters were chosen and weighted according to feedback from biologists and natural resource managers; literature reviews; minimization of redundancy, area dependence, and spatial overlap; balancing different ecotypes; data reliability; and examination of output from different combinations. They were then divided into three tiers.

Model output was reviewed by field ecologists and county planners, and compared to a forest reserve system proposed by Baltimore County's (Maryland) Department of Environmental Protection and Resource Management. Hub locations were largely consistent with existing natural areas according to these sources, although small features and undigitized rare species locations may have been missed, and given the large amount of land involved, the rankings were considered necessary to prioritize acquisition or easement efforts. The percentage of the total area studied, and the percentages of NHA, WSSC, unmodified wetlands, interior forest, and existing protected lands falling within hubs are shown below. The majority of Maryland's terrestrial natural resources fell within a relatively small fraction of the state.

Corridors are linear features linking hubs together, to allow animal and plant propagule movement between hubs, in the hope of creating viable and persistent metapopulations. Corridor identification and delineation was based on many sets of data, including land cover/land use, wetlands, roads, streams, slope, floodplains, MBSS aquatic resource data, and fish blockages. Three different linkages were considered. Within hubs, terrestrial (upland interior forest), wetland (large wetlands or WSSC), and aquatic (as discussed above) core areas were identified, and corridor suitability surfaces were tailored for each ecotype. In general, preference, based on literature reviews, was given to streams with wide riparian buffers and healthy aquatic communities. Other good wildlife corridors included ridge lines, valleys, forest, etc. Urban areas and other unsuitable features were avoided. Least-cost path analysis was used to identify potential linkages between core areas, and these were given width according to the neighboring topography.

Nodes are patches of interior forest (with 300 ft transition zone), wetlands (with an upland buffer), SSPRA, or protected areas along least cost pathways. These serve as "stepping stones" for wildlife movement along corridors.

Gaps are developed, agricultural, or mined lands within hubs, corridors, and nodes, that could be targeted for restoration. Wetlands impacted by dredging, draining, filling, etc., could also be targeted for restoration. Structures such as underpasses or bridges can facilitate wildlife movement where roadways and railways cross corridors and hubs. Similarly, stream blockages can be identified for fish ladders, bypasses, or other structures. Field investigations will reveal additional gaps caused by silvicultural practices, exotic species, etc. Gaps are caused by natural disturbances as well, and are a vital part of healthy landscapes, but these should not be targeted for restoration.

Prioritization efforts for protection and restoration of Green Infrastructure elements are ongoing. Hubs and corridors were ranked for a variety of ecological and development risk parameters, as well as combinations of these, within their physiographic region (see Figure 4). Prioritization of the Green Infrastructure was also done by individual grid cell (approximately a third of an acre) for ecological importance. Vulnerability to development is also being addressed at this scale. This finer scale allows a more detailed analysis for site prioritization within hubs. Finally, gaps within hubs and corridors are being prioritized for restoration efforts, according to their ecological benefits and reclamation ease.

The Green Infrastructure will continue to be refined based on input on ecological significance and landscape risk to development from local governments and other organizations. Finally, an effort is underway to better integrate the assessment methodology with the Mid-Atlantic Gap Analysis Project. As a result, the green infrastructure will be modified to incorporate additional biological diversity conservation information derived from GAP.

Implementation of the Green Infrastructure Assessment will be preceded by photographic and field assessment. Because of limitations in data resolution, maps of model output are only meaningful at a 1:100,000 scale or smaller. Acquisition dates varied between 1980 and 1997, mostly between 1991-7. For site-specific planning, maps should be photo and field verified, and boundaries defined using aerial photographs and property maps. Figure 5 depicts an example of delineating a randomly sampled riparian corridor using aerial photography. Field assessment will also allow discrimination between mature forest and successional or regenerating forest, or between natural forest and pine plantations. Data from ground surveys can be combined non-parametrically, as the GIS ranking of hubs and corridors; or standardized, weighted, and combined (as in HGM wetland assessment or the Baltimore County forest reserve assessment). Areas ranking high in the landscape-scale GIS assessment, but low on the ground, should be given lower priority for conservation than areas ranking high in both categories.

Wetland Creation/Restoration Targeting

Several recent initiatives in Maryland that have made the development of a GIS-based wetland creation and restoration targeting tool a necessity. In May 1997, Governor Glendening committed the State to the creation of 60,000 acres of wetland. That same year, the General Assembly passed a comprehensive package of "Smart Growth" legislation, which included the Rural Legacy program. The Rural Legacy program is designed to preserve the rural character of those portions of the state with high natural resource and agricultural values, as well as to have landowners, through enhanced incentive programs, optimize the installation and use of best management practices, including wetland restoration or creation. Another program that targets wetland restoration or enhancement is the Conservation Reserve Enhanced Program, which has set an independent goal of restoring 10,000 acres of wetlands through State, federal and nonprofit group funding of up to 100 percent of the costs. Since 1991, the Nontidal Wetland Protection Program has had a goal of an overall net gain in wetland acreage and function. Permits are required for activities which impact wetlands and mitigation is required of permit recipients for larger permitted losses of wetlands. In order to meet the net gain goal, the program actively seeks out opportunities for wetland creation and restoration. Lastly, the Green Infrastructure Assessment identifies important lands for both protection and restoration. Wetlands (existing and restorable) are important features of the Green Infrastructure.

To support these efforts, a wetland creation/restoration targeting tool (WCRTT) has been developed as a module within IWAMS. The Watershed Prioritization module prioritizes watersheds statewide, based on user defined criteria, and then within watersheds, the WCRTT locates potential creation and restoration sites, as well as providing managers with landowner contact information from a digital property tax and assessment information known as Maryland Property View.

Two key underlying principles of the WCRTT are to locate potential wetland restoration/creation sites where costs to construct will be low and functional value will be high. Development of the WCRTT initially focused on the first principle, with work now ongoing focusing on the second.

The location of soils with hydric characteristics as determined by available statewide spatial data sets is the foundation upon which the targeting protocol is based. Existing wetlands are not considered potential sites to construct a replacement wetland, therefore potential target areas have existing wetlands removed from consideration. Current land use is an important consideration for locating restoration sites, so the polygons representing potential wetland creation/restoration areas have been tagged with their current land use.

Preparation of the targeting data layer started with the Maryland Office of Planning Natural Soils Groups data layer being used as the basis for defining wetland restoration targeting areas. All the map codes identifying hydric soils or near hydric soils were extracted from the Natural Soils Group data layer to create new shape files consisting only of the selected soils. Each soils group was given a weight reflecting it's conduciveness to wetland restoration/creation. For example, very poorly drained soils with high water tables were given the most weight and moderately drained soils with a water table usually at 24 inches below the surface were given less weight. These shape files were merged into regional files and the regional files were merged into one statewide shape file. NWI digital files were then joined with the hydric soils layer, and any hydric soils polygons identified as wetlands by the NWI layer were subtracted leaving only hydric soils not classified by NWI as wetlands. This data layer is known as "non-wetland hydric soils."

Land use from the Maryland Office of Planning's land use files was intersected with the non-wetland hydric soils data layer and the land use code of each polygon was copied to the data base of the target soils file. As with soils group, land use codes were given different weights according to their suitability to wetland restoration/creation. Forested and developed land use classifications were given the lowest weights while classifications such as sand and gravel pits, and various agricultural uses were given high weights. The 6-digit, 8-digit, and 12-digit hydrographic unit codes were copied to the database of each non-wetland hydric soil polygon to enable the data set to be sorted by any scale of watershed deemed appropriate by the end user.

The WCRTT allows the user to choose and weight indicators with which to develop a wetland restoration/creation suitability index. A user could choose just soils data, or soils plus land use plus watershed-based indicators residing in IWAMS, and weight each indicator. The WCRTT then produces a numeric value for each non-wetland hydric soil polygon from 0 to 10, with 10 being the highest suitability for wetland restoration/creation.

Once the user has shifted from the regional landscape view toward the parcel level in the targeting process, additional data becomes useful. For example, a stream layer is used to determine the relationship of the non-wetland hydric soils to streams. The same concept applies to wetlands and can be accomplished by bringing the digital wetlands layer back into the system. A whole host of functional related queries can be made using these data sets. Also, with a road or highway data layer, the proximity and juxtaposition of a potential wetland restoration site to roadways can be analyzed for either positive or negative impacts. For targeting wetland creation or restoration at the parcel level, the most useful additional data layer is Maryland Property View which contains property unit centroid and ownership information. Once the user has narrowed down the search for suitable restoration/creation sites, the property view layer is added enabling the user to retrieve property ownership information for any potential outreach efforts.

Future directions for the WCRTT include the addition of enhanced functional assessment capabilities through a cooperative effort with the U.S. Fish and Wildlife Service. Specifically, wetland loss or gain trend analysis will be completed (which will also be used to create a watershed-based indicator for use in the suitability index), and a buffer analysis will locate stressors that are likely to have adverse impacts on water quality or wildlife habitat quality. Next, wetland classifications will be enhanced to include HGM-type descriptors (landscape position, landform, and water flow path). Additional photo-interpretation will be performed to review potential wetland restoration sites identified previously by the WCRTT to identify their suitability for restoration. Each potential site will be classified as either a Type 1 restoration site (former wetland) or Type 2 site (existing functionally impaired wetland) and then further classified based on the nature of the restoration needed (e.g., fill removal, hydrologic correction, etc.).

The project will also entail developing protocols for identifying wetlands of potential significance for a variety of functions such as surface water detention, streamflow maintenance, sediment/other particulate retention, nutrient cycling, fish habitat, waterfowl and waterbird habitat, other wildlife habitat, rare habitats, and maintenance of biodiversity. A watershed-based wetland characterization report will be generated and will include a summary of wetlands by various types (NWI types plus HGM-descriptor types), a tabulation of the extent of ditching, an assessment of the condition of wetland and stream buffers, an inventory of potential wetland restoration sites (by site type and nature of restoration required), and a preliminary assessment of the functions for wetlands in the subject watershed.

The study will also involve developing an ecological integrity index or similar index to reflect the amount of natural areas (uplands, wetlands, and water bodies) remaining in the watershed and the condition of these features plus the buffer around wetlands and water bodies. This index will utilize both data newly developed from this project and pre-existing land use/cover data from the Maryland's geographic information database. (Note: The NWI report is posted on website: wetlands.fws.gov where it is listed under reports and publications.)

Tying It All Together: Doing Integrated Assessments and Targeting

INRA and IWAMS provide a framework that allows for watershed and landscape assessments at various scales using data covering aquatic and terrestrial habitat, water quality, human induced alterations (such as percent of watershed in impervious surface), and socio-economic conditions (such as population density). Watershed-based information allows users to prioritize watersheds from a statewide or regional perspective for management action. Once one or several watersheds are selected for more in-depth analysis, landscape data and tools such as the GIA and WCRTT can be used by themselves or together to identify specific areas of the landscape for protection or restoration activities.

For example, let's assume there is a need to start from a statewide perspective and locate wetland creation or restoration sites where the greatest ecological benefit will occur, and that our funding source is incremental Nonpoint Source Program funds (§ 319) from the Clean Water Action Plan. To begin the assessment, IWAMS is utilized to assess (at the 8-digit watershed scale) the Priority Category 1 watersheds (i.e., those in need of restoration) as determined by the Unified Watershed Assessment, and prioritize those watersheds using the "estimated wetland loss" indicator (see Figure 2). The prioritization results show that the Upper Pocomoke watershed on Maryland's lower eastern shore has lost the largest acreage of wetlands (80,903 acres of wetland loss).

Because the "estimated wetland loss" indicator is based on GIS derived landscape data and not field collected or monitored data, it is possible to look within the Upper Pocomoke watershed and prioritize the 12-digit watersheds using the same indicator. The prioritization of the 12-digit watersheds results in relatively small areas (averaging 8 square miles) being ranked based on wetland loss (Figure 7).

Simultaneously with the multi-scale watershed assessment, landscape assessment with the GIA is completed for the Upper Pocomoke watershed. This assessment will identify the significant ecological features as well as "gaps", which are prime potential restoration sites (Figure 8).

The highest ranking 12-digit watershed boundaries are then intersected with the GIA results to determine potential restoration sites or gaps within the green infrastructure of the 12-digit watershed. Once gaps are identified, the WCRTT is applied to determine the suitability of these areas for wetland creation/restoration (Figure 9). If any areas appear to be suitable for wetland creation/restoration, property owner information is obtained from Maryland PropertyView, and landowners are contacted to determine their interest in habitat restoration. A field survey is then conducted to determine the area's desirability from both a physical and cost perspective.

Conclusions

Technology and data now exist that allows resource managers to conduct integrated assessments and prioritizations for targeting restoration and protection programs and activities. The hope is that these tools will allow managers to be more strategic in where resources are focused, resulting in greater measurable benefits from restoration and protection activities. The use of GIS can save time, and produce technically defensible results. Many programs that historically have not used this type of information are seeing the benefits and are providing more support for these kinds of efforts. With data sets such as satellite imagery becoming more widely available at increased resolution, more explicit spatial analysis, including trend analysis, will be more routine.

The Department of Natural Resources is basing much of its strategic planning on the capabilities of the kinds of assessment described above. With the Internet becoming more prominent as a vehicle for service delivery, the Department is hopeful that the results of GIS-based assessments will be available to the public in the near future.

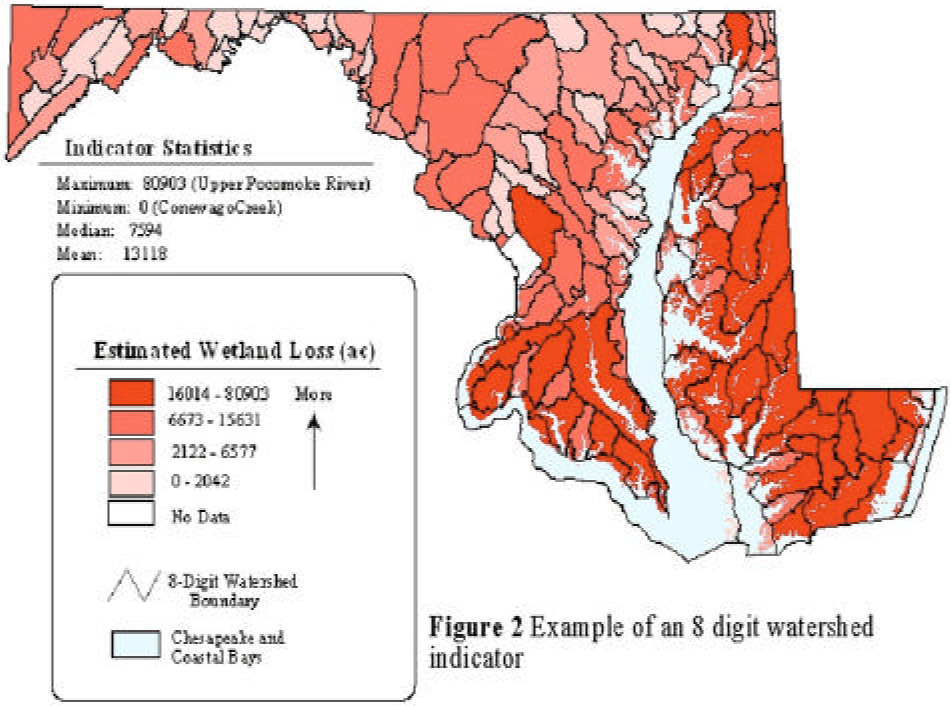


Figure 2 Example of an 8 digit watershed indicator

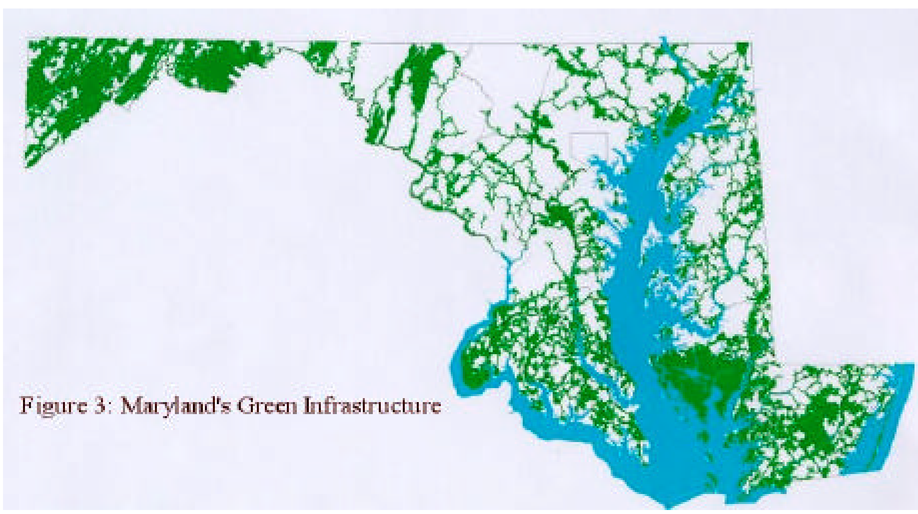


Figure 3: Maryland's Green Infrastructure

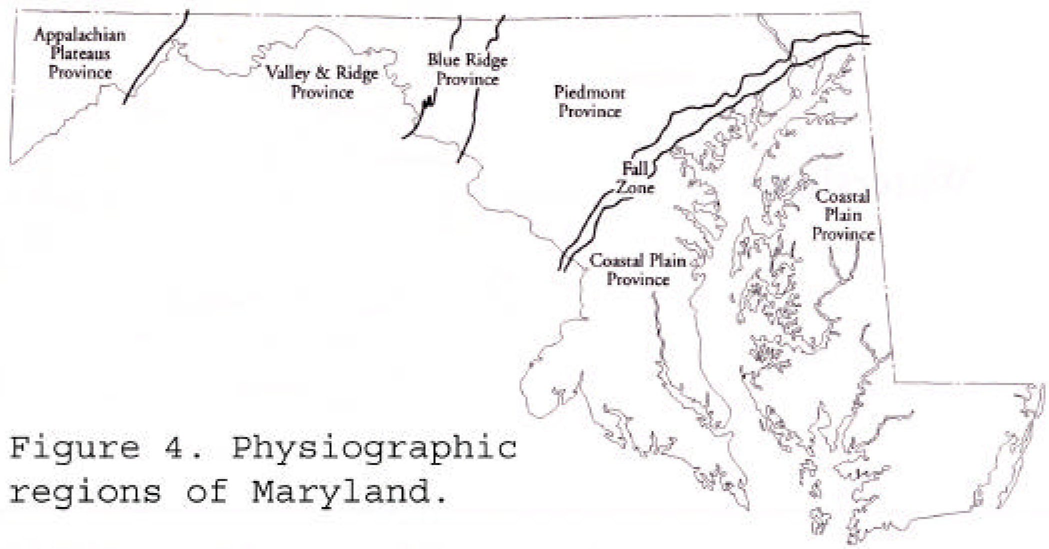
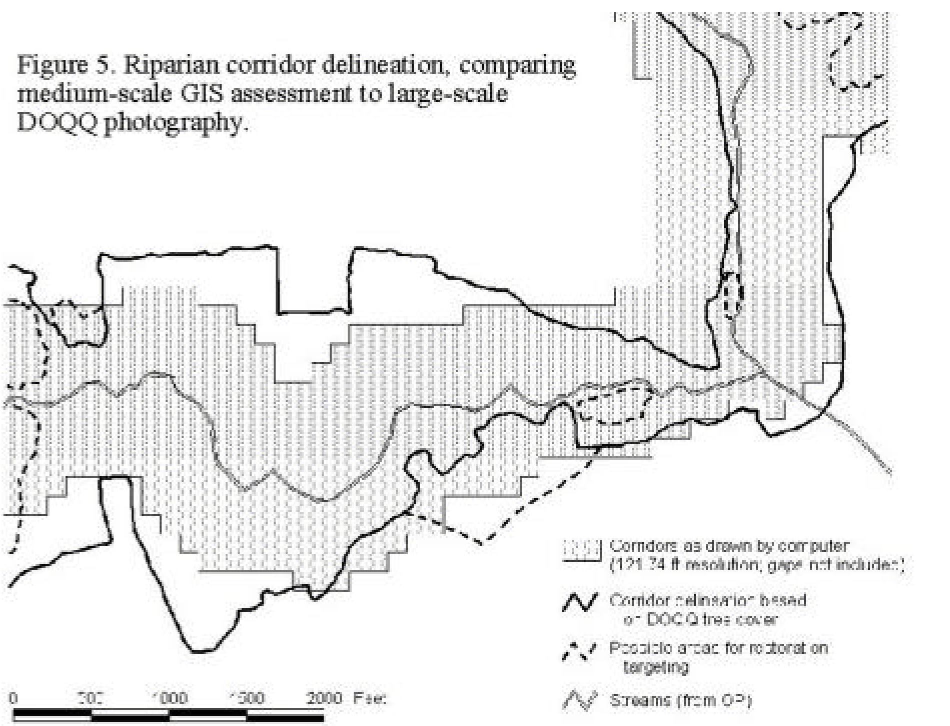


Figure 4. Physiographic regions of Maryland.

Figure 5. Riparian corridor delineation, comparing medium-scale GIS assessment to large-scale DOQQ photography.



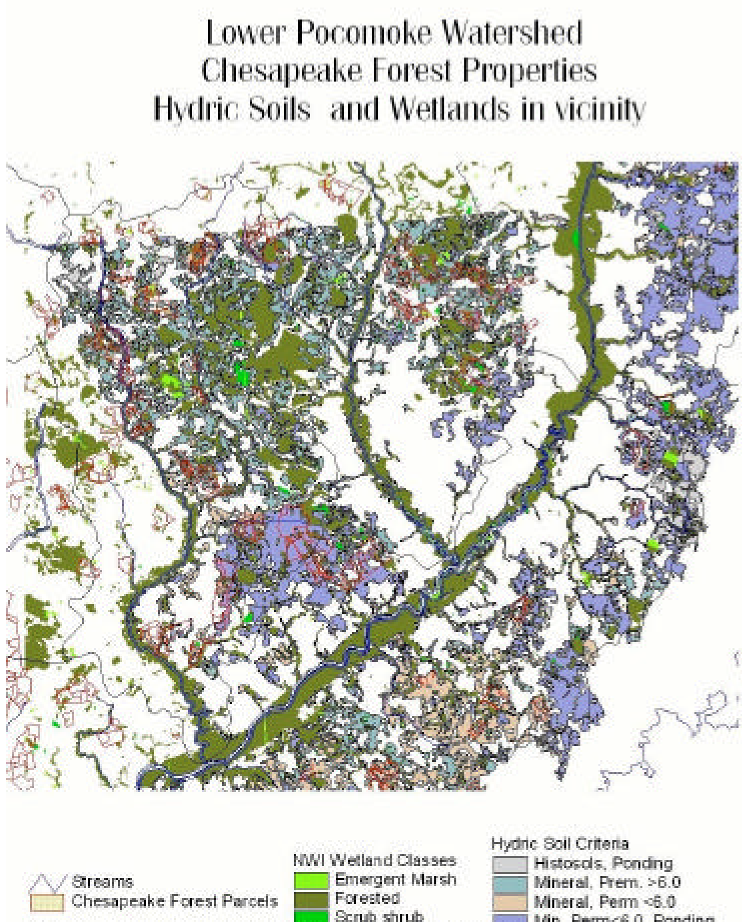


Figure 6 WCRTT results of non-wetland hydric soil assessment on newly squired public land. Blue areas are non-wetland hydric soils with specific characteristics

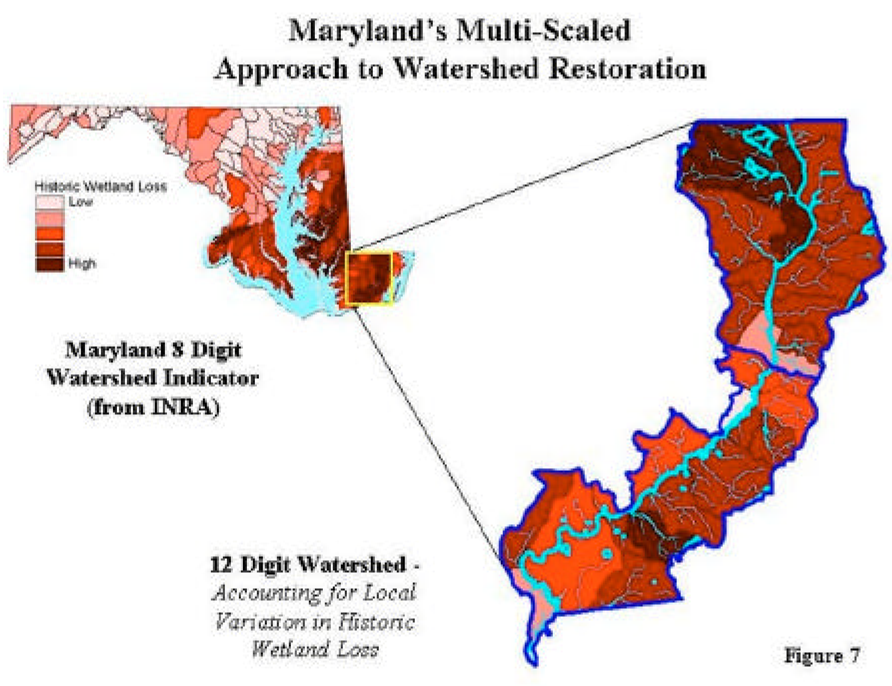


Figure 7

Identifying Gaps in Vegetated Cover within the Green Infrastructure

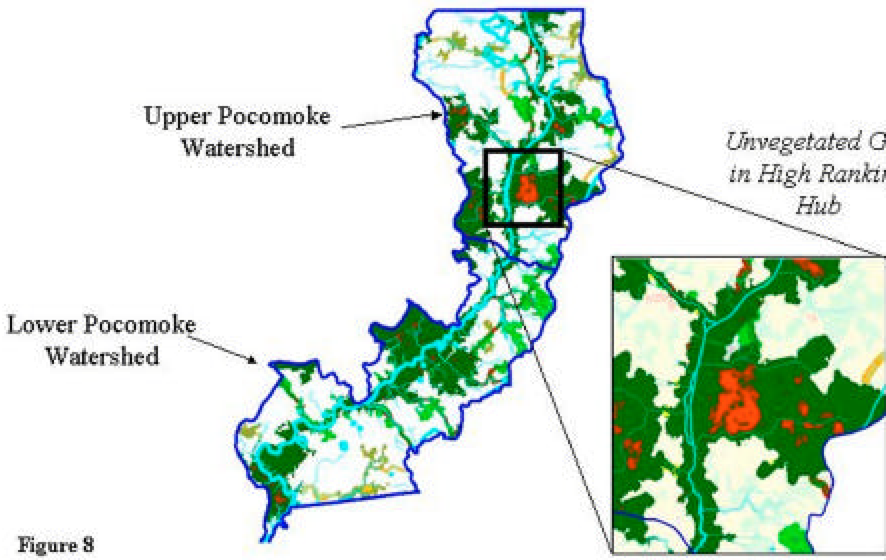


Figure 8

Identifying Candidate Wetland Restoration Sites within Green Infrastructure Gaps

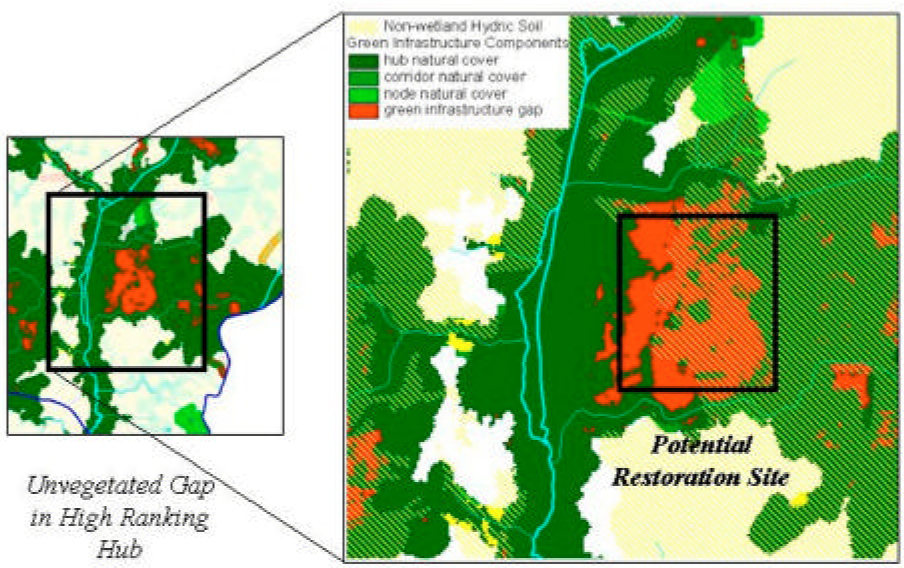


Figure 9

The Need for Process-driven, Watershed-based Wetland Restoration: The Washington State Experience

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Abstract

The continued decline in the health of aquatic species and ecosystems indicates that something is dramatically wrong with our current approach to resource management in the Pacific Northwest. Causes for this lack of success fall into two general areas. First, very complex ecosystems have been over-simplified or dissected into individual parts to facilitate regulation and management. Second, existing regulations and recovery efforts typically focus on structural components at a site scale. Considerable evidence suggests that process-driven, watershed-based tools that look at multiple spatial and temporal scales need to be developed to provide the conceptual framework for organizing and coordinating management and recovery actions at the site scale. In 1994, Washington State Department of Ecology initiated a landscape-scale wetland restoration program to meet an objective of the Puget Sound Water Quality Management Plan. In 1998, an interdisciplinary technical team was assembled to build on this work in concert with emerging concepts in the literature to develop tools that help recover threatened and endangered salmon runs, improve degraded water quality, and address causes of increased peak flows and declining stream baseflow. This paper presents insights gained during development and initial implementation of these landscape-scale process-based assessment efforts.

Introduction

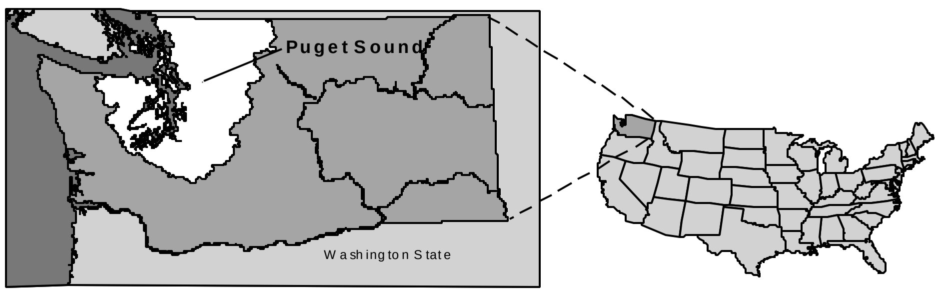
The Puget Sound region of Washington State is a unique and unparalleled ecological resource. Located in northwest Washington State (Figure 1), this rich and varied ecosystem is the result of the region's great topographic diversity and the interactions of physical, biological, and chemical processes at many spatial scales. These processes create the ecological diversity and interdependent relationships that produce Puget Sound's abundant natural resources and striking beauty.

Contributing to the regions unique qualities are an equally diverse and abundant suite of wetland resources that include estuarine salt marshes, large forested floodplain wetland complexes, *Sphagnum* bogs, glacial kettles, rugged high elevation meadows, and extensive fens. The distribution and diversity of these aquatic systems are the result of local geomorphology, climate, and disturbance history (Naiman and Anderson 1997).

While these wetland resources seem immense, they are finite. As in other parts of the United States, agricultural, commercial, and residential development has resulted in a substantial loss of wetland resources. Estimates of statewide wetland loss vary from 33 to 50 percent, with Puget

Sound experiencing losses to tidally influenced emergent wetlands in excess of 70 percent (Canning and Stevens 1989). One study documented wetland losses of over 95 percent in some urbanized areas (Bortelson 1980).

Figure 1. Puget Sound region of Washington State.



Adding to these extensive wetland losses is the very real threat of future resource degradation. At least 3 factors significantly threaten Puget Sound's natural resources: 1) the number and distribution of people, 2) the amount of resources they consume, and 3) the waste they produce. In 1950, Washington State supported 2.4 million people. Today, the state's population exceeds 5.6 million people and, if projections are correct, the states population will grow to 7.7 million by 2020 and 11 million by the year 2045 (Washington State Department of Natural Resources 1998). This level of population growth means that over the next 45 years, the State of Washington will add the equivalent of 29 new cities the size of Tacoma or Spokane (Washington State Department of Natural Resources 1998), and much of this growth will occur in the Puget Sound region.

This rapid growth, and land use decisions associated with that growth, has resulted in the listings of salmon and steelhead under the Endangered Species Act (ESA), water quality degradation to over 750 water bodies (Washington State Department of Ecology 1998), increased peak flows in urbanizing streams, and a decline in some stream baseflows. The troubling status of these key natural resources indicates that something is dramatically wrong with our current approach to resource management in the Pacific Northwest (Karr 1995). It is clear that the existing regulatory framework and implementing agencies have fallen short of expectations (Governor's Salmon Recovery Team 1999).

Causes for this lack of success are open to debate and potentially numerous. It is becoming more apparent that a lack of watershed-based tools restricts development of a conceptual framework for organizing and coordinating recovery actions (Alder 1995; Angermeier and Schlosser 1995; Frissell 1996). I suggest two generalized problem areas. First, the ecological context and nexus of natural systems is often lost as resource managers specialize in a single component of an ecosystem (e.g., wetlands, stream channels, lakes). This has resulted in very complex ecosystems being over-simplified to facilitate regulation and management. While classification systems have been essential in the study and assessment of individual resource components, they can lead to a level of confidence in specialization that impedes our ability to understand the system as a whole. For example, the alluvial floodplains of virtually all major river systems flowing into Puget Sound were once an intricate interconnected mosaic of main channel, side channel, wetland, forested riparian, and hyporheic systems that were constantly being reworked by natural disturbance factors within and outside of the floodplain area. While substantial work has been done to study wetlands, riparian systems, stream channel morphology, and more recently the hyporheic zone (Naiman and Anderson 1997), ecologists are just beginning to see and understand the system as a whole (Naiman et al. 1992). I suggest that resource managers must first recognize and understand the complex interactions of natural systems as a whole to establish the context for the detailed study of a systems individual resource components.

Second, the recurring need to address individual species (e.g., chinook salmon) or species guilds (e.g., migratory waterfowl) for economic or social reasons tends to direct resource managers to site-specific, structure-based components of a species habitat and not on the landscape-scale processes that create and maintain habitat structure. This has led to a dependence on engineered structure-based fixes to resource problems. Fisheries biologists in the Pacific Northwest have often focused research on site- or reach-specific habitat conditions and the productivity of important life stages of individual species of salmon or steelhead. An example of this type of assessment comes from the Salmon Recovery Planning Act (ESHB 2496) of 1998 that requires a limiting factors or reach-specific structure-based analysis of habitat bottlenecks for habitat restoration (Governor's Salmon Recovery Team 1999). While site-specific structure-based assessment is an essential component of a resource recovery plan, it is not the only essential component. A growing body of evidence indicates that assessing habitat forming ecological processes at landscape scales (Beechie and Bolton 1999; Kauffman et al. 1997; Montgomery et al. 1995; Naiman et al. 1992) establishes the needed context for directing assessment at finer scales.

The combination of past resource degradation, rapid growth in the future, and continuing resource degradation will put tremendous pressure on Puget Sound's remaining intact natural resources. However, substantial opportunities for change exist as local jurisdictions, tribes, and state and federal governments respond to the ESA listings of salmon and steelhead. Over the past two years, key pieces of legislation have been passed that recognize the need for comprehensive, scientifically-based, and locally-implemented solutions (Governor's Salmon Recovery Office 1999). Moreover, substantial resources are now being committed at the local, state, federal, and tribal levels to stop resource degradation and begin to reverse past losses.

As early as 1990, the Puget Sound Water Quality Authority (1990) called for the development and implementation of a watershed-based non-regulatory wetland restoration program to assist in reaching the goal of restoring and protecting the biological health and diversity of Puget Sound (Puget Sound Water Quality Authority 1990). From the program's beginning in 1994, a conceptual framework for landscape-scale wetland restoration planning in Puget Sound was established (Gersib et al. 1994), and methods development began in the 719-square mile Stillaguamish River Basin (Gersib 1997) and refined in the Nooksack and Snohomish River Basins.

One fundamental goal of the wetland restoration program is to use wetland restoration to help solve problems important to river basin residents. With this goal in mind, it soon became clear that while wetland degradation was an important factor that adversely affected fish habitat and water quality and quantity, it was only one of sometimes many degradation factors that have cumulatively resulted in the current problems that river basin residents are experiencing. Lessons learned from this program served as the catalyst for a broader interdisciplinary river basin characterization effort that looked more holistically at resource degradation and recovery.

This paper describes these two developing landscape-scale recovery tools and shares lessons learned through their development and early implementation. The purpose of this paper is to stimulate thought and discussion that helps refine and expand existing landscape-scale process-based concepts for ecosystem recovery.

Study Area

The Puget Sound region lies in the northwest part of Washington State and encompasses an area of approximately 17,400 square miles, about 2,556 square miles of which is salt water (Vaccaro et al. 1998). Described regionally as the Puget Sound Lowland, this area is bounded by the natural hydrologic divide lying near the Canada-United States boundary on the north, the drainage divide of the Cascade Range on the east, and on the west by both the drainage divide of the Olympic Mountains and the Strait of Juan de Fuca. The southern boundary is approximately defined by the extent of Pleistocene glaciation, which is characterized by a series of low hills that lie south of Olympia, Washington (Figure 2).

Topographic features in the Puget Sound Lowland are the result of tectonic events and repeated continental and alpine glaciation that occurred during the Tertiary and Quaternary periods (Jones 1996). Low elevation areas are typically alluvial river valleys surrounded by glacial outwash and till plains. This area has a mid-latitude humid, Pacific Coast marine climate that results in a native land cover dominated by dense coniferous forests.

Seventeen major Water Resource Inventory Areas (WRIA) have been delineated within the Puget Sound Lowland. While most WRIAs are primarily one contiguous drainage system (e.g., Snohomish or Nooksack River Basins), a few, like the San Juan Islands, are made up of a series of smaller disconnected drainages (Figure 2).

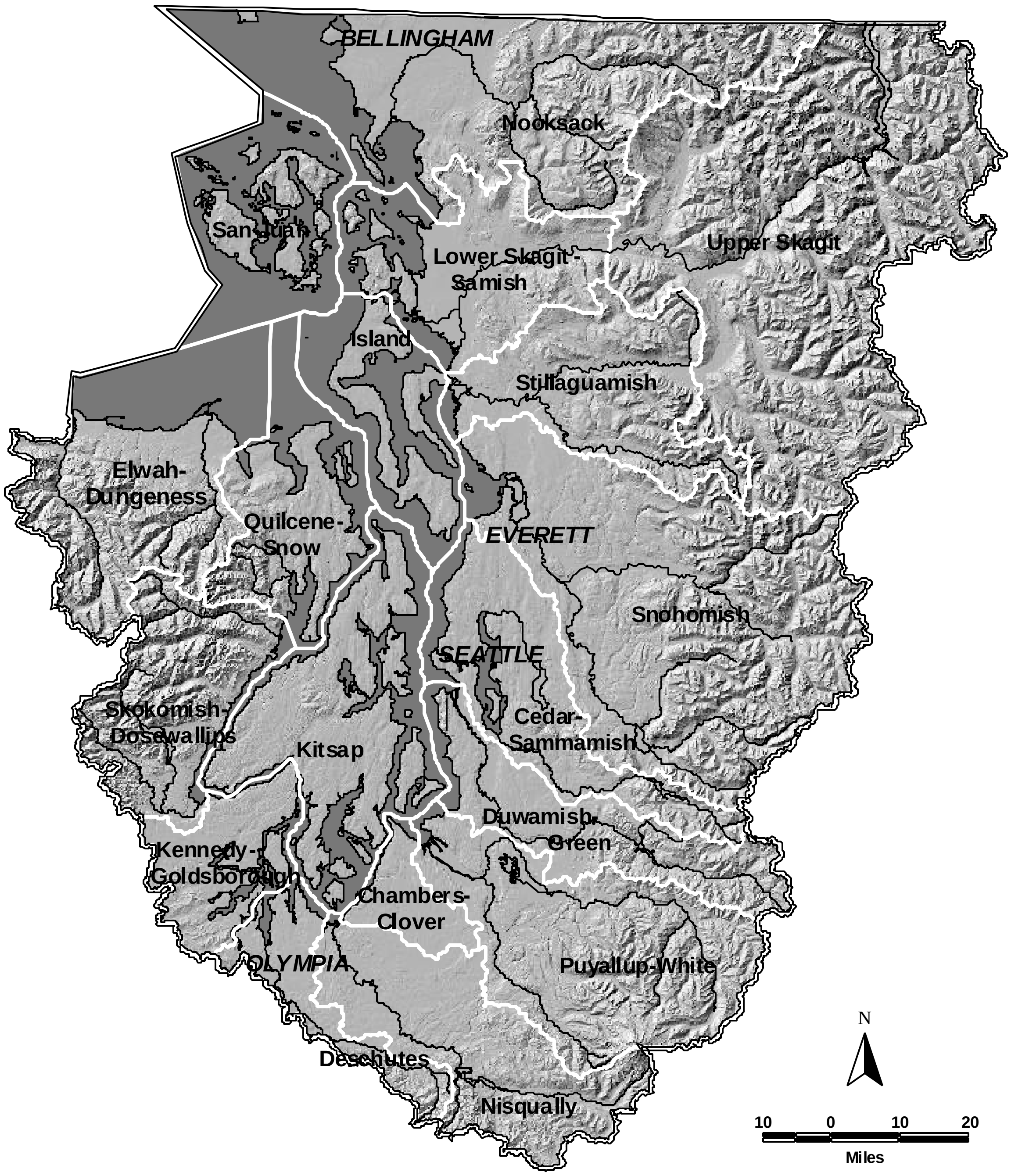


Figure 2. Puget Sound Lowland drainages.

Elements of Watershed Tools

Washington State Department of Ecology (Ecology) has initiated two landscape-scale characterization efforts to support natural resource management decision-making. These initiatives include the Puget Sound Wetland Restoration Program and River Basin Characterization. An overview of each is presented here to establish the context for a discussion of lessons learned.

Puget Sound Wetland Restoration Program

The Puget Sound Wetland Restoration Program developed from clearly stated guiding principles that served as the philosophical building blocks for this program. From these building blocks came the following program goals.

1. Restore and maintain wetlands of sufficient quantity and quality to assist in meeting the Puget Sound Water Quality Management Plan purpose of restoring and protecting the biological health and diversity of Puget Sound.
2. Identify ecological problems within watersheds and, where wetland restoration can address problems, reestablish lost or degraded natural functions.
3. Identify community needs within a watershed and restore natural wetland functions which contribute to meeting human health, safety, and quality of life needs of residents.
4. Support state and federal policy goals of no-net-loss and a long-term net gain in acreage and function of wetlands.

From these goals and input from both technical and non-technical advisory groups, the following key components of a wetland restoration program conceptual framework were identified.

1. The program should be a voluntary, non-regulatory initiative.
2. Wetland restoration should be a coordinated public/private approach that seeks to help solve problems and meet needs of residents.
3. Assessment work should use public input and existing technical data.
4. Information and technical assistance must be accessible to all natural resource managers.
5. Implementation should not mean just producing a database but teaching local resource managers to use this tool and working with them in cooperative efforts with landowners to facilitate wetland restoration.
6. Most resource managers are not wetland restoration ecologists. Technical information and support are needed before most managers can consider wetland restoration as a viable resource management tool.

Methods development began with the following constraints.

1. Past wetland restoration planning has been on an opportunistic site-by-site basis.
2. Existing wetland inventories were often ineffective at delineating forested wetlands and did not specifically inventory potential wetland restoration sites.
3. Limited tools were available to efficiently evaluate many wetland restoration sites at large landscape scales.
4. Few, if any, landscape-scale methods are available to characterize potential wetland functions of sites prior to restoration.

In an effort to address these constraints and allow for more focused planning, a process termed '*wetland function characterization*' was developed. From a landscape perspective, this tool identifies potential wetland restoration sites and then prioritizes them based on their expected ability to perform key watershed functions once restored.

A local technical work team and a more general advisory group were formed in each river basin to guide program development to meet the specific resource management needs in the river basin. These teams identified important ecological problems and community needs to be addressed, helped develop function characterization models that predict each site's potential to perform a wetland function, and provided insight and Geographic Information System (GIS) data sets for landscape scale assessment.

A GIS coverage of potential wetland restoration sites was then created by overlaying existing wetland inventories (e.g., the National Wetlands Inventory coverage, the Washington Department of Natural Resources waterbodies coverage, U.S. Forest Service wetland data, Washington Department of Fish and Wildlife wetland data imbedded in their Priority Habitats and Species coverage) and hydric soils coverages (e.g., Washington Department of Natural Resources hydric soils, and U.S. Forest Service soils coverage). After dissolving all interior polygons, a coverage was developed that represents the greatest potential extent of pre-development wetlands. Black and white (1:12,000) aerial photos were used to confirm the presence or absence of each potential wetland restoration site indicated by any of these coverages and to identify additional sites visible in the photos. Site boundaries were adjusted or new sites added to form a photo-verified coverage of potential wetland restoration sites. Limited site visits were used to validate assumptions made during photointerpretation.

Each potential wetland restoration site was assigned a series of attributes based on photointerpretation. Development of the attributes has evolved over the past four years, but the overall objective is to describe the current and historic hydrogeomorphic class of the wetland (Brinson 1993; Brinson 1995; Smith et al. 1995), the type of hydrologic and vegetative alterations at the site, nearby land use, restoration potential based solely on existing development at the site, and specific characteristics of the site which can affect the functions it may provide (e.g., percent open water, evidence of groundwater discharge, and area of nonvegetated river bar).

For each river basin, a team of specialists with wetland expertise and knowledge of the area refined and, when necessary, developed new GIS models that use landscape and wetland characteristics to predict which wetlands are likely, once restored, to perform each of 18 wetland functions. Functions currently modeled include:

- Temperature maintenance
- Fecal coliform control
- Sediment retention/transformation
- Nutrient retention transformation
- Groundwater nutrient retention
- Flood flow storage and desynchronization
- Base flow maintenance
- Groundwater recharge

- Amphibian diversity and abundance
- Anadromous and resident fish diversity and abundance
- Migratory water bird diversity and abundance
- Aquatic diversity and abundance
- Rare, threatened and endangered species diversity/abundance
- Food chain support
- Active and passive recreation
- Outdoor education
- Cultural significance/unique qualities
- Shoreline stabilization

Model outputs are incorporated into a GIS coverage and searchable database from which it is possible to produce custom reports indicating those wetlands whose restoration is a high priority in order to address a particular watershed issue such as stream temperature.

An accuracy assessment of the photointerpretation was conducted for both the Nooksack and Snohomish River Basins. During visits to 58 sites in each river basin, the same attributes assigned through photointerpretation were independently assigned in the field. Consistency between aerial photo interpretation and the field assessment was determined and used to refine the methods. Function characterization model validation is planned but, as yet, not funded.

Program implementation in three Puget Sound river basins serve as the foundation for our learning. The 719-square mile Stillaguamish River Basin was the initial system used to develop many of the landscape-scale wetland restoration concepts (Figure 2). This basin was followed by assessment in the 1,624-square mile Nooksack River Basin and the 1,909-square mile Snohomish Basin.

A GIS coverage and database were developed for the Stillaguamish and Nooksack River Basins that provides information on the location, size, and potential wetland functions provided by 1,737 and 3,513 potential wetland restoration sites, respectively. Work continues in the Snohomish River Basin were 5,137 potential wetland restoration sites have been photointerpreted, but is pending functional characterization. A methods document was prepared for the Stillaguamish Basin (Gersib 1997) and is being written for the Nooksack Basin.

River Basin Characterization

With new ESA listings for salmon and steelhead and continuing water quality and quantity problems, it has become clear that current methods for addressing these problems are not working. In late 1997, an ad hoc group of scientists at Ecology was organized to explore available tools for landscape-scale assessment. Using the conceptual framework and lessons learned from the Puget Sound Wetland Restoration Program, this work group recommended the formation of a technical team to develop science-based assessment tools that help address declining fish runs and water quality and quantity problems in the state (Washington State Department of Ecology 1998).

In 1998, an interdisciplinary technical team was formed consisting of a geomorphologist, hydrogeologist, fisheries biologist, two part-time water quality specialists, an ecologist, and a GIS specialist. The goals of this team were:

1. Develop a process-based technical framework for evaluating human impacts to water quality, stream base flow, flooding, and anadromous fish habitat;
2. Assist in developing an overarching recovery framework for integrating efforts seeking to address water quality, stream base flow, flooding, and anadromous fish habitat problems at a river basin-scale; and
3. Provide new technically-sound landscape-scale information that can assist resource managers in selecting areas for preservation and restoration actions that have the greatest potential to result in measurable positive change.

Our overall approach to river basin characterization is based on the following key assumptions.

1. Problems must be assessed at the scale in which they occur.
2. Assessment is needed at multiple spatial and temporal scales to provide the best opportunity to understand cause and effect relationships between human landuse and their effects on water quality, water quantity, and anadromous fish habitat.
3. Ecological processes are the physical agents of landscape pattern formation and maintenance that create and maintain the physical, biological, and chemical features of aquatic resources.
4. Landscape-scale assessment should start at the largest appropriate spatial scale for the specific problem being addressed and advance sequentially through finer landscape scales and levels of analysis.
5. Restoring natural ecological processes result in a self-maintaining system, while simply replacing the structural components of a natural system are not self-maintaining.

The initial step identified key watershed components to be assessed. Using a decision-making technique called "Watson's circles" (Coughlan and Armour 1992), core watershed-scale problems were identified. Then based on existing landscape ecology literature for the Pacific Northwest, core problems were linked to specific ecological processes at a landscape-scale. Operating on the premise that the delivery and routing of water, sediment, large wood debris, nutrients/toxicants/bacteria, and heat are the key ecological processes that create and maintain structure and function in Puget Sound river basins, a pilot project was initiated in the 1909 square mile Snohomish River Basin in Washington State.

The Snohomish Basin was spatially subdivided into 62 sub-basins for analysis (Figure 3). Land use/land cover coverages were developed for the following three temporal scales: a) pre-European settlement using 1870's General Land Office surveyor data, b) current conditions using Landsat imagery, and c) future build-out based on Growth Management Act planning. Team members then used available data and existing technical literature to assess the comparative risk that human land use has altered or will alter each key ecological process at the sub-basin scale.

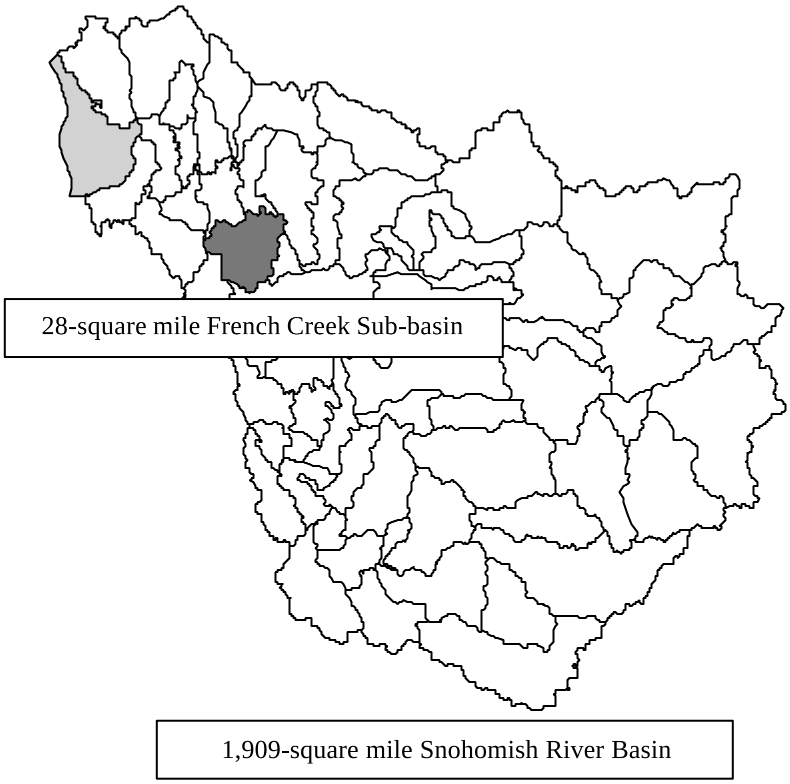


Figure 3. Example of river basin- and sub-basin-scales used in this paper.

Products of this work are a GIS coverage and database that displays the results of comparative risk assessments of key ecological processes in 62 Snohomish sub-basins. A document was developed that presents methods used in river basin characterization and examples of how results can be used to develop a recovery framework for the Snohomish River Basin (Gersib et al. 1999). Presentations on river basin characterization have been made to various levels of resources managers in the state and concepts are being introduced into a number of new policy and regulatory initiatives including Washington's proposed Shoreline Management Act guidelines, The Puget Sound Water Quality Management Plan, and state and regional watershed assessment templates specified by National Marine Fisheries Service final 4(d) rule.

Specifically, river basin characterization:

1. Provides a river basin-scale conceptual framework for ecosystem recovery;
2. Develops new information that supports decision-making at finer scales;
3. Establishes a foundational understanding of the river basin, that is, the core ecological processes that create and maintain ecosystem function, the effects of human development on natural processes, and the resulting water quality, baseflow, anadromous fish habitat, and flood storage/desynchronization functions;
4. Establishes general links between human development and a loss in river-basin function;
5. Helps resource managers understand process alteration in developed or managed areas of a river basin and the potential degree of process alteration under future conditions;
6. Describes pre-disturbance, current, and future conditions of the river basin, when possible;
7. Serves as a coarse sieve to identify sub-basins that warrant further analysis for preservation or restoration;
8. Minimizes potential for conflict associated with single-species management by focusing recovery efforts on the restoration of natural processes that create and maintain ecosystem health.
9. Provides a neutral platform for discussions between neighboring political jurisdictions that need to share in future landscape-scale protection and restoration efforts.

Program Integration

River basin characterization is used to predict where human land use has the greatest risk of altering key ecological processes at a river basin-scale. We see this information as being the foundation that directs where assessment work will be completed at the sub-basin-scale. The Puget Sound Wetland Restoration Program is one component of sub-basin-scale assessment that provides essential information needed to develop cause and effect relationships resulting in a change in key processes. This integration also provides a level of efficiency in that potential wetland restoration site identification and function characterization are done only in targeted sub-basins rather than the entire river basin. Sub-basin assessment, in turn, provides the foundation that directs assessment of individual site functions and feasibility. This hierarchical framework for natural resource characterization and assessment is shown in Figure 4.

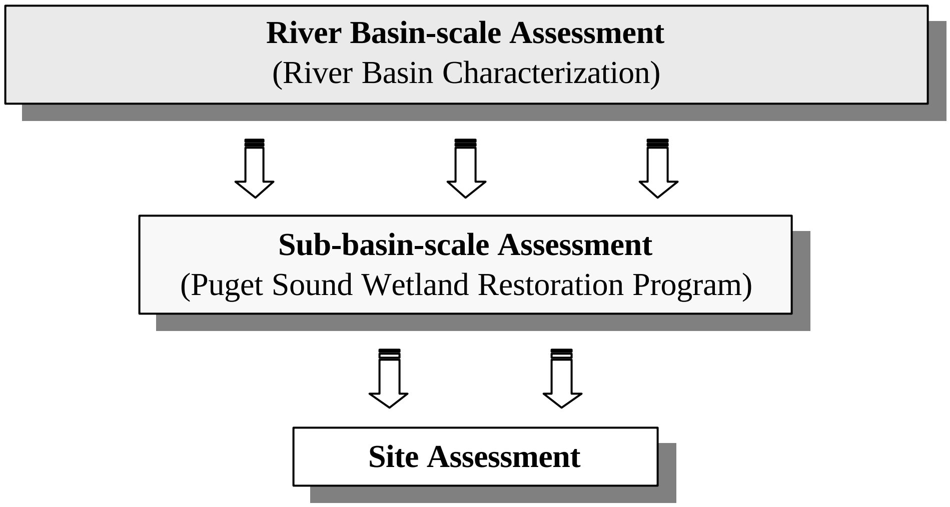


Figure 4. How programs relate in the hierarchical assessment framework.

Discussion of Lessons Learned

The development and initial implementation of the Puget Sound Wetland Restoration Program in the Stillaguamish, Nooksack, and Snohomish Basins and River Basin Characterization in the Snohomish Basin of Washington State have provided unique opportunities to apply concepts discussed conceptually in the literature, but rarely implemented at larger landscape-scales. The following are lessons learned and insights gained through these landscape-scale efforts.

Issues of Scale

The scale of the problem dictates the initial scale of assessment. While recovery efforts are implemented through a series of site-specific actions, assessment of problems must begin at the scale of the problem. This means that if a water quality problem is from a point source, then the problem should be assessed at the site-scale. However, if the problem is depressed anadromous fish populations in the Pacific Northwest, then analysis should begin at the Pacific Northwest-scale and then move through finer scales of assessment. In the case of ESA listed Pacific salmon, the National Marine Fisheries Service has done this large landscape-scale assessment and subdivided the region into Evolutionarily Significant Units (ESU) for management and assessment at finer scales. I suggest that assessment at the site-scale limits recovery to site specific problems. Assessment at a watershed-scale allows recovery opportunities at the watershed-scale. Site-based “fixes” to landscape scale problems is analogous to random acts of kindness to the landscape that are not capable of addressing landscape-scale problems. The scale of the problem needs to be determined as well as the scale at which you are capable of working/doing analysis.

Landscape-scale assessment should include multiple spatial scales. The need exists for more consideration of the scale or scales from which to manage natural systems (Haskell et al. 1992, Franklin 1993). Richards and others (1996) support this by noting that habitats are influenced by factors operating at a number of spatial and temporal scales. River basin characterization demonstrates the value of a hierarchical decision-making tool. Initial work in the Snohomish

Basin was used to direct assessment and recovery efforts to sub-basins having the greatest potential of human-induced process alteration. As more detailed assessments are done in these sub-basins, individual projects will be identified for site-specific assessment and implementation. Without this type of science-based hierarchical tool, assessment would be required in all 62 sub-basins or best professional judgement employed to select key basins for assessment. The cost and time required to do detailed sub-basin scale assessment work in all 62 Snohomish sub-basins would be prohibitive and best professional judgement has not proven to be an effective option based on the lack of success in resource recovery, to date.

As early as 1983, Warren and Liss (1983) describe a classification system that views a landscape as a nested hierarchy of drainage basins. River basin characterization has shown us that we learn different things at different scales. At a river basin-scale, coarse-sieve characterization develops a foundational understanding of the river basin and the ecological processes that create and maintain functions important to people, assesses the comparative risk that human landuse has altered key ecological process, and provides the short-term context for preservation and recovery actions until finer scales of assessment are completed. At a sub-basin-scale, characterization work identifies areas and land use practices that account for the alteration of key ecological processes and establishes a list of priority areas for preservation and restoration. At a site- or reach-scale, projects are comparatively assessed for feasibility and functions gained that results in a preservation and restoration site priority list.

Work done by the Puget Sound Wetland Restoration Program to assess many potential wetland restoration sites is considered to be a sub-basin-scale of assessment. Ideally, once a river basin characterization has identified sub-basins to be targeted for sub-basin-scale assessment, wetland restoration work would focus on those targeted sub-basins rather than the entire river basin. Landscape-scale assessment of potential wetland restoration sites facilitates the identification and comparison of many potential sites, while establishing the context for how wetland restoration can be used to restore key ecological processes.

Landscape-scale assessment should include multiple temporal scales. As early as 1978, Wolman and Gerson (1978) noted that humans have altered many of the natural processes that control the form and development of landscapes, watersheds, and wetlands. To assess the comparative potential for process alteration, river basin characterization started with the creation of a pre-development land cover coverage using General Land Office surveyor data of vegetation from the early 1870s. The project plan was to compare this pre-development coverage with a current land use/land cover coverage. While this is an essential assessment step, it became apparent that, with the population growth projections for Puget Sound, a future land use/land cover coverage was needed to assess the effects of future development compared to current conditions. Based on long-term planning documents required of local jurisdictions under the state's Growth Management Act, a future build-out land use/land cover coverage was developed. Using pre-development, current, and future build-out land use/land cover coverages, individual team members assessed the comparative risk of process alteration by sub-basin from pre-development to current conditions and from current to future build-out conditions. This assessment at multiple temporal scales has been shown to be a powerful tool in the development of an overarching recovery framework for a river basin.

Look "big picture" first and then focus down. Planning at the landscape-level is the only way we are going to avoid undesirable, if not unacceptable, landscape dysfunction (Franklin 1993).

Because biologists are often trained to do site-specific work, we tend to collect and analyze data at that scale. Assessment at multiple scales is, in many ways, like running river gravel through a series of sieves. The coarsest sieve (river basin characterization) allows you to assess the largest pieces of aggregate. This in turn establishes the context for evaluating medium-sized rocks captured in the moderate sieve. And finally, the combined knowledge gained from the coarser sieves allows improved comparison of the finest grains. By collecting data at the site- or reach-scale first, you are only capable of looking at the finest grains without benefit of its parent material.

This site- or reach-scale approach also restricts the ability of resource managers to identify core problems through an understanding of cause-and-effect relationships. For example, a perched culvert (the cause) results in a fish passage barrier (the effect). Replace the culvert and the true cause of the problem is addressed. Rarely are cause-and-effect relationships that straight-forward. A more likely scenario occurs when biologists indicate that riverbed scour is a potential limiting factor for chinook on a river reach. The task of identifying limiting factors is monumental in its own right, but when the only available choice for resource managers is to correct alterations on the reach where scour is occurring, success becomes unlikely. Scour is not the core problem, but the effect of one or more ecological process changes that occurred upstream of the site. Scour may be the limiting factor for chinook production in this particular reach of river, but it is the symptom of a human-induced change in how the watershed delivers and routes water, sediment, and wood. Unless we begin to focus on the core problems, recovery efforts will not be successful.

Looking “big picture” means looking at ecosystem health rather than the health of individual parts (Franklin 1993). It also provides an opportunity to understand and assess the cumulative function of all wetlands in a river basin which may be different than the additive function of the individual wetlands themselves (Johnston et al. 1990). Norton (1992) suggests five axioms of ecological management that create a framework for the assessment of ecosystem health by looking at ecosystem processes. Gaining an understanding of ecosystem health and where and how the system is compromised establishes the foundation for an overarching recovery framework that targets core problems.

Selection of the highest priority wetland restoration sites requires a landscape perspective. Past efforts have focused on assessing functions and values of individual wetlands. This requires a detailed site-specific assessment of physical, biological, and chemical attributes. The issue is one of efficiency. In the Nooksack Basin, nearly 5,400 sites were evaluated, of which 3,513 (or 65%) were determined to have restoration potential. The cost and time required to do site-specific function assessment on all 5,400 sites is unrealistic and unnecessary, if a coarse-sieve characterization approach is taken. While assessing functions at a site-scale is ultimately necessary for the highest priority sites, a landscape-scale assessment is needed to efficiently identify those high priority sites from the hundreds, even thousands, of potential wetland restoration sites in a river basin.

Wetlands in an Ecosystem Context

An assessment of function is only one component of a more comprehensive wetland assessment. Wetland ecologists need to move beyond function assessment in their comparative evaluation of wetlands. Wetland functions are the physical, biological, and chemical processes or attributes of a site (Adamus et al. 1987; Hruby et al. 1999). Ecological processes are the physical agents of landscape pattern formation and maintenance that create and maintain the physical, chemical, and biological attributes of a site (Gersib et al. 1999). Site-specific function assessment is an essential tool to wetland ecologists at a site-scale. However, without a landscape context provided by an assessment of the ecological processes, function assessment can be evaluating the symptoms of core ecological problems that can exist many miles from the assessment site. This requires that wetland function assessment be nested in a larger landscape context.

Through implementation of the wetland restoration program, we have begun to realize that landscape context is essential to function characterization. In Western Washington, a vast majority of precipitation moves as subsurface flow in a native coniferous forest landcover. Surficial geology, topography, and land cover are important factors that dictate the amount, extent, and retention of sub-surface water. Large wetlands located on recessional outwash in the Nooksack Basin lowland have developed in remnant meltwater channels carved in the outwash plain as continental ice sheets receded. These outwash plains are deep sands and gravels that support the basin's largest surficial aquifer. This shallow aquifer discharges at topographic breaks providing the dominant water source for wetlands in this area. Immediately adjacent to the outwash plains are glacial marine deposits that have a near-impervious "hardpan" layer approximately 32 inches below the soil surface. In these deposits, precipitation percolates quickly down to the hard pan layer and then moves laterally to a topographic break where it discharges as a spring/seep or to a geologic break where it moves downward as groundwater. Wetlands occurring on glacial marine deposits are much smaller in size and receive groundwater discharge only during times of prolonged precipitation. This has resulted in wetlands on the outwash plain developing peat soils, while wetlands on glacial marine deposits maintain mineral soils. Understanding this water movement, both above and below ground, as well as the effects of human land use on that movement, help establish the landscape context needed to assess the functions that a wetland provides.

Functions are the product of ecological processes. It is an issue of scale. Assessing wetland functions at a site-level limits a majority of the assessment to alterations at that scale. However, a wetland's opportunity and effectiveness at performing a function are dependent not only on site-specific structural features but landscape-scale processes as well. For example, a drainage ditch placed through a wetland is a site-specific structural modification that reduces water permanence and the site's effectiveness at providing summer rearing for juvenile coho salmon. An example of a landscape-scale alteration is one where effective impervious surface and surface/sub-surface water withdrawals reduce summer low flows to the point where they restrict access to the site by juvenile coho salmon. In both cases, the wetland's effectiveness at providing habitat for juvenile coho salmon was lost, one resulting from a site-specific structural alteration, the other from the cumulative alteration of landscape-scale processes many miles from the wetland site. Function assessment must move beyond simply comparing how one wetland functions in relation to another. Assessment should also be capable of answering the following questions: Is the wetland maximizing it's potential to perform each function? What are the core problems that are restricting the site from maximizing function performance?

Wetlands must be understood and protected as part of the larger landscape and not as a separate entity. Our goal should be to protect the autonomous, self-integrative processes of nature (Haskell et al. 1992) rather than a select few parts. To do this, we must first establish the landscape context for wetland resources, the processes that influence them, how they influence processes, and how human land use affects each. This context is needed to understand the role that wetlands provide. Human activities must be understood in the larger context of self-organizing systems (Haskell et al. 1992). Norton (1992) suggests that we choose between protecting features that are familiar to our culture, ecological features that support certain essential "services", or long-standing features that provide the geological context for ecological processes. Concepts learned through river basin characterization support the latter.

Wetlands in Ecosystem Management

Focus recovery efforts on preserving and restoring ecological processes. Understanding the processes responsible for shaping Puget Sound river basins and maintaining their biodiversity is fundamental to successful ecological management. Past recovery efforts have often focused on site-specific structure-based fixes to environmental problems. For example, increased stormwater runoff in urbanizing areas and development in the floodplain have resulted in increased flood damage along many Puget Sound rivers. Structure-based fixes such as higher dikes, straightened channels, and excavated floodways all treat symptoms, rather than the core problems. Until management focuses on the human-induced changes to the delivery and routing of water, recovery efforts will be elusive and short-lived. However, these short-term successes have led to the belief that, in principle, all processes can be submitted to human management by means of science and technology (Faber et al. 1992). We are learning that this is not a valid assumption.

Further, there is a growing body of evidence that structural "fixes" are rarely self-maintaining and serve, at best, as a short term fix to a long-term problem (Beschta et al. 1994; Frissell and Nawa 1992; Elmore and Beschta 1989; Beschta et al. 1991). Ehrenfeld (2000) suggests that when inputs of physical energy, in the form of water or wind movement, are dominating forces in structuring an ecosystem, then ecosystem processes should be the primary focus in developing restoration plans. In Puget Sound, water is the dominant force in structuring the ecosystem of this region. Alterations to the natural delivery and routing of water have contributed substantially to many of our natural resource problems. Only through the restoration of ecological processes will we begin to address these problems in the long-term.

One suggested approach adapted from the fluvial restoration objectives of the National Research Council (1992) includes the following objectives for landscape-scale ecosystem restoration.

1. Restore altered ecological processes.
2. Restore natural landscape form, only if the restoration of ecological processes alone does not.
3. Restore natural plant communities, only if the restoration of ecological processes and natural landscape form does not.
4. Restore native plants and animals, only after steps 1-3 are completed and they do not recolonize on their own.

In the Pacific Northwest, wetlands are a keystone natural resource that has an important role in how a river basin delivers and routes water, sediment, nutrients, large wood, and heat. By only assessing the functions of individual wetlands at a site-scale, little consideration is given to their cumulative contribution to ecosystem health at landscape-scales.

Landscape-scale recovery of ecological processes minimizes potential for conflict associated with single-species management. It is well documented that management actions developed at only the site-scale and focused on only one species have the potential to adversely affect other species (Jackson et al. 1995; Frissell et al. 1997; Ehrenfeld 2000). Meffe (1992) described attempts to restore individual salmon fisheries in the Pacific Northwest as "techno-arrogance," while Franklin (1993) noted that trying to conserve ecosystem diversity on a species-by-species basis is going to exhaust our patience, pocketbooks, and the time and knowledge available. Knowing this provides added pressure on natural resource managers when called upon to develop recovery plans for ESA listed species or management plans for economically important game species.

We are learning that the solution lies in our ability to focus on the natural habitat-forming processes rather than the habitat of a species. In the final rule governing the take of 14 threatened salmon and steelhead species in the Pacific Northwest, the National Marine Fisheries Service (2000) defines properly functioning condition as the sustained presence of natural habitat-forming processes that are necessary for the long-term survival of salmonids throughout the full range of environmental variability. By targeting the restoration of ecological processes that create and maintain habitat for all native species, the focus is on ecosystem health rather than individual species, reducing or completely eliminating the need for value judgements that place higher priority on one species over others.

Habitat management vs. the management of habitat forming processes is clearly an issue of scale in both a spatial and temporal sense. Fast-changing human cultures are interacting with larger-scale, slow-changing ecosystems. There is a strong need to develop policies that allow human cultures to thrive without changing the life support functions, diversity, and complexity of ecological systems (Haskell et al. 1992). I suggest that this need can best be accomplished through the management of ecological processes.

An overarching recovery framework at the river basin-scale is an essential planning tool for integrating disparate natural resource management programs and initiatives. Through our work in the Snohomish Basin, it is became apparent that the development of a technically sound recovery framework allows for both focused planning for salmon recovery, water quality, baseflow, and peak flow improvements and the integration of each into a multi-faceted recovery framework. In the Pacific Northwest, natural resource managers are expressing frustration with the myriad of planning efforts that are underway. This planning and implementation is being done by different people, for different purposes, at different scales, and with different timelines. Without an overarching landscape-scale recovery framework, these disparate planning efforts will remain uncoordinated. Work in the Snohomish Basin has demonstrated that the development of an overarching recovery framework for a river basin is possible. The challenge then is to develop and maintain the societal discipline needed to work in a coordinated fashion within the framework and maintain the recovery trajectory established.

Conclusions

Despite dramatic increases in effort, strong mandates, and massive expenditures for environmental protection over the past 20 years, the overall condition of natural ecosystems continues to decline (Karr 1995; Montgomery et al. 1995). A growing body of evidence indicates that declines in ecosystem integrity are perpetuated by existing policies and traditional techniques that treat local symptoms of habitat damage and fail to address the root biological and physical causes of ecosystem degradation and population decline (Angermeier and Schlosser 1995; Montgomery et al. 1995; Reeves et al. 1995; Ebersole el al. 1997).

For these reasons, natural resource management should begin to move away from a site-specific structure-based paradigm for natural resource recovery and toward a more ecologically-based, landscape-scale, process-based approach. Three caveats are important. First, while resource managers are beginning to acknowledge that the existing structure-based paradigm is not working, much of the published literature on this new process-based paradigm is conceptual in nature and highly experimental in practice. Decades, rather than years, will be needed to evaluate its effectiveness. Second, while restoration will be the key driver for any natural resource recovery efforts, the preservation of intact functioning processes and systems should be the cornerstone of any resource recovery plan. Finally, while our understanding of basic linkages within natural systems and the effects of human land use on those linkages is still quite poor, it should not stop managers from merging what is known with professional judgement to advance our understanding of natural systems. The wetland restoration program and the river basin characterization project seek to advance our understanding of resource assessment within the Puget Sound region. Efforts, like these, will require continued development and evaluation, while new initiatives need to build on existing landscape-scale, process-based assessments. If natural resource management truly implies movement toward desired end results, this transition in recovery paradigms must continue.

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Watershed Wetlands Restoration Planning:
The Massachusetts Experience

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Abstract

Since the inception of its statewide program for proactive (non-regulatory) wetland restoration in 1994, Massachusetts has pioneered watershed-based wetlands restoration planning and project implementation. Under this innovative program, potential wetland restoration sites are initially located through aerial photointerpretation. Watershed context is provided through evaluation of existing information to identify landscape-level functional deficits pertaining to water quality, water quantity (flood storage and stream baseflow), and fish and wildlife habitat. Sites are evaluated and prioritized according to their capacity to help improve the watershed pursuant to community restoration goals. An extensive public participation process ensures community support. Project implementation relies on numerous project sponsors who receive assistance through the state program from a variety of public and private funding and technical assistance mechanisms. Pre- and post-construction monitoring is conducted to ensure project goals are met and to help improve understanding and application of restoration techniques.

Introduction

The Massachusetts Wetlands Restoration & Banking Program (WRBP) was established in 1993 to help implement the state's policy of "no net loss of wetlands in the short-term and a net gain in the long-term." While a strong state regulatory program has minimized continuing wetland losses, there is recognition of the need for a program to address historic losses that have exceeded 28 percent since colonial times. WRBP's mission is, in part, to implement a statewide program for voluntary (non-regulatory) wetland restoration. For purposes of this program, wetland restoration is defined as *the act, process, or result of returning a wetland or a former wetland to a close approximation of its condition prior to disturbance.* The program design is based on the premise that wetland restoration can both reclaim functions at individual sites and help address the effects of cumulative wetland loss by restoring functions at the watershed level. With technical support from the US Army Corps of Engineers, New England Division, WRBP developed a method for identifying and evaluating wetland restoration sites within watersheds. (WRBP 1996a) During a pilot study of the Neponset River watershed, WRBP further developed an approach to prioritizing sites based on watershed restoration goals relating to habitat, water quality, and water quantity adopted by the watershed community. The entire process – identifying, characterizing, and prioritizing restoration sites, and engaging watershed stakeholders in setting goals – is part of WRBP's "watershed wetlands restoration planning" process (WRBP 1996b).

Methods

Watershed wetlands restoration plans (WWRPs) are developed using a technical approach to support a community-based planning process. The technical process described here has evolved considerably since its first application in the Neponset River watershed. Since this is a new process, none of the WWRPs have been completed at this time.

Technical Process

There are three steps in the technical process: a) site identification, b) site characterization, and c) site prioritization.

<u>Site identification</u>. Potential restoration sites initially are identified through aerial photointerpretation techniques. National Wetlands Inventory maps are updated as needed to provide a current base map of wetlands for the study area. Updated maps are digitized and wetlands examined for characteristics to indicate their potential as restoration sites. Individual sites form two new digital data sets: 1) wetland areas with internal structural changes (polygon data) and 2) wetland edges abutting external land uses that probably are causing degradation (arcs data).

<u>Site characterization</u>. Potential restoration sites are classified as type 1 (former wetlands) or type 2 (existing but degraded wetlands). For restoration sites one acre or greater in size, information describing the site and the surrounding landscape is generated. A data table is created including the following site-specific information: 1) existing type of wetland (Cowardin et al. 1979), 2) type of wetland to be restored (Cowardin et al. 1979), 3) site size, 4) adjacent land uses, 5) type of impact to site (e.g., ditching, excavation, or fill), 6) type of ownership (public or private), 7) action needed to restore, 8) range of costs for restoration, and 9) difficulty of restoration (high, medium, low). Additionally, the sites are evaluated by specific parameters to determine their potential to help restore key functions. For example, evaluation parameters for wildlife habitat restoration potential include size, habitat diversity, and connectivity. Parameters to evaluate flood storage potential include upstream proximity to known flood locations, location in the 100-year floodplain, presence of a constricted outlet, sites conducive to sheet flow conditions, sites with dense vegetation, and size. Water quality enhancement potential is determined based on physical factors influencing nutrient removal and transformation as well as sediment and toxicant retention. Such factors include vegetation density, gradient, and soil type. The application of these parameters to potential restoration sites is described in "Site Identification and Evaluation Procedures" (WRBP 1996a). Sites that have been ditched or filled are presumed to have the potential, if restored, to improve stream baseflow and groundwater recharge. Evaluation for this function was not included in the 1996 procedures but was added to the WWRP process during study of the

Neponset River watershed in response to input from the watershed community. The evaluation process for groundwater recharge and stream baseflow potential is described in the "Draft Neponset River Watershed Wetlands Restoration Plan" (WRBP 1998).

Site prioritization. Restoration priorities are driven by restoration goals established through an iterative public planning process described below. Goals are based on an evaluation of "watershed functional deficits" that examines the ways in which watershed-level functions have been compromised by cumulative wetland loss. This analysis is based on existing information about the watershed. In Massachusetts, the availability of information regarding water quality, flooding, habitat, groundwater, and stream flow varies from watershed to watershed. Once problems within the watershed have been evaluated, the deficit analysis is presented to watershed stakeholders and a set of restoration goals is adopted. Restoration sites that have been characterized as positive are identified for each of the goals. These are the potential restoration projects that can have a positive impact on reaching the goals on at least the site level. Another level of analysis is applied to determine which sites have the potential to make a contribution to the watershed beyond the site level. For example, priority sites to improve water quality are considered those that are located within ½ mile downstream of an identified pollutant source. Similarly, priority sites for flood storage improvements beyond the site level are 65 acres or greater in size or are located within ½ mile upstream of an identified flood damage area. Additional priority sites may be those that: 1) represent unique or rare habitat types, 2) address flood storage, pollution attenuation, and habitat, and 3) are located within a recharge area of a public water supply. This approach can be modified to reflect the needs and priorities of individual watersheds.

Planning Process

The planning process for developing WWRPs complements the development of technical information and analysis. Plan development is an iterative process that delivers technical information to watershed stakeholders and helps them use this information to frame an action plan using wetland restoration to address watershed concerns. The planning process is further supported through an aggressive public outreach and education program. The process is described in "Watershed Wetlands Restoration Planning Guidance" (WRBP 1996b).

Steps in the planning process include:

Phase I – Initiation. A WWRP proposal document is generated to notify the public of an agency's or group's intention to develop a WWRP for a specific watershed. Anyone can ask to be put on a mailing list to receive notices of public meetings and of the availability of planning documents. Public education and outreach activities such as news releases, displays, site walks, and distribution of curriculum materials are begun during Phase I and continued throughout the project.

Phase II – Evaluation and Goal Setting. A Preliminary Report is developed presenting information on the condition of the watershed and its wetlands, as well as maps locating and tables describing potential restoration sites. This information is used as a vehicle for developing wetland restoration goals for the watershed through a series of public meeting discussions.

Phase III – Plan Preparation. A Draft Plan is prepared which prioritizes wetland restoration sites based on community restoration goals. Following another round of public notice and meetings, a Final Plan is prepared that includes a detailed implementation strategy for restoring the watershed's priority wetlands.

Phase IV – Implementation. Project sponsors (e.g., any organization, group or landowner) take the lead on individual restoration projects. WRBP stays involved to assist in implementation at individual restoration sites through the GROWetlands (Groups Restoring Our Wetlands) Initiative. GROWetlands projects are community-based initiatives that may receive funding, technical, and other support from WRBP and its many partners including federal and state agencies and corporate donors. GROWetlands projects will be included in WRBP's wetland restoration site monitoring program which is planned for initiation in the fall of 2000. The program will be tailored to the specific goals of each project. GROWetlands projects may also host scientific research and education programs.

Results

To date, a watershed wetlands restoration plan has been completed for the Neponset River watershed and WWRPs are in various stages of development for ten additional watersheds (Paskamanset, Otter, Upper Ipswich, Mill/Manhan, Upper Blackstone, Shawsheen, Concord, Charles, Ten Mile, and Narraganset Bay/Mt. Hope Bay Shores) (see Figure 1 at end of paper). WRBP hopes to complete WWRPs for the state by 2010.

Completed in January 2000, the Neponset WWRP identified a total of 171 potential restoration sites in this 13-town drainage area. (see Figure 2 at end of paper). Following issuance of a Preliminary Report and an extensive public dialogue, restoration goals were adopted for the Neponset River watershed as follows.

- Improve water quality.
- Restore salt marshes.
- Improve wildlife habitat.
- Improve flood storage.
- Address problems related to invasive species.
- Improve cold water fisheries habitat.
- Improve groundwater recharge and stream baseflow.

A Draft Plan was developed presenting an evaluation of 171 potential restoration sites relative to the restoration goals adopted for the watershed. Table 1 shows the number of sites identified that support each of the restoration goals.

Table 1. Number of sites that support Neponset River Watershed wetland restoration goals. Note that many sites address more than one goal.

Restoration Goal	No. Sites
Improve Water Quality	68
Restore Salt Marshes	16
Improve Wildlife Habitat	76
Improve Flood Storage	84
Address Problems Related to Invasive Species	38+ (Data incomplete)
Improve Cold Water Fisheries Habitat	5
Improve Groundwater Recharge and Stream Baseflow	69

A Final Plan was prepared which further evaluated the importance of potential restoration sites to the watershed. Out of 171 sites, 65 sites were identified as priority sites that could help address watershed-level goals. Priority categories were:

- High Functional Value Sites. These sites have the potential to improve the watershed overall for at least three parameters: water quality, flood storage, and fish and wildlife habitat. Please note that a number of these sites also are significant for improvement of groundwater recharge and stream baseflow.

- Additional Significant Groundwater Recharge and Stream Baseflow Sites. These are other sites that were significant for groundwater recharge and stream baseflow because they are 65 acres or greater in size and/or because they overlay an approved Zone 2 recharge area or Interim Wellhead Protection Area.

- Salt Marsh Restoration Sites. All salt marsh restoration sites were considered priority sites due to historic losses and their ecological significance, especially for estuarine and marine fishes.

- Cold Water Fisheries Sites. Restoration sites that may benefit cold water fisheries were considered priority sites.

In order to ensure public support for an ongoing restoration program, an extensive public outreach program was implemented during the planning process. Education and outreach activities included public meetings, a display that circulated among area libraries, distribution of curriculum materials to area teachers, newsletter articles, and participation in Neponset River celebrations and other community activities.

The Final Plan sets forth the following Action Agenda for WRBP to support its implementation.

- Restore 130 acres of Neponset River watershed wetlands by 2010.
- Promote wetland restoration at priority wetland restoration sites.
- Promote wetland restoration within ecologically significant areas.
- Provide technical support, assistance in obtaining project funding, and other help to project sponsors through the GROWetlands Initiative.
- Promote the use of wetland restoration sites for education and research.

As a direct result of the WWRP process, a dozen projects in the Neponset have been nominated for implementation under the GROWetlands (Groups Restoring Our Wetlands) program. Following are summaries of example projects:

Neponset Marshes. This project involves restoration of 20 or more acres of state-owned salt marsh degraded by dredge spoil disposal. The project is being funded with a grant from the U.S. Fish and Wildlife Service, state funds, a corporate monetary donation, and corporate donation of in-kind services from an environmental consulting firm. Restoration work will begin in the fall of 2000 (see Figure 3 at end of paper).

Pope John Paul II Park. Restoration of two acres of salt marsh in the Neponset estuary was completed in 1999 in conjunction with a landfill closure and state park construction project (Figure 3).

Gulliver's Creek. This 75-acre state-owned salt marsh is being invaded by common reed (*Phragmites australis*). Through a corporate donation of in-kind services, an environmental consulting company will evaluate the condition of the marsh and develop restoration options (Figure 3).

Turners Pond. WRBP is conducting its first release of 10,000 *Galerucella* sp. beetles in an attempt to control purple loosestrife (*Lythrum salicaria)* in cooperation with the Town of Walpole, Massachusetts. The project is being funded, in part, by a corporate monetary donation (see Figure 4 at end of paper).

Billings Creek. This 20-acre salt marsh in the City of Quincy is tidally restricted and has been subject to extensive fill in the past for a naval airbase. The Army Corps is cooperating with WRBP and the City to study restoration options (see Figure 5 at end of paper).

Walpole White Cedar Swamp. White cedar swamp is a habitat type of special concern to the state's Natural Heritage Program. This 250-acre site has been adversely impacted by adjacent land uses. A study financed by state watershed funds will explore the impacts of stormwater runoff from adjacent residential subdivisions and impoundment of the swamp by a railroad embankment, and will develop restoration options (see Figure 6 at end of paper).

Conclusions and Recommendations

While more experience is needed to refine the WWRP technical approach, the first application in the Neponset River watershed has had the desired result of stimulating restoration activity with strong public support. The current availability of public and private funds and the donation of in-kind services have facilitated restoration project development and implementation. The identification and evaluation of priority restoration sites within a watershed generates confidence that financial contributions are reaching projects of the highest ecological significance and of most concern to the watershed communities.

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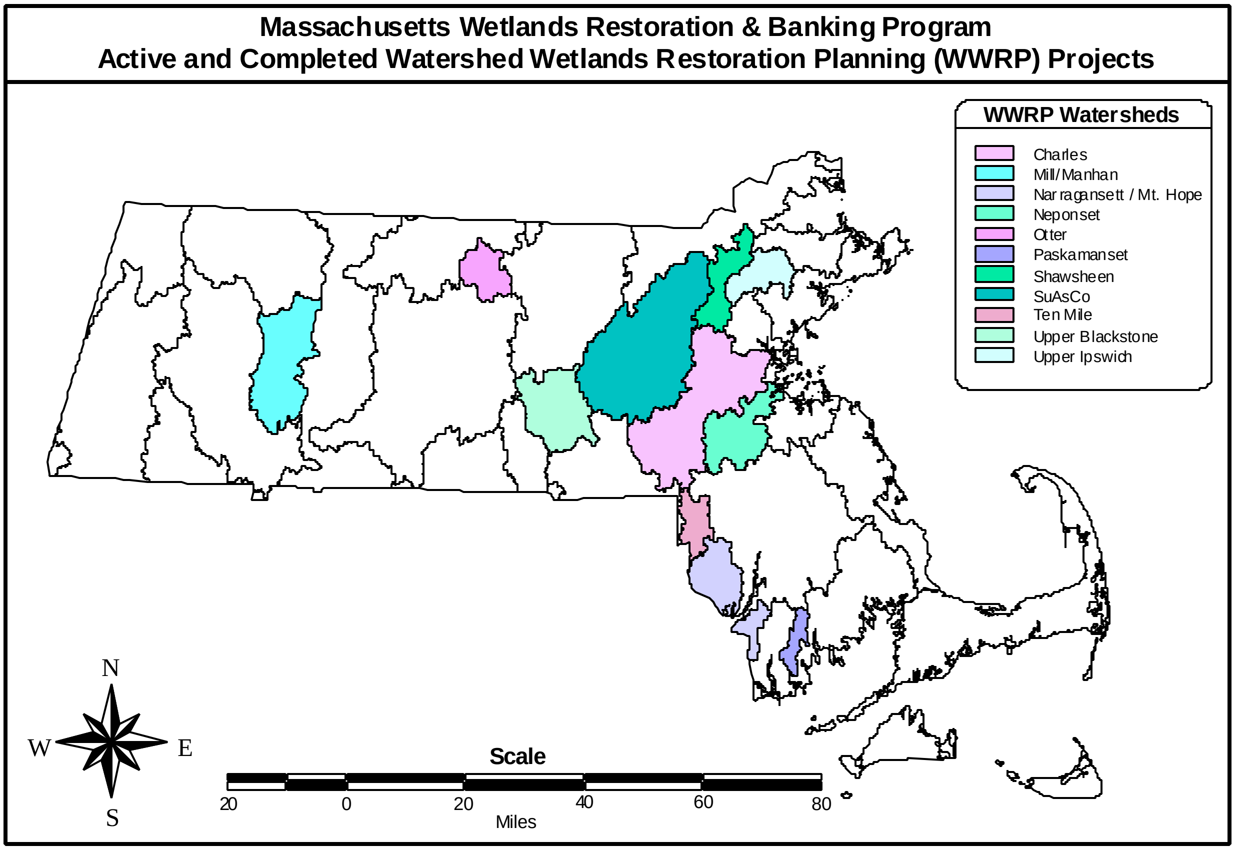

Figure 1. Watersheds where wetland restoration planning is active or completed.

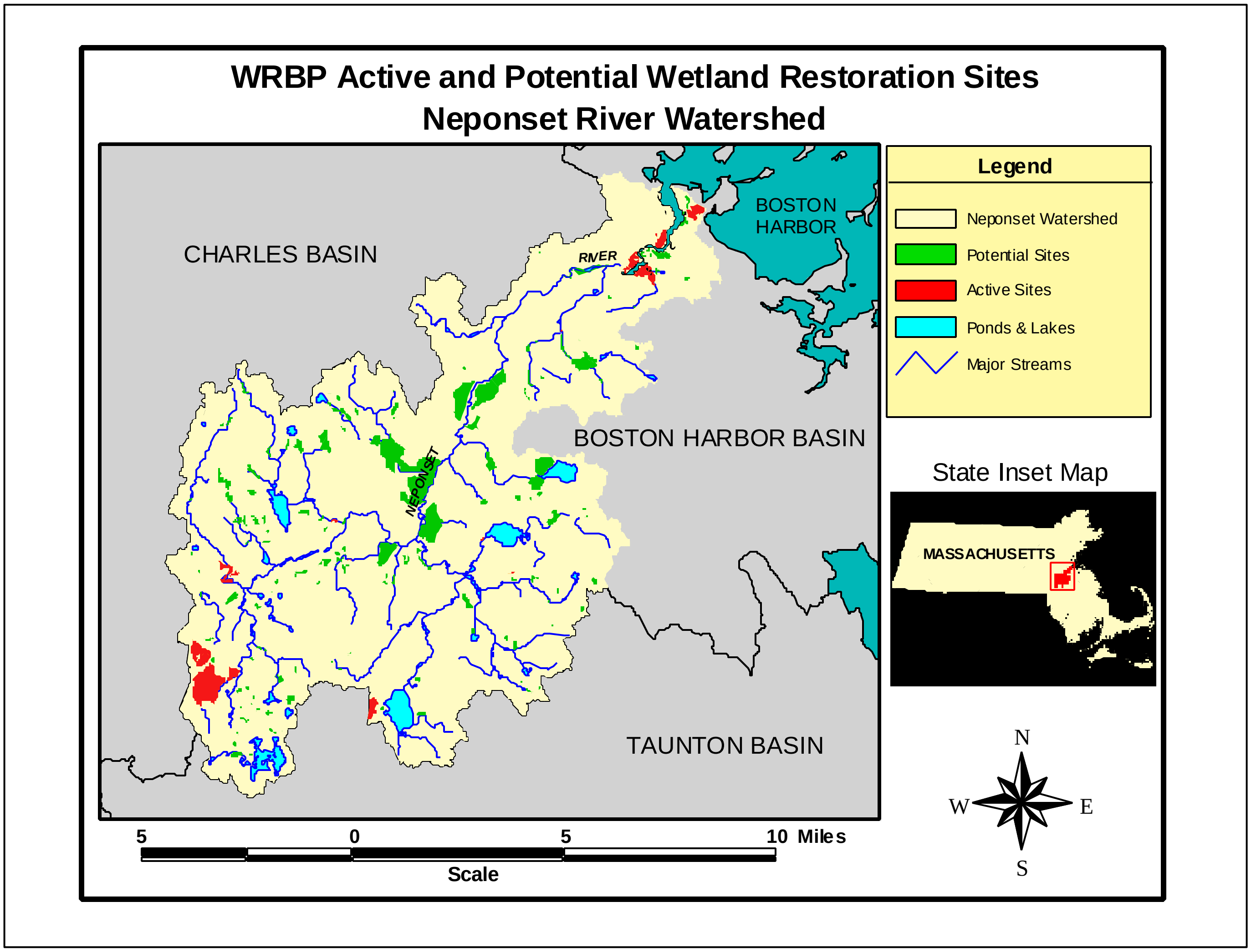

Figure 2. Neponset River watershed and location of potential wetland restoration sites.

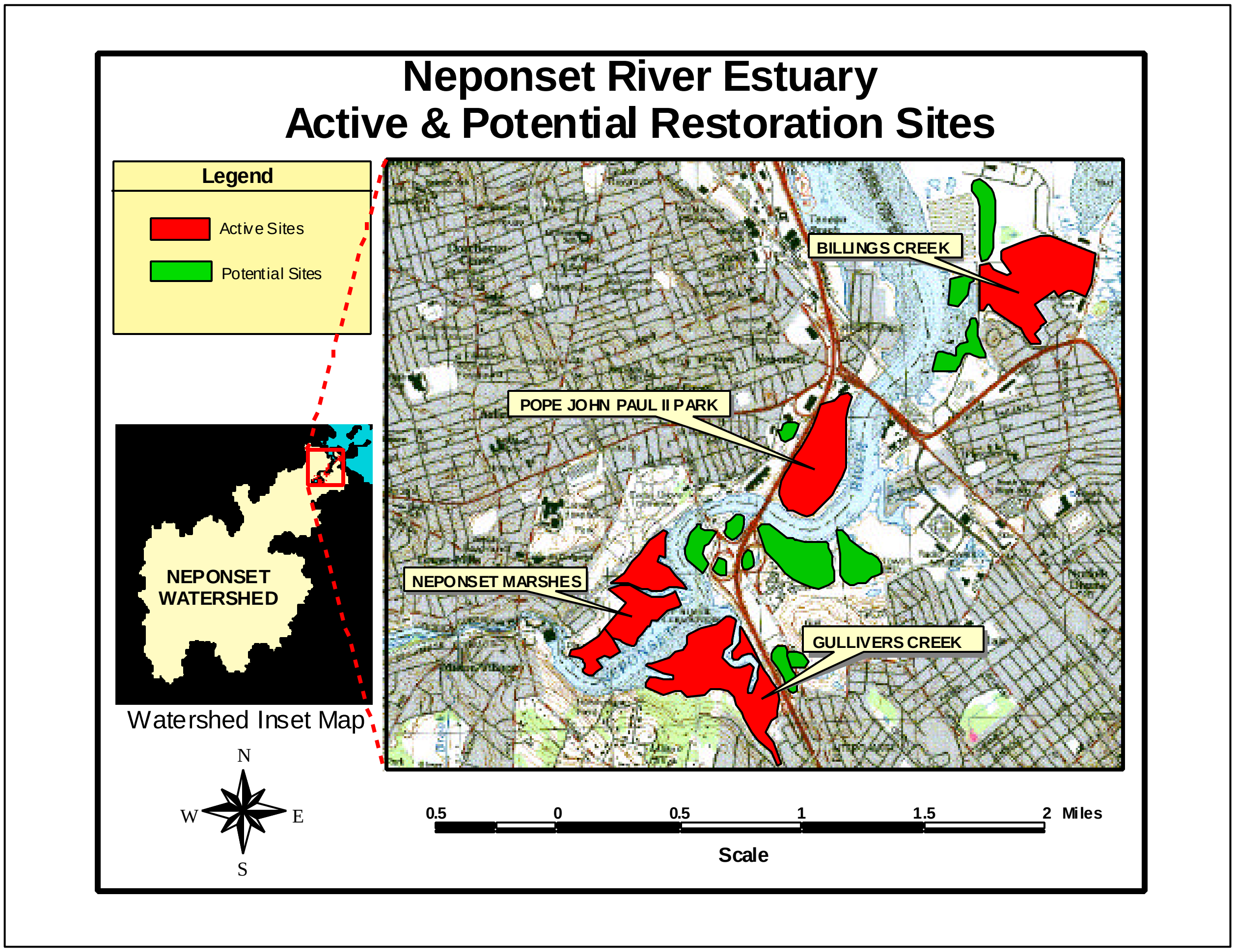

Figure 3. Close-up view of some restoration sites in lower part of watershed.

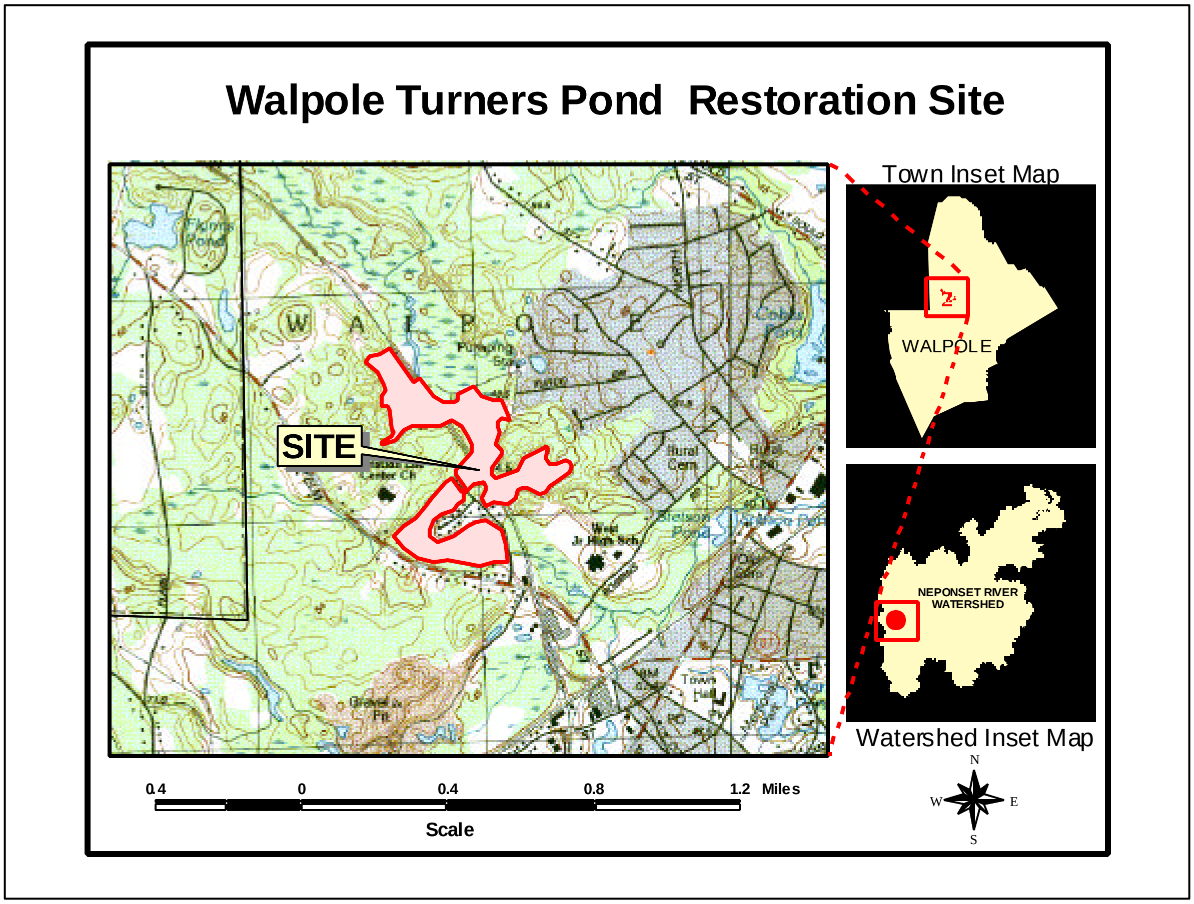

Figure 4. Turners Pond restoration site.

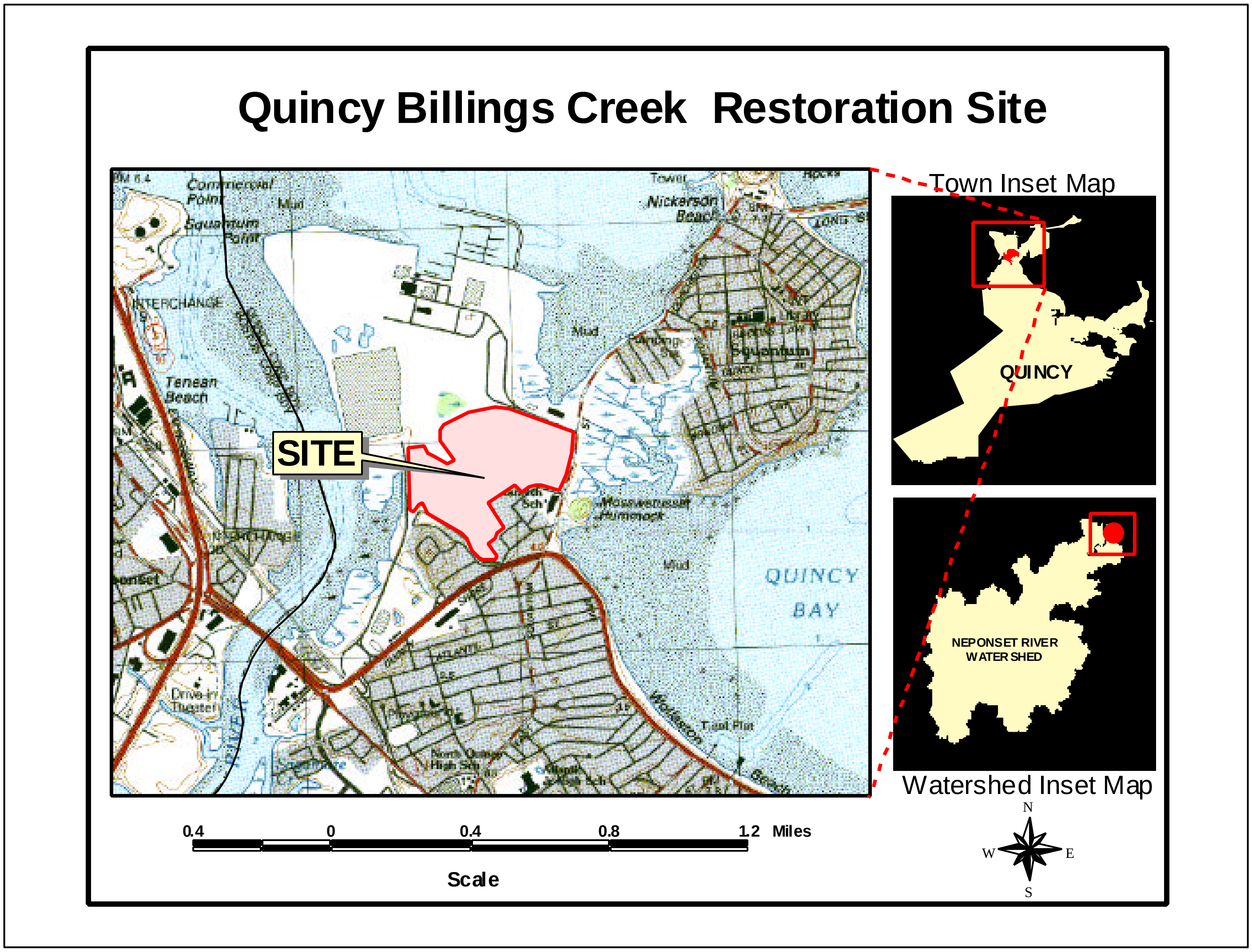

Figure 5. Billings Creek wetland restoration site.

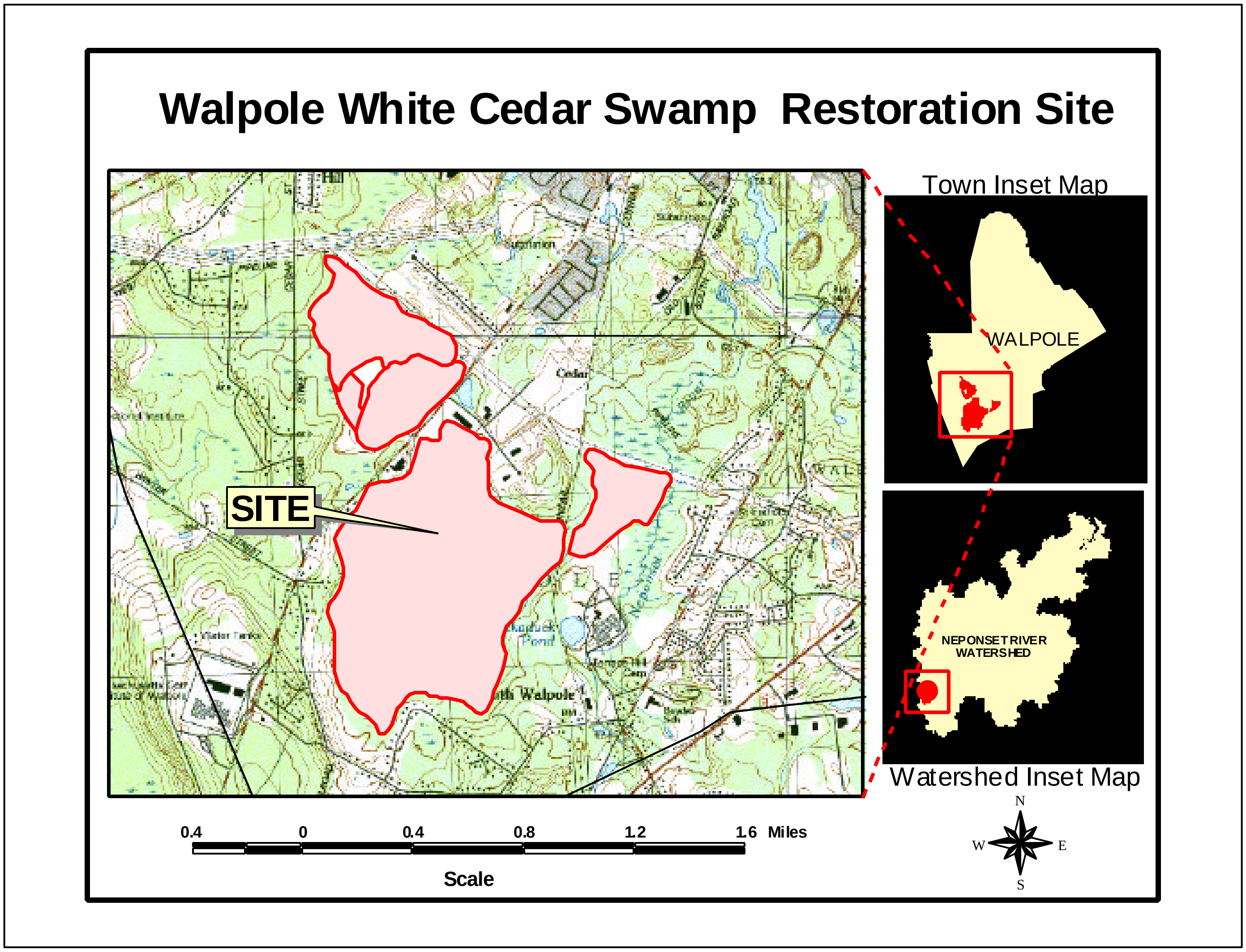

Figure 6. White Cedar Swamp restoration site.

Remotely-sensed Natural Habitat Integrity Indices for Assessing the General Ecological Condition of Watersheds

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Abstract

Using geographic information system technology, the U.S. Fish and Wildlife Service has developed a set of indices to help describe the condition of natural habitats in watersheds. Indices are derived primarily from data collected through remote sensing to allow for large geographic area coverage and periodic updating. Eleven remotely-sensed natural habitat indices have been created to evaluate and report on the status of wetlands, streams, and other natural habitats. Seven habitat extent indices address the current distribution of natural habitat within the watershed, along stream and river corridors, and around wetlands, ponds, and lakes. Three disturbance indices relate to streams and wetlands. The habitat extent and disturbance indices can be aggregated to generate a composite index for the watershed. The data collected in this type of analysis allow for calculating numeric indices as well as for producing maps showing the distribution of natural habitat and vegetation throughout the watershed. These indices are useful for reporting on the current status of natural habitat in the watershed, for monitoring trends in these resources, and for informing the public on the status and fate of natural resources in watersheds. They are one of several indicators of ecological condition that can be evaluated for these purposes.

Introduction

Traditionally, natural resource management in the United States has emphasized individual species or guilds (e.g., waterfowl, furbearers, and endangered species), particular habitats (e.g., forests, waterfowl habitat, farmland, and wetlands), or Aprotected@ lands (e.g., national and state forests, parks, and wildlife refuges or wildlife management areas and private wildlife sanctuaries). More recently, there has been growing interest in pursuing a watershed-based approach to environmental planning, management, and restoration (e.g., Naiman 1992; Williams et al. 1997). Wetlands are often the vital link between land and water resources, while rivers and streams connect many different ecological communities.

The widespread availability and use of geographic information system (GIS) technology and the existence of digital geospatial datasets have made it possible to analyze the status and trends of natural habitat and other resources for large geographic areas, including major watersheds. For more than 20 years, the U.S. Fish and Wildlife Service through its National Wetlands Inventory (NWI) Project, has been a leader in producing geospatial data on wetlands and waterbodies, while other agencies, especially state natural resource agencies, have generated digital geospatial information on land use and land cover on a periodic basis.

The NWI=s experience with geospatial databases and remote sensing techniques and interest in fish and wildlife habitat conservation led to the development of GIS tools to aid resource managers and planners in developing strategies to improve the status of natural ecosystems. Since 1995, the NWI Program has been seeking ways to use geospatial data to analyze and represent wetland functions at the watershed-scale. In particular, a new product - the watershed-based wetland characterization report - has been developed and prepared for several watersheds (see Tiner 2002). This type of report includes descriptions of major wetland types and a preliminary assessment of wetland functions (identifying wetlands potentially significant for performing varied functions) for a given watershed. While this information is useful by itself, activities occurring beyond the wetland edge often have a tremendous impact on the quality or health of the wetland. The significance of outside influences on wetlands and aquatic habitats induced a desire to examine and describe the condition of natural habitat beyond wetlands. Given the availability of other geospatial data (especially for land use and land cover), we felt that it might be relatively simple to gather and assimilate geospatial information on other landscape-level properties sufficient to allow for periodic reporting on the overall condition of natural habitats for watersheds or other large geographic regions. This would be valuable information to resource managers and for informing the public on the changing status of natural landscape. It might also serve as a measure that could be frequently reported in state-of-the-environment reports published by various state agencies and possibly for a national report of this kind.

At the watershed-scale, there are many important features determining the overall health of the natural ecosystems. Out of the rather long list of features, several that could be evaluated through remote sensing were identified: 1) extent of natural habitat, 2) condition of stream corridors, 3) extent of wetlands, 4) condition of wetland and other waterbody buffers, 5) extent of waterbodies, 6) extent of altered wetlands, 7) dammed stream length, and 8) channelized stream length. A series of Anatural habitat integrity indices@ were created to develop a simple numeric index reflecting the condition of these key watershed features. Other photointerpretable features that may be of interest include inventories of potential wetland restoration sites and the extent of ditching. These features are not expressed as indices, but instead may be depicted on maps and conveyed in acreage summaries.

The purpose of this paper is to describe these indices and provide an example of their application for a watershed. The sample watershed is Nanticoke River watershed on the Atlantic coastal plain in eastern Maryland.

Remotely-sensed Natural Habitat Integrity Indices

There are many ways to assess land cover changes and habitat disturbances. The health and ecological condition of a watershed may be assessed by considering such features as the integrity of the lotic (streamside) wetlands and riparian forests (upland forests along streams), the percent of land uses that may adversely affect water quality in the watershed (% urban, % agriculture, % mining, etc.), the actual water quality, the percent of forest in the watershed, and the number of dams on streams, for example. Recent work on assessing the condition of watersheds has been done in the Pacific Northwest to address concerns for salmon (Wissmar et al. 1994; Naiman et al. 1992). A Wisconsin study by Wang et al. (1997) found that instream habitat quality declined significantly when agricultural land use in a watershed exceeded 50 percent, whereas when 10-20 percent of the watershed was urbanized, severe degradation occurred.

To help assess the overall ecological condition of watersheds, the Northeast Region of the U.S. Fish and Wildlife Service developed a set of Aremotely-sensed natural habitat integrity indices.@ The variables for these indices are derived through air photointerpretation and/or satellite image processing coupled with knowledge of the historical extent of wetlands and open waterbodies. They are coarse-filter variables for assessing the overall ecological condition of watersheds. The Anatural habitat integrity indices@ do not supplant the need for other environmental assessments. They do, however, provide a GIS-based assessment tool that can be used for developing a broad perspective of the extent and condition of natural habitat for a watershed. For fine-filter assessments, site-specific techniques for determining the ecological integrity of aquatic habitats such as indices of biological integrity (IBI) for stream macroinvertebrates and fishes (Karr et al. 1986; Karr 1991; Angermeier and Karr 1994; Lyons et al. 1996) and procedures for evaluating wetland functions by establishing and examining reference wetlands (see Brooks et al. 2002) may be employed. The natural habitat integrity indices can be used to develop Ahabitat condition profiles@ for individual watersheds at varying scales (i.e., subbasins to major watersheds). Indices can be used for comparative analysis of subbasins within watersheds and to compare one watershed with another. They may also serve as one set of statistics for reporting on the state-of-the-environment by government agencies and environmental organizations.

The indices are cost-effective, rapid-assessment measures that allow for frequent updating (e.g., every 5-10 years). They may be used to assess and monitor the amount of Anatural habitat@ compared to the amount of disturbed aquatic habitat (e.g., channelized streams, partly drained wetlands, and impounded wetlands) or developed habitat (e.g., cropland, pasture, mined land, suburban development, and urbanized land). The index variables include features important to natural resource managers attempting to lessen the impact of human development on the environment. The indices may also be compared with other environmental quality metrics such as indices of biological integrity for fish and/or macroinvertebrates or water quality parameters. If significant correlations can be found, they may aid in projecting a Acarrying capacity@ or threshold for development for individual subbasins.

To date, a total of 11 indices have been developed. Each of them, in one way or another, represents habitat condition in a watershed. Seven indices - habitat extent indices - address natural habitat extent (i.e., the amount of natural habitat occurring in the watershed and along wetlands and waterbodies): natural cover, stream corridor integrity, river corridor integrity, wetland buffer integrity, pond and lake buffer integrity, wetland extent, and standing waterbody extent. Three indices emphasize human-induced alterations to streams and wetlands. These stream and wetland disturbance indices deal with damming and channelization of streams and wetland alteration. The 10 specific indices may be combined into a single index called the composite natural habitat integrity index for the watershed. All indices have a maximum value of 1.0 and a minimum value of zero. For the habitat extent indices, the higher the value, the more habitat available. For the disturbance indices, the higher the value, the more disturbance. For the composite natural habitat integrity index, all indices are weighted, with the disturbance indices subtracted from the habitat extent indices to yield an overall Anatural habitat integrity@ score for the watershed.

Presently, the indices do not include certain qualitative information on the condition of the existing habitats (habitat quality) as reflected by the presence, absence, or abundance of invasive species or by fragmentation of forests, for example. It may be possible to add such data in the future. Another consideration would be establishment of minimum size thresholds to determine what constitutes a viable Anatural habitat@ for analysis (e.g., 0.04 hectare/0.1 acre patch of forest or 0.4 hectare/1 acre minimum?). Other indices may also need to be developed to aid in water quality assessments (e.g., index of ditching density for agricultural and silvicultural lands).

ANatural Habitat@ Defined

Use of terms like Anatural habitat@ and Anatural vegetation@ have stirred much debate, yet despite this, we feel that they are useful for discussing some of the effects of human activities on the environment. We use these terms loosely and not in the sense of native or endemic species. Instead, we view them as expressions of areas that support wildlife of forests, vegetated wetlands, shrub thickets, old fields, and sand dunes, for example.

For purposes of this analysis, natural habitats are defined as areas where significant human activity is limited to activities like nature observation, hunting, fishing, and forestry and where vegetation is allowed to grow for many years without irrigation, annual introduction of chemicals (e.g., herbicides and pesticides), or annual mowing or annual harvesting of vegetation or fruits and berries for commercial purposes. Natural habitats may be managed habitats, but they are places where wetland and terrestrial wildlife find food, shelter, and water. They are not developed sites (e.g., impervious surfaces, lawns, turf farms, cropland, pastures, nurseries, orchards, vineyards, mowed hayfields, or mined lands). Commercial forests are included as natural habitat, whereas orchards and vineyards are not. ANatural habitat@ therefore includes habitats ranging from pristine woodlands and wetlands to commercial forests planted with loblolly pine and wetlands now colonized by invasive species (e.g., Phragmites australis or Lythrum salicaria). We recognize that there are differences in habitat quality among areas classified as natural habitat, but these differences are not accounted for. The focus of this coarse-filter analysis is quantitative (i.e., to identify how much wildlife habitat remains - presence or absence) and not to do a qualitative assessment of such habitats. The latter analysis typically requires field investigations (consistent with a fine-filter evaluation). Readers should also note that identifying an area as having Anatural vegetation@ does not imply that substantial groundcover must be present, but simply means that the area reflects the vegetation that is capable of growth and reproduction in accordance with site characteristics (e.g., coastal sand dunes).

Data Sources

Data for these indices are drawn from several sources. Wetland and deepwater habitat data are derived from existing or enhanced NWI digital database. Stream data come mainly from the U.S. Geological Survey=s digital hydro layer based on 1:24,000 mapping, while in some areas, more detailed digital stream data may be available from state or other government agencies. Land use and land cover data may be obtained from several sources: U.S. Geological Survey or state agencies, county or local governments, or be derived by processing current satellite imagery or interpreting recent aerial photography.

Habitat Extent Indices

These indices have been developed to provide some perspective on the amount of natural vegetation remaining in a watershed. The following areas are emphasized: the entire watershed, stream and river corridors, vegetated wetlands and their buffers, and pond and lake buffers. The extent of standing waterbodies is also included to provide information on the amount of open water habitat in the watershed. Each index is briefly described below.

The Natural Cover Index (I_{NC}) represents the percentage of a watershed that is wooded (e.g., upland forests or shrub thickets and forested or scrub-shrub wetlands) and Anatural@ open land (e.g., emergent wetlands or Aold fields;@ but not cropland, hayfields, lawns, turf, or pastures). These areas are lands supporting Anatural vegetation;@ they exclude open water of ponds, rivers, lakes, streams, and coastal bays.

> $\mathbf{I_{NC} = A_{NV}/A_W}$, where A_{NV} (area in natural vegetation) equals the area of the watershed=s land surface in Anatural@ vegetation and A_W is the area of "watershed" excluding open water.

The Stream Corridor Integrity Index (I_{SCI}) reflects the condition of the stream corridors:

> $\mathbf{I_{SCI} = A_{VC}/A_{TC}}$, where A_{VC} (vegetated stream corridor area) is the area of the stream corridor that is colonized by Anatural vegetation@ and A_{TC} (total stream corridor area) is the total area of the stream corridor.

The width of the stream corridor may be varied to suit project goals, but for this index, a 100-meter corridor (50m on each side of the stream) will be evaluated at a minimum, due to its well-recognized role in water quality maintenance and contributions to aquatic habitat quality. If wildlife travel corridors are a primary concern, a larger corridor (e.g., 200m to 1000m) may be examined. The stream corridor may be restricted to Astreams@ (linear tributaries on a 1:24,000 map) or expanded to include Arivers@ (polygonal features at this scale). When rivers are included in the stream corridor integrity index, the index should be called River/Stream Corridor Integrity Index (I_{RSCI}). When the river corridor is analyzed separately, then the index should be called River Corridor Integrity Index (I_{RCI} ; use equation $I_{RCI} = A_{VC}/A_{TC}$ to calculate).

A 100m-wide buffer has been reported to be important for neotropical migrant bird species in the Mid-Atlantic region (Keller et al. 1993) and streamside vegetation providing canopy coverage over streams is important for lowering stream temperatures and moderating daily fluctuations that are vital to providing suitable habitat for certain fish species (e.g., trout). Review of the literature on buffers suggests wider buffers, such as 500m or more for certain species of wildlife (e.g., Kilgo et al. 1998 for southern bottomland hardwood stream corridors). The condition of stream buffers is also significant for locating possible sources of water quality degradation. Wooded corridors should provide the best protection, while developed corridors (e.g., urban or agriculture) should contribute to substantial water quality and aquatic habitat deterioration. For literature reviews of wetland and stream buffers, see Castelle et al. (1994) and Desbonnet et al. (1994).

The Wetland Buffer Index (I_{WB}) is a measure of the condition of wetland buffers within a specified distance (e.g., 100m) of mapped wetlands for the entire watershed:

> $\mathbf{I_{WB} = A_{VB}/A_{TB}}$, where A_{VB} (area of vegetated buffer) is the area of the buffer zone that is in natural vegetation cover and A_{TB} is the total area of the buffer zone (excluding open water).

This buffer is drawn around existing vegetated wetlands. While the buffer zone may include open water, the buffer index will focus on vegetated areas. Note that the buffers of this index were included with the pond and lake buffers in an index called Wetland and Waterbody Buffer Index (I_{WWB}) in earlier analyses; such as the example for the Nanticoke watershed given later in this paper. As mentioned previously, buffer width can be varied according to regional needs and conditions. For our work, the buffer examined will be at least 100m wide.

Semlitsch and Jensen (2001) emphasize that Awetland buffers@ should be better described as Acore habitat@ for semiaquatic species and they urge that such areas be protected and managed as vital habitats. They found that 95 percent of the breeding population of mole salamanders lived in the adjacent forest within 164m of their vernal pool wetland. An interesting article by Finlay and Houlahan (1996) indicates that land use practices around wetlands may be as important to wildlife as the size of the wetland itself. They reported that removing 20 percent of the forest within 1000m of a wetland may have the same effect on species as destroying 50 percent of the wetland.
The Pond and Lake Buffer Index (I_{PLB}) addresses the status of buffers of a specified width around these standing waterbodies:

> $\mathbf{I_{PLB} = A_{VB}/A_{TB}}$, where A_{VB} (area of vegetated buffer) is the area of the buffer zone that is in natural vegetation cover and A_{TB} is the total area of the buffer zone (excluding open water).

See comments under the wetland buffer index above. Ponds are mapped as palustrine unconsolidated bottoms and unconsolidated shores by NWI. Vegetated ponds are mapped as a vegetated wetland type and their buffers are not included in this analysis, but instead are evaluated as wetland buffers.

The Wetland Extent Index (I_{WE}) compares the current extent of vegetated wetlands (excluding open-water wetlands) to the estimated historic extent.

> $\mathbf{I_{WE} = A_{CW}/A_{HW}}$, where A_{CW} is the current area of vegetated wetland in the watershed and A_{HW} is the historic vegetated wetland area in the watershed.

The I_{WE} is an approximation of the extent of the original wetland acreage remaining in the watershed. For example, a watershed with a current coverage of 10 percent wetland would have an I_{WE} of 1.00 where the estimated original extent of wetlands was 10 percent (i.e., no wetlands were lost) or it would have an I_{WE} of 0.50 where 20 percent of the watershed once contained wetlands (i.e., half of the wetlands were lost). When data on historical wetland area are not available, it may be possible to predict this extent. It may be calculated by either evaluating a relatively undisturbed subwatershed in the watershed (i.e., one with similar properties of landscape, soils, and surficial geology) or using the area of hydric soils (and possibly the Amade-land@ area) as the historic extent of vegetated wetlands. Although not the typical case, one should recognize that areal extent of historic hydric soils may be less than the current wetland extent due to level of mapping detail (e.g., scalar issues) or to wetland-creation activities, especially due to beaver influence and shallow pond construction. When this happens, for purposes of this landscape-level assessment, it is assumed that wetland change has not been significant and the I_{WE} is recorded as 1.0.

The Standing Waterbody Extent Index (I_{SWE}) addresses the current extent of standing fresh waterbodies (e.g., lakes, reservoirs, and open-water wetlands - ponds) in a watershed relative to the historic area of such features.

> $\mathbf{I_{SWE} = A_{CSW}/A_{HSW}}$, where A_{CSW} is the current standing waterbody area and A_{HSW} is the historic standing waterbody area in the watershed.

In most cases, watersheds have experienced an increase in standing water due to reservoir, artificial lake, impoundment, and pond construction. Where this is true, the I_{SWE} value is 1.0+ which indicates a gain in this aquatic resource. For this situation, one should use a value of 1.0 when applying this index to determine the composite natural habitat integrity index for the watershed. If one suspects a loss of waterbody habitat, additional calculations are necessary. The historic and present acreages may be created by consulting older USGS topographic maps, comparing them against newer topographic maps (or NWI maps and statistics), and generating numbers showing acreage differences. Readers should note, however, that every wetland trends study that we have conducted over the past 20 years has shown a net increase in open freshwater habitat due to pond construction.

Stream and Wetland Disturbance Indices

A set of three indices have been developed to address alterations to streams and wetlands. For these indices, a value of 1.0 is assigned when all of the streams or existing wetlands have been modified.

The Dammed Stream Flowage Index (I_{DSF}) highlights the direct impact of damming on rivers and streams in a watershed.

> $\mathbf{I_{DSF} = L_{DS}/L_{TS}}$, where L_{DS} is the length of perennial rivers and streams impounded by dams (combined pool length) and L_{TS} is the total length of perennial rivers and streams in the watershed.

It does not attempt to predict the magnitude of downstream effects from such dams. The stream length of the dammed section is determined by drawing a centerline through the impounded polygon. It is, therefore, likely to be a conservative estimate of original stream length which often contains meanders or bends. The total stream length used for this index will be greater than that used in the channelized stream length index, since the latter emphasizes existing streams and excludes dammed segments.

The Channelized Stream Length Index (I_{CSL}) is a measure of the extent of channelization of streams within a watershed.

> $\mathbf{I_{CSL} = L_{CS}/L_{TS}}$, where L_{CS} is the channelized stream length and L_{TS} is the total stream length for the watershed.

Since this index addresses channelization of existing streams, the total stream length does not include the length of artificial ditches excavated in farmfields and forests or the length of dammed sections of streams. It will usually emphasize perennial streams, but could include intermittent streams, if desirable.

The Wetland Disturbance Index (I_{WD}) focuses on alterations of existing wetlands. As such, it is a measure of the extent of existing wetlands that are diked/impounded, ditched, or excavated:

> $\mathbf{I_{WD} = A_{DW}/A_{TW}}$, where A_{DW} is the area of disturbed or altered wetlands and A_{TW} is the total wetland area in the watershed.

Wetlands are represented by vegetated and nonvegetated (e.g., shallow ponds) types and include natural and created wetlands. Since the focus of our analysis is on Anatural habitat,@ diked or excavated wetlands (or portions thereof) are viewed as an adverse action. We recognize, however, that many such wetlands may serve as valuable wildlife habitats (e.g., waterfowl impoundments), despite such alteration.

Composite Habitat Index for the Watershed

The Composite Natural Habitat Integrity Index (I_{CNHI}) is a combination of the preceding indices. It seeks to express the overall condition of a watershed in terms of its potential ecological integrity or the relative intactness of Anatural@ plant communities and waterbodies, without reference to specific qualitative differences among these communities and waters. Variations of I_{CNHI} may be derived by considering buffer zones of different widths around wetlands and streams (e.g., $I_{CNHI\ 100}$ or $I_{CNHI\ 200}$) and by applying different weights to individual indices or by separating or aggregating various indices (e.g., stream corridor integrity index, river corridor integrity index, or river/stream corridor integrity index).

An example of this composite index is given below emphasizing a 100-meter buffer:

$$I_{CNHI\ 100} = (0.5 \times I_{NC}) + (0.1 \times I_{SCI200}) + (0.1 \times I_{WB100}) + (0.1 \times I_{PLB100}) + (0.1 \times I_{WE}), + (0.1 \times I_{SWE}) - (0.1 \times I_{DSF}) - (0.1 \times I_{CSL}) - (0.1 \times I_{WD})$$

where the condition of the 100m buffer is used throughout. (Note: With this size buffer, the stream corridor width becomes 200m.)

A second example shows how weighting may be changed when a river corridor integrity index is added to the equation:

$$I_{CNHI\ 100} = (0.5 \times I_{NC}) + (0.05 \times I_{SCI200}) + (0.05 \times I_{RCI200}) + (0.1 \times I_{WB100}) + (0.1 \times I_{PLB100}) + (0.1 \times I_{WE}), + (0.1 \times I_{SWE}) - (0.1 \times I_{DSF}) - (0.1 \times I_{CSL}) - (0.1 \times I_{WD})$$

The weighting of the indices is debateable, but as long as a standard weighting scheme is applied, the results of this analysis would be comparable between subbasins and watersheds. The same weighting scheme must be used whenever comparisons of this index are made from one watershed to another, for example.

An Application of the Natural Habitat Integrity Indices for a Watershed

In 2000, we conducted a watershed study of two watersheds (Nanticoke and Coastal Bays) in eastern Maryland for the Maryland Department of Natural Resources. The study included the following for each watershed: a wetland characterization, preliminary assessment of wetland functions, an inventory of potential wetland restoration sites, an inventory of wetland and waterbody buffers (100m), an evaluation of the extent of ditching, and calculation of natural habitat integrity indices. This was the first time we applied these indices to a large watershed. The complete results are available for viewing and downloading at the NWI homepage (wetlands.fws.gov). The findings for natural habitat integrity indices will be given for the Nanticoke watershed to illustrate their application for watershed evaluation.

Study Area

The study area is the Maryland portion of the Nanticoke River watershed, a 323-square mile drainage area in eastern Maryland. Major tributaries of this portion of the watershed are Marshyhope, Rewastico, Quantico, and Wetipquin Creeks. It is composed of 61 percent upland, 8 percent deepwater habitat, and 31 percent wetland. Forty-two percent of the watershed is in agricultural usage and 6 percent is developed, while the rest remains in Anatural vegetation@ (e.g., wetlands, forests, thickets, and old fields). The watershed extends into Delaware, but that portion was not evaluated at that time. The Maryland portion includes parts of Dorchester, Wicomico, and Caroline Counties.

Almost 1,400 wetlands were mapped by the NWI in this watershed (Tiner et al. 2000). Roughly 64,000 acres of wetlands occurred in this watershed. Palustrine wetlands were most abundant, covering nearly 47,000 acres (73% of the wetlands), with forested wetlands predominating (80% of the palustrine wetlands). The bulk of the remaining wetlands (or 26%) is represented by estuarine wetlands, mostly emergent types (salt and brackish marshes). From the hydrogeomorphic (HGM) perspective, about two-thirds of the wetlands were terrene (52% of the wetlands, excluding ponds). Interfluve and fringe wetlands were the main types (37% and 35%, respectively) and outflow was the major water flow path descriptor (about 50% of the wetland acreage).

Methods Overview

The foundation of this project was construction of a fairly comprehensive, geospatial wetland database. The existing wetland digital data for Maryland included the NWI data (based on 1:24,000 maps derived from mostly early 1980s-1:58K color infrared photography) and the State=s wetland data (based on digital orthophoto quarter-quads produced from 1989-1:40K color infrared photographs). The State data were used as collateral data to improve the delineation of wetlands in the NWI database. A 100m buffer was positioned around wetlands, waterbodies, and ditches. To evaluate the condition of the upland buffer, we created a land use/land cover data layer by combining existing digital data with new photointerpretation. The state=s digital data on land use and land cover were used as the baseline data and were updated by interpreting 1998 aerial photography (1:40,000 black and white) using a digital transfer scope. The Anderson et al. (1976) land use and land cover classification system was used to classify upland habitats to level two. The following categories were among those identified: developed land (residential, commercial, industrial, transportation/communication, utilities, other, institutional/government, and recreational, farmsteads/farm-related buildings), agricultural land (cropland/pasture, orchards/nurseries/horticulture, and feedlots/holding areas), forests (deciduous, evergreen, mixed, and clear-cut), wetlands (from NWI data), and transitional land (moving toward some type of development or agricultural use, but future status unknown). Data layers were constructed for the entire Aland@ area of each watershed so that information could also be used for assessing their overall ecological condition.

Results for the Nanticoke River Watershed

The values for eight indices for the Nanticoke watershed are calculated and presented in Table 1. The composite natural habitat integrity index had a value of 0.53 using the formula:
$I_{CNHI\,100} = (0.6 \times I_{NC}) + (0.1 \times I_{SCI200}) + (0.1 \times I_{WWB100}) + (0.1 \times I_{WE}), + (0.1 \times I_{SWE}) - (0.1 \times I_{DSF}) - (0.1 \times I_{CSL}) - (0.1 \times I_{WD})$. These indices provide evidence of a stressed system. A pristine watershed has an index value of 1.0 for natural habitat integrity. The value of 0.53 for the Nanticoke watershed signifies significant human modification. While stream corridors seem to be in reasonable shape with natural vegetation encompassing 66 percent of the 200m corridor and 73 percent of the 100m corridor, about half of the wetland and waterbody buffer has been developed (Figure 1). Overall, the Nanticoke watershed has lost about half of its natural habitat and almost 40 percent of its streams have been channelized. While slightly more than half (52%) of the land in the watershed is covered with Anatural vegetation,@ about 42 percent is in agriculture and only 6 percent is developed (Figure 2). Application of these indices to individual subbasins within the watershed could aid in targeting areas for preservation and restoration.

Table 1. Index values for the Maryland portion of the Nanticoke River watershed (Tiner et al. 2000).

Index	**Calculation**	**Value**	**Comment**
Natural Cover	98,544/188,410	0.52	52% of the watershed contains Anatural vegetation@
Stream Corridor Integrity (200m)	13,581/20,552	0.66	66% of the stream corridors are vegetated with Anatural vegetation@
Wetland and Other Waterbody Buffer (100m)	23,181/46,978	0.49	49% of these buffers are colonized by Anatural vegetation@
Wetland Extent	25,387/31,761	0.79	Based on Dorchester Co. portion only which is the least altered section of the watershed; the actual wetland extent is much less than this index suggests
Standing Waterbody Extent	No calculation	1.0+	There has been a net increase in standing open water in the watershed over time, due to the construction of impoundments and ponds.
Dammed Stream Flowage	6.5 miles/259.3	0.03	Only 3% of the perennial stream length has been dammed.
Channelized Stream Length	101.3/259.3	0.39	39% of the perennial stream length has been channelized.
Wetland Disturbance	22,767/64,139	0.35	35% of the wetlands have been partly drained (through ditching), excavated, and impounded (diked)

Figure 1. Condition of wetland and waterbody buffer (100m) in the Maryland portion of the Nanticoke watershed.

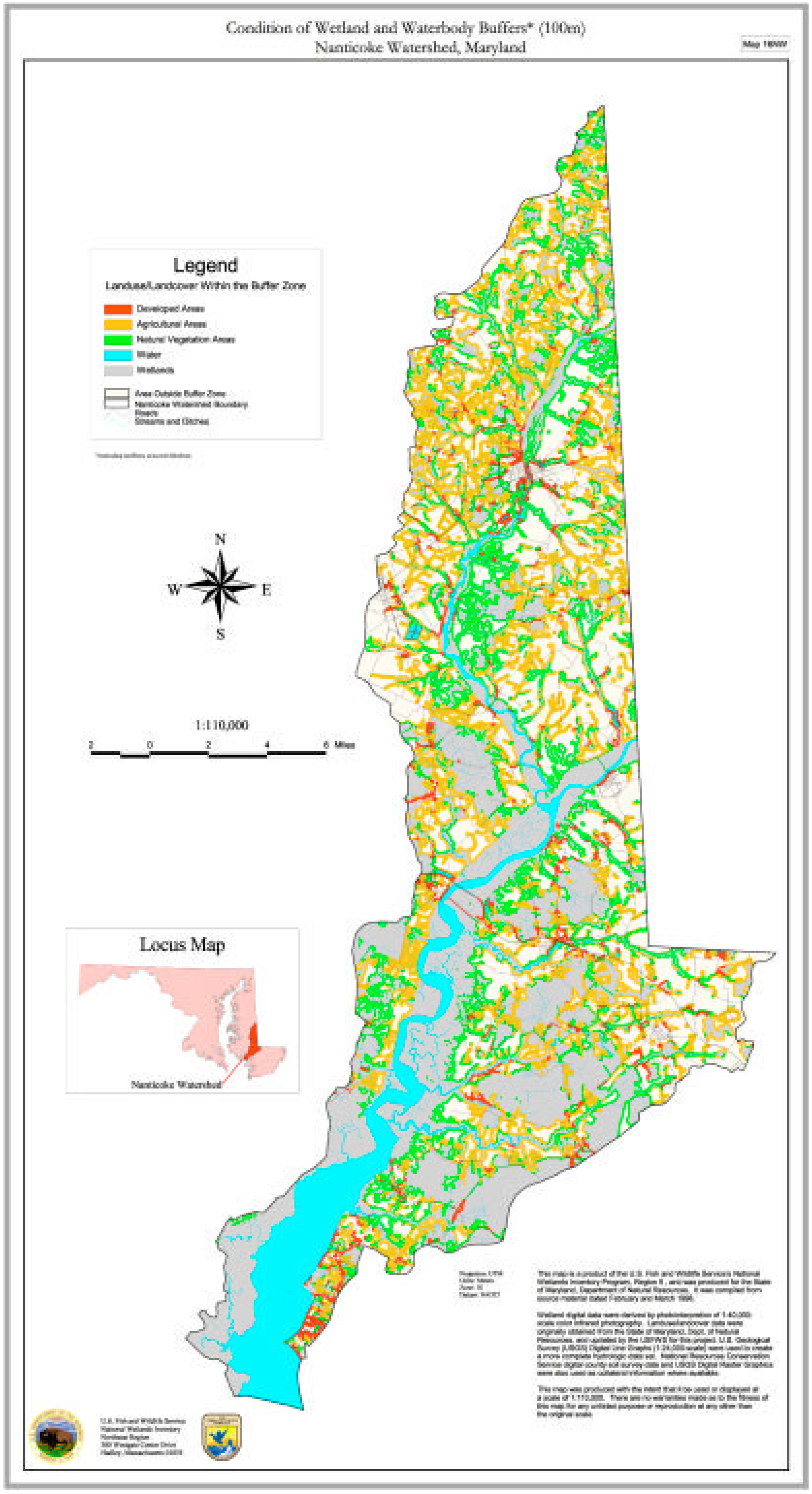
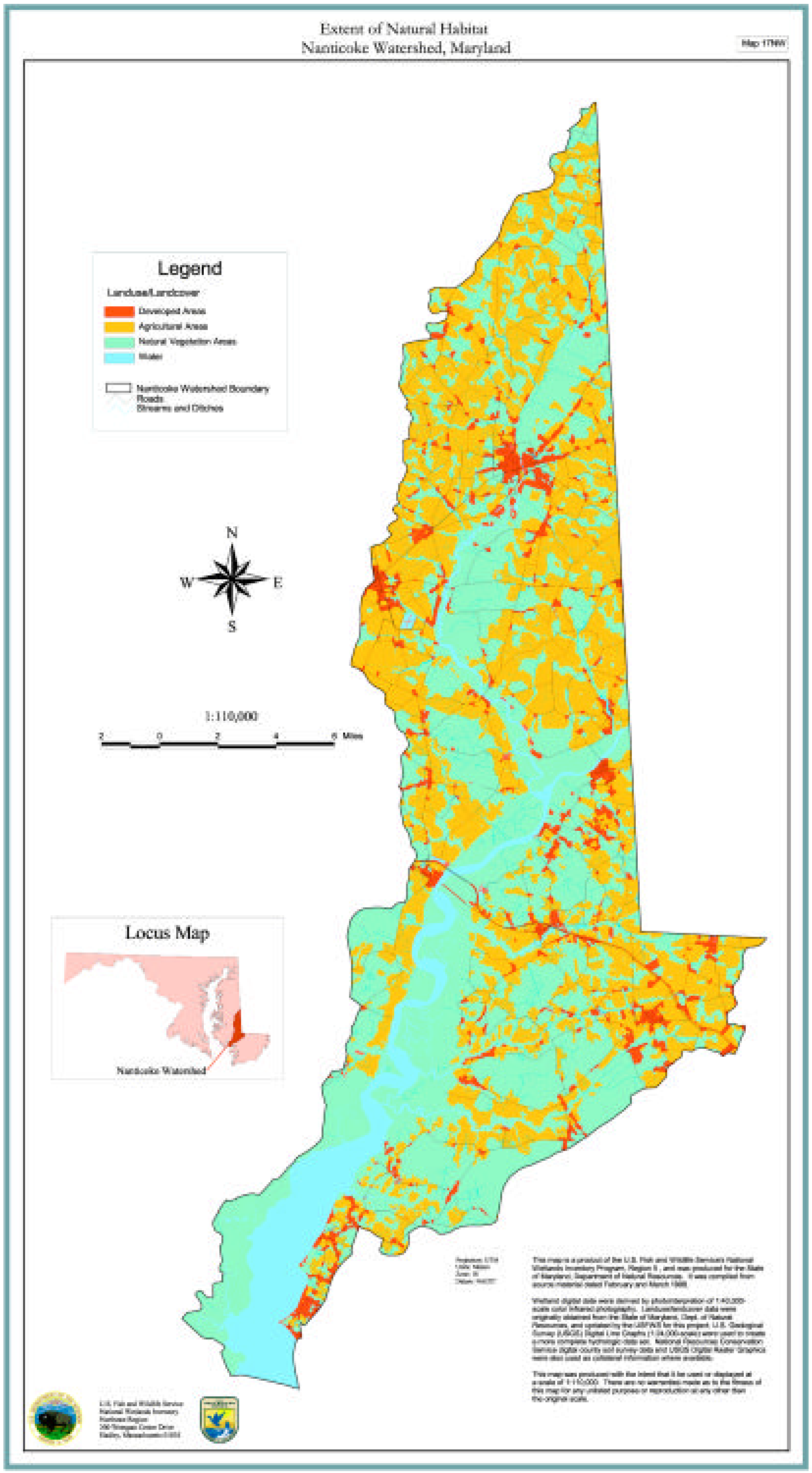


Figure 2. Extent of natural vegetation and developed lands in Maryland's Nanticoke River watershed.



Concluding Remarks

The indices provide valuable information for resource planners and decision-makers. They present a picture of how much natural habitat is present in a watershed and the amount of stream and wetland alteration that has taken place. Moreover, the specific locations of encroachments to wetland and waterbody buffers can be shown on maps which can be prepared using GIS technology. After this type of analysis, maps can be prepared to show the following features: 1) land cover and land use in river and stream corridors and buffers around wetlands, lakes, and ponds, 2) potential sites for restoring vegetated corridors and buffers, 3) channelized streams vs. nonchannelized streams, 4) dammed stream segments vs. free-flowing segments, 5) altered wetlands vs. nonaltered wetlands, and possibly 6) former wetlands vs. current wetlands (where digital soil data are available). Other information can be added to this type of analysis to provide a more complete view of wetlands and disturbance in the watershed, such as a preliminary assessment of wetland functions, an inventory of potential wetland restoration sites, and a map showing fragmented wetlands.

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