AD-A251 739

2

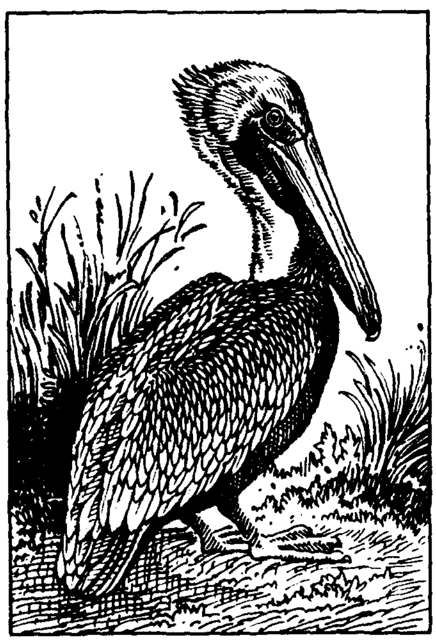
US Army Corps
of Engineers
Waterways Experiment
Station

Wetlands Research Program Technical Report WRP-DE-2

Wetland Evaluation Technique (WET)

Volume I: Literature Review and Evaluation Rationale

by Paul R. Adamus, Lauren T. Stockwell, Ellis J. Clairain, Jr.,
Michael E. Morrow, Lawrence P. Rozas, R. Daniel Smith



October 1991 – Final Report

The following two letters used as part of the number designating technical reports of research published under the Wetlands Research Program identify the area under which the report was prepared:

	Task		Task
CP	Critical Processes	RE	Restoration & Establishment
DE	Delineation & Evaluation	SM	Stewardship & Management

REPORT DOCUMENTATION PAGE

Form Approved
OMB No. 0704-0188

1. AGENCY USE ONLY *(Leave blank)*

2. REPORT DATE
October 1991

3. REPORT TYPE AND DATES COVERED
Final report

4. TITLE AND SUBTITLE
Wetland Evaluation Technique (WET); Volume I: Literature Review and Evaluation Rationale

5. FUNDING NUMBERS

6. AUTHOR(S)
Paul R. Adamus, Lauren T. Stockwell, Ellis J. Clairain, Jr., Michael E. Morrow, Lawrence P. Rozas, R. Daniel Smith

7. PERFORMING ORGANIZATION NAME(S) AND ADDRESS(ES)
NSI Technology Services Corporation, Corvallis, OR 97333;
HC 66, Box 768, West Southport, ME 04576;
USAE Waterways Experiment Station, Environmental Laboratory,
3909 Halls Ferry Road, Vicksburg, MS 39180-6199

8. PERFORMING ORGANIZATION REPORT NUMBER
Technical Report WRP-DE-2

9. SPONSORING/MONITORING AGENCY NAME(S) AND ADDRESS(ES)
US Army Corps of Engineers, Washington, DC 20314-1000;
US Department of Transportation, Federal Highway Administration, Washington, DC 20590;
US Environmental Protection Agency, Office of Wetlands Protection, Washington, DC 20460

10. SPONSORING/MONITORING AGENCY REPORT NUMBER

11. SUPPLEMENTARY NOTES
Available from National Technical Information Service, 5285 Port Royal Road, Springfield, VA 22161

12a. DISTRIBUTION/AVAILABILITY STATEMENT
Approved for public release; distribution is unlimited

12b. DISTRIBUTION CODE

13. ABSTRACT *(Maximum 200 words)*

This is Volume I of a two-volume manual on the Wetland Evaluation Technique. Volume I provides a detailed review of 11 functions and values commonly ascribed to wetlands, including ground water recharge and discharge, floodflow alteration, sediment stabilization, sediment/toxicant retention, nutrient removal/transformation, production export, aquatic diversity/abundance, wildlife diversity/abundance, recreation, and uniqueness/heritage. Each function is examined with respect to important processes and interactions with other functions. Volume I also cites reference material for predictors of these wetland functions as used by the Wetland Evaluation Technique in Volume II.

14. SUBJECT TERMS
See reverse

15. NUMBER OF PAGES
297

16. PRICE CODE

17. SECURITY CLASSIFICATION OF REPORT	18. SECURITY CLASSIFICATION OF THIS PAGE	19. SECURITY CLASSIFICATION OF ABSTRACT	20. LIMITATION OF ABSTRACT
UNCLASSIFIED	UNCLASSIFIED		

14. (Concluded).

Aquatic diversity
Contaminants
Erosion
Evaluation
Fisheries
Floodflow alteration
Ground water
Heritage
Landscape
Metals
Nutrients
Recreation
Riparian
Sediments
Shoreline anchoring
Social significance
Toxicants
Uniqueness/heritage
Values
Water quality
Wetland
Wildlife

Contents

List of Tables

List of Figures

Preface

The Wetland Evaluation Technique (WET), Volume I, is a revision of the Method for Wetland Functional Assessment, Volume I, authored by Mr. Paul R. Adamus and Ms. Lauren T. Stockwell under contract to the Federal Highway Administration (FHWA) in 1983. The FHWA publication is hereafter referred to as Version 1.0. Revisions include modification of the organizational structure; incorporation of additional references; and incorporation of changes suggested by numerous editorial reviewers.

WET is a joint product of the Wetlands Research Program (WRP) of the Environmental Laboratory (EL), US Army Engineer Waterways Experiment Station (WES), Vicksburg, MS, and the US Environmental Protection Agency (USEPA) Environmental Research Laboratory (ERNL), Corvallis, OR. This report was prepared by Mr. Adamus of NSI Technology Services Corporation, under contract with the USEPA ERNL. Other authors include Ms. Stockwell, Mr. Ellis J. Clairain, Jr., Dr. Michael E. Morrow, Dr. Lawrence P. Rozas, and Mr. R. Daniel Smith of the Wetlands and Terrestrial Habitat Group (WTHG), EL. The report was edited by Ms. Jessica S. Ruff of the WES Information Technology Laboratory.

The work was sponsored primarily by the Headquarters, US Army Corps of Engineers (HQUSACE). Partial funding for development of this technique was provided by the FHWA under Order No. DTFH 61-84-Y-30025 to the USACE and by the USEPA under Contract No. 68-C8-0006 to NSI Technology Services Corporation. It has been subjected to peer review processes of both the Corps and USEPA and has been approved for publication. The HQUSACE Technical Monitors for the WRP were Mr. Joseph Wilson, Mr. E. Zell Steever, Mr. David Buelow, Mr. James W. Wolcott, Mr. Phillip C. Pierce, and Mr. John Bellinger. The Contracting Officer's Technical Representatives for the FHWA were Messrs. Douglas Smith and Charles DesJardins.

Many agencies, organizations, and individuals contributed to the revision of WET. Soon after the FHWA published Version 1.0, the US Fish and Wildlife Service initiated a workshop sponsored by 17 Federal agencies to review the method. An Interagency Wetland Values Assessment Coordinating Group (IAWVACG), representing those 17 participating Federal agencies, was formed to coordinate the workshop and provide recommendations. The IAWVACG continues to meet and has been instrumental in the development of WET. The National Wetlands Technical Council held four regional workshops to review Version 1.0 and recommend improvements. The USEPA sponsored three workshops on bottomland hardwood wetlands which provided valuable technical information relevant to the evaluation of those systems. The State of Washington Department of Ecology held a workshop to examine the state of understanding of wetland functions in the Northwest. A symposium held in Portland, ME, by the Association of State Wetland Managers, Inc., provided recommendations that improved WET. Version 1.0 has also been used in Corps of Engineers training, and students have been instrumental in influencing the development of WET.

The authors wish to thank Dr. James S. Wakeley, Dr. Thomas H. Roberts, and Mr. Charles J. Newling of the WTHG, EL, who reviewed this document. In addition, the authors wish to thank the following people for their contributions to this effort: Jon A. Kusler, Joseph S. Larson, Thomas Muir, Henry Sather, William O. Wilen, and Richard Young.

WET has undergone considerable review, reorganization, and revision prior to publication; however, it is expected that it will continue to be modified in response to further reviews, field use, and development of new information concerning wetland functions and values. Users are encouraged to submit their comments to Mr. Ellis J. Clairain, Jr., US Army Engineer Waterways Experiment Station, ATTN: CEWES-ER-W, 3909 Halls Ferry Road, Vicksburg, MS 39180-6199.

The work was monitored at WES under the direct supervision of Dr. Hanley K. Smith and Mr. E. Carl Brown, Chiefs, WTHG, and under the general supervision of Dr. Conrad J. Kirby, Chief, Environmental Resources Division, EL. Dr. Dana R. Sanders, Sr., Mr. Russell F. Theriot, and Dr. Robert M. Engler were Managers of the WRP. Dr. John Harrison was Chief, EL. Project Officer for the USEPA was Dr. Eric M. Preston.

At the time of publication of this report, Director of WES was Dr. Robert W. Whalin. Commander and Deputy Director was COL Leonard G. Hassell, EN.

This report should be cited as follows:

Adamus, Paul R., Stockwell, Lauren T., Clairain, Ellis J., Jr., Morrow, Michael E., Rozas, Lawrence P., and Smith, R. Daniel. 1991. "Wetland Evaluation Technique (WET); Volume I: Literature Review and Evaluation Rationale," Technical Report WRP-DE-2, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

PRINTED ON RECYCLED PAPER

Conversion Factors, Non-SI to SI Units of Measurement

Non-SI units of measurement used in this report can be converted to SI units as follows:

Multiply	By	To Obtain
acres	4,046.873	square meters
acre-feet	1,233.489	cubic meters
Fahrenheit degrees	5/9	Celsius degrees[1]
feet	0.3048	meters
inches	2.54	centimeters
miles (US statute)	1.609347	kilometers
pounds (mass)	0.4535924	kilograms
square miles	2.589998	square kilometers

[1] To obtain Celsius (C) temperature from Fahrenheit (F) readings, use the following formula: C = (5/9)(F - 32).

1.0 Introduction

1.1 Overview of the Manual

Considerable need has developed for a wetland evaluation technique that considers all wetland functions and can be employed rapidly, accurately, and consistently. Although numerous methods were published prior to 1981, none was considered adequate to meet the rapidly growing need (Lonard et al. 1984/*).[1] In 1983, the Federal Highway Administration (FHWA) published a two-volume manual on wetland functional assessment (Adamus 1983/*, Adamus and Stockwell 1983/*). A consensus seemed to emerge that this method was well documented, technically sophisticated, and comprehensive in its coverage of the full range of wetland functions. However, some initial users found the FHWA method to be complex, cumbersome, time consuming, and sometimes confusing (Sather and Stuber 1984/*).

This manual, also in two volumes, represents a revision of the 1983 FHWA method. Considerable effort has been made to simplify and clarify the presentation of the method, now known as the Wetland Evaluation Technique (WET). Volume I provides updated background material and lays out the conceptual basis for WET. Specifically, Chapter 2 of Volume I describes wetland functions in terms of their processes and interactions with other functions. Chapter 2 also provides a relatively comprehensive review of technical literature (mostly through 1987) on each of the functions, and considers whether wetlands, in general, are likely to perform each function. In Chapter 3, the predictors used for determining the probability ratings for wetland functions are discussed. For each predictor, wetland types are ranked with respect to their probability for performing a particular function, and reasons for this ranking are discussed. Chapter 4 discusses the concept of wetland social significance as it is used in WET.

---

[1] References illustrate, where possible (a) where the study was performed, (b) wetland type investigated, and (c) if the reference represents a literature synopsis. More detailed information on the citation format is presented in Chapter 2.0.

Volume II (Adamus et al. 1987) outlines the steps required to implement this method, and discusses its application and limitations in detail. Volume II also includes documentation for a computer program designed to facilitate data analysis in WET.

Development of these manuals reflects a national interest in wetlands. In response, a number of Federal statutes and executive orders have been implemented for the management and protection of US wetlands. Section 404 of the Federal Water Pollution Control Act (Public Law 92-500) as amended is of particular importance in that it gave the US Army Corps of Engineers (USACE/*fo) the authority to regulate dredge and fill activities in "waters of the United States." Executive Order 11990 (Protection of Wetlands), as signed in 1977, established wetland protection as an official policy of the Federal Government. Specifically, Executive Order 11990 stated that "Each agency shall provide leadership and shall take action to minimize the destruction, loss or degradation of wetlands, and to preserve and enhance the natural and beneficial values of wetlands...." This Executive Order, however, does not apply to the Corps of Engineers regulatory program. In implementing the policy to "restore and maintain the biological, chemical, and physical integrity of the Nation's waters" (Nation's waters include wetlands) as set forth in Section 101 of the Clean Water Act (CWA), USACE regulations pursuant to the CWA state that no permit will be granted for the alteration of wetlands that perform functions important to the public interest except when proposed benefits of the alteration outweigh any damage to the wetland resource (33 CFR 320.4(b)(4). In addition, many states have implemented policies or statutes for wetland protection.

1.2 Use of WET

The 1983 version of this method was introduced to meet a relatively specific need for wetland assessment by State and Federal highway departments. It became apparent that the method might be applied to a greater variety and number of wetland assessment needs. In some cases attempts were made to apply the technique to situations that were incompatible, or beyond its designed capability. This led to criticism of the original method.

WET is a broad-brush approach to wetland evaluation, and is based on information about correlative predictors of wetland functions that can be gathered relatively quickly. It is primarily intended for use by persons who do not have ready access to an interdisciplinary team of technical experts on a daily basis. Although WET lacks the region and site specificity and "common sense" of a skilled professional, it is often more replicable, explicit, and capable of tracking a wider range of variables and functions than can be normally evaluated by any single expert.

WET is designed to alert planners, regulators, and others to the probability that a particular wetland performs specific functions, and also to provide insight as to the local, regional, and national significance of those functions. It should not be used where questions regarding wetland functions must be answered definitively. Ratings generated by WET merely provide one of many inputs into the decision-making process. However, it is useful as a screening tool to decide whether or not one should resort to more quantitative methods such as the Habitat Evaluation Procedure (HEP) (US Fish and Wildlife Service 1980/*), the Habitat Evaluation System (HES) (USACE 1980/*fo), or further scientific study and analysis.

Under most circumstances, WET should not be used to decide whether or not any mitigation is required. Rather, its purpose is to provide one perspective regarding how much mitigation effort is justified, after regulatory agencies have decided whether or not mitigation is required in a particular case.

WET can be used to compare ratings of a wetland under various future management or impact scenarios with its present ratings. However, users must understand three important limitations of such use. First, predictions of future physical and biological conditions must be accurate in order for WET to make accurate predictions about functions in the future. Second, because WET is built upon correlative variables rather than causative ones, postimpact ratings should be screened carefully. Last, WET cannot anticipate the cumulative impacts of various combined activities over time.

Subject to the precautions and guidelines specified fully in Volume II, WET can be applied in individual permit applications, advanced identification of important wetlands, selection of least-damaging alternatives, design and monitoring of created/restored wetlands, and prioritization of wetlands for detailed delineation efforts, acquisition, or research. Rapid assessment methods offer a realistic tool for demonstrating consistent considerations of many potential wetland values in situations where interdisciplinary expertise is not available.

1.3 Relationships to Existing Methods for Evaluating Wetlands

The development of this manual would have been more difficult were it not for the existence of other methodologies for evaluating wetlands. Twenty of these were critiqued in detail by Lonard et al. (1981, 1984/*). We acknowledge drawing from, and hopefully improving upon, many of these sources.

Inevitably, the question arises: Why another procedure? The reasons are many, but the most important may be summarized as follows: No wetland evaluation methodology presently exists which (1) addresses all important, presently recognized wetland functions and wetland types, (2) provides specific guidance for estimating the effects of potential impacts, and (3) specifically uses the US Fish and Wildlife Service (USFWS) wetland classification scheme. The importance of using the USFWS classification scheme will be described in Section 1.4. The need for devising a single unified procedure for addressing wetland functions should be apparent after considering the many implicit trade-offs among wetland functions, as described in Chapter 2. Other evaluation methods commonly address only specific functions. For example, only wildlife and fisheries aspects of wetlands are considered by HEP and HES.

1.4 Wetland Classification

Although compatible with most other definitions and classification schemes of wetlands, this manual requires an understanding of the USFWS classification system (Cowardin et al. 1979/*). For example, superficial reading of Cowardin et al. (1979/*) has led some users to conclude that any wetland fringing a river is a "riverine wetland." In fact, many riverside wetlands are considered "palustrine" by the USFWS definition.

The Cowardin et al. (1979/*) classification scheme is used for two reasons:

- The FWS wetland classification scheme, when fully applied, is extremely sensitive to differences among wetlands. By using the classification system, over 100,000 distinct types of wetlands may be distinguished.
- National Wetland Inventory maps, which are commonly used during WET evaluations, use the USFWS wetland classification system.

2.0 Wetland Functions: Processes and Interactions

Wetland **functions** are physical, chemical, and biological processes or attributes of wetlands that are vital to the integrity of the wetland system, and operate whether or not they are viewed as important to society. **Values,** on the other hand, are wetland attributes that are not necessarily important to the integrity of the wetland system itself, but are perceived as being valuable to society.

WET evaluates wetland functions and values in terms of social significance, opportunity, and effectiveness. **Social significance** is the value society places on wetland functions and values as evidenced by their economic worth or official recognition (see Chapter 4 for a more complete discussion of social significance). **Opportunity** is the chance a wetland has to perform a specific function. For example, a wetland that receives sediment has the opportunity to retain sediment, and therefore can perform the function of sediment/toxicant retention, whereas a wetland that receives no sediment has no opportunity to perform this function. **Effectiveness** is the capability of a wetland to perform a function because of its physical, chemical, or biological characteristics.

The 11 functions and values addressed by WET are listed in a tabulation to the right.

Functions	Values
Ground water recharge (GWR)	Recreation (R)
Ground water discharge (GWD)	Uniqueness/ heritage (U/H)
Floodflow alteration (FFA)	
Sediment stabilization (SS)	
Sediment/toxicant retention (S/TR)	
Nutrient removal/transformation (NR/T)	
Production export (PE)	
Aquatic diversity/abundance (AD/A)	
Wildlife diversity/abundance (WD/A)	

Although all 11 functions and values are evaluated for social significance, only floodflow alteration, sediment/toxicant retention, and nutrient removal/ transformation are assessed for opportunity. All nine functions are evaluated for effectiveness. In addition, **habitat suitability** is rated for individual species and species groups for Aquatic Diversity/Abundance and Wildlife Diversity/Abundance.

This report does not devote equal attention to all these functions. Rather, more detailed discussions are given for those functions that are more complex or least understood. Each function is defined and discussed in terms of processes and interactions with other functions and economic values. Interactions are also summarized in Table 1. Consideration of these interactions is important to understand that seldom can a single wetland provide all the functions and values ascribed to wetlands generally.

Technical literature has been extensively cited in this report. To help ensure the validity of the information, most citations were drawn from a review of the peer-reviewed literature. Secondary sources (e.g., theses, reports, other "gray" literature) were used sparingly. Although many of these appear to contain excellent data sets and analyses, their quality is more difficult to verify.

The statements found in the technical literature range from suggestive to conclusive, and judgments of the underlying limitations tend to be very subjective. Few studies objectively quantify the uncertainties involved in extrapolating their conclusions to other wetland types or regions. Moreover, our understanding of wetlands continues to evolve with time. Still, interim

Table 1
Interactions of Wetland Functions and Values (row and column functions in the same wetland)(* = compatible, x = probable conflict, 0 = no significant interaction or effect is unknown)

	Interaction with										
Function	**GWR**	**GWD**	**FFA**	**SS**	**S/TR**	**NR/T**	**PE**	**AD/A**	**WD/A**	**R**	**U/H**
GWR		0	•	0	x	0	0	0	0	0	•
GWD	0		0	x	x	0	0	0	0	0	•
FFA	•	x		•	0	•	0	0	0	0	•
SS	0	x	•		•	•	0	x	x	x	•
S/TR	0	0	•	•		•	0	x	x	x	•
NR/T	•	0	•	•	•		x	0	0	x	•
PE	x	•	0	0	0	0		•	0	0	•
AD/A	x	•	•	0	0	0	0		•	x	•
WD/A	x	•	•	•	0	0	0	0		x	•
R	x	•	0	•	x	0	0	•	•		x
U/H	0	•	0	•	0	0	0	•	•	0	

NOTE: Interactions are not necessarily symmetrical. For example, floodflow alteration is less effective in a wetland that performs ground water discharge, whereas performing floodflow alteration does not necessarily make a wetland less effective for ground water dischage.

generalizations must be made if science is to find application in the policy arena.

In general, wherever citations were included in this document, they support all **OR PART** of the statement made. Where an inference was made, it was mainly in situations where a wetland function had been studied in a different wetland type (or lake). In these situations, we extrapolated results despite the possibility that the various studies used different methods, involved dissimilar taxonomic assemblages, and had other discrepancies. In some extensive references, the particular page number is included in the citation to indicate specifically where the information was obtained.

The number of citations has no bearing on the extent or quality of the knowledge of that particular predictor. This is extremely important for two reasons. First, lack of citation does not necessarily mean lack of knowledge. Some relationships are so obvious or trivial that no one studies them (e.g., wetlands without outlets retain most of their sediment inputs).

Second, extensive listing of information sources does not imply that a broad consensus of opinion exists that the wetland function is easily predicted, or that a great deal is known. **In reality, our understanding of the relationships of rapid, landscape-level predictors to wetland functions is very tenuous and region-specific**, although perhaps no more so than for some other ecosystems. A major purpose of WET is to comprehensively present the knowledge of wetland functions in a comprehensive and systematic manner, so that available information for predicting functions can be rapidly accessed and investigated further.

To reveal more about possible limitations of the cited literature as applied to **different** wetland types, some references in Chapter 5 include additional information regarding (a) where the research was done (state, province, nation, or continent abbreviation), (b) wetland type(s) investigated, if apparent from the author's descriptions, and (c) whether the cited work is a discussion paper or literature synopsis (indicated by a *) or is based primarily on original data, reanalyzed, or modeled data. For the wetland type, the following abbreviations were used (modified from Cowardin et al. 1979/*):

E = Estuarine	em = emergent
M = Marine	fo = forested or scrub-shrub
P = Palustrine	ab = aquatic bed
R = Riverine	t = tidal
L = Lacustrine	

For example, a citation that reads (/Man:Pem) indicates that the research was conducted in Manitoba on a palustrine emergent wetland. A citation that reads (/fo*) indicates a literature review pertaining to forested and/or scrub-shrub wetlands, with no particular geographic focus.

Also, as explained in the introduction to Chapter 3 and included in the Chapter 3 subheadings "Confidence in Ranking," we have described (and if possible, documented) circumstances that could lead to different conclusions than the ones stated. In some cases, only imagination seems to limit the possibilities, so we have focused on those circumstances that seem most probable.

Finally, the user should be aware that although this manual considers those functions most widely ascribed to wetlands, other functions and values may occasionally deserve attention. These include, but are not limited to, the following: harvest of commercial timber, crops, and peat; grazing; extraction of mineral resources; aquaculture; waterborne commerce; urban development; alteration of local, regional, and world climate; importance to the sulfur cycle and oxygen production; retention and detoxification of heavy metals and other hazardous substances that are not necessarily adsorbed to sediment particles; and importance to insects and disease vectors.

2.1 Ground Water Recharge and Discharge

2.1.1 Definition

Ground water recharge (Figure 1) is the movement (usually downward) of surface water, whereas ground water discharge (Figure 2) is the movement (usually laterally or upward) of ground water into surface water (e.g., springs). Shallow recharge and minor ground water discharges are sometimes termed leakage or seepage. When discharge to streams occurs during dry seasons, it is termed low (or base) flow augmentation. This manual does not differentiate between shallow, lateral, and deep recharge. Shallow and lateral recharge are local phenomena of direct value to fewer water users than deep recharge, which is more pertinent to regional ground water systems.

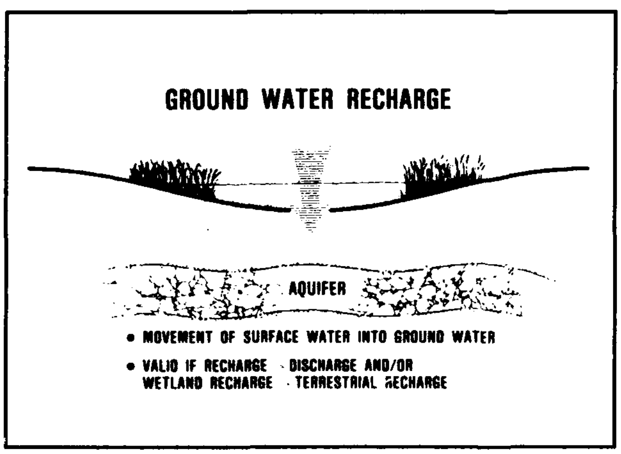


Figure 1. Schematic representation of ground water recharge

The little information available on ground water exchange relationships of wetlands is largely contradictory, suggesting a need for highly site-specific analyses. For recharge, adjacent undeveloped uplands are usually, but not always, more important than wetlands. No inherent wetland characteristic encourages recharge or discharge.

In a sample of 63 glaciated Wisconsin lakes, many of them dominated by wetland vegetation, Born et al. (1979/WI:L) reported that about 20 percent recharged ground water to some degree on a net annual basis. Moreover, prairie potholes studied by Lissey (1971/Man:Pem) were more important than uplands for recharge. Of four potholes studied by LaBaugh et al. (1987/ND:Pem), one recharged ground water and two predominantly discharged ground water. Recharge of ground water by wetlands has also been documented by Wood and Osterkamp (1984), Mills and Zwarich (1986/MAN:P), and Siegel (1988a/AK:P). Some seasonally flooded wetlands adjacent to riverine systems, which are frequently scoured of impermeable sediments that otherwise inhibit ground water movement, may be more important than uplands for net annual recharge. Intermittently flooded wetlands were implicated as a major recharge mechanism in an Arizona riverine system (Hanks et al. 1981/AZ:R). It is important to realize that the recharge function of wetlands has been inadequately studied and in some regions has apparently been totally unmeasured by systematic water budget studies.

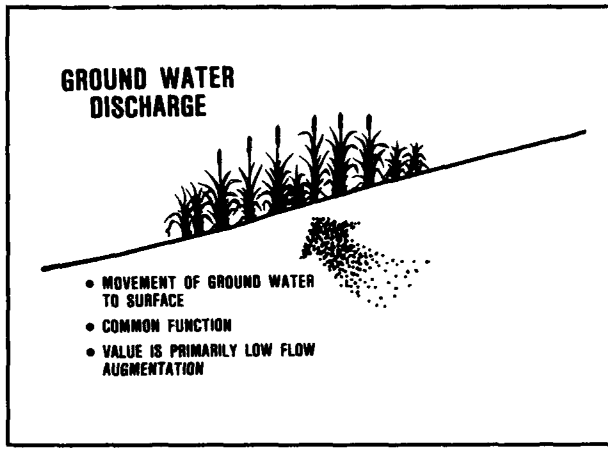


Figure 2. Schematic representation of ground water discharge

Existing data suggest that wetlands are commonly important ground water discharge areas (Carter and Novitzki 1988/VA,NY:P), but discharge is generally so minor that few wetlands augment base flow (Carter et al. 1979/*; Siegel 1988a/AK:P,E). This is because most of the discharge to streams probably originates from the shallow upper layer of the water table (O'Brien 1977/MA:P, Schwartz and Milne-Home 1982) and occurs during wet seasons when streams least need it. Much discharge during dry seasons may be lost through evapotranspiration (Verry and Boelter 1979/*P). In fact, in a study of Wisconsin wetlands, Novitzki (1979/WI) found that wetlands as a whole worsened low-flow conditions. Several studies (Bay 1969/MN:P; Woo and Valverde 1981/ONT:Pfo; Eli and Rauch 1982/WV:P) indicate little effect on base flows that can be attributed to wetlands, except possibly in the case of an unusually long drought, or during the hours immediately following a storm. Nevertheless, many wetlands with minimal erect vegetation located in humid regions of very low potential evapotranspiration, and having basin morphologies conducive to storing large amounts of water, probably have some capacity for augmenting low flows. For example, in a study of 38 Minnesota drainage basins, Ackroyd et al. (1967/MN:R) concluded that lakes and wetlands, in general, augment flow by means of temporary surface storage of run-off, followed by release to streams via ground water flow. Low-flow augmentation by wetlands was also documented in a paired-watershed study in southern Ontario (Bertulli 1982/ONT:fo).

2.1.2 Processes

The processes or factors affecting ground water movement are as follows:

- Ground water flow rates and storage capacity.
- Direction and location (within the wetland) of ground water movement.
- Evapotranspiration.

Ground Water Flow Rates and Storage Capacity. The movement of ground water to or from a wetland depends primarily upon (1) the elevation of the wetland relative to the ground water (elevation head), (2) the mass and pressure of water (pressure head), and (3) the physical characteristics and frictional resistance of the sediments and underlying strata (hydraulic conductivity). The elevation head is influenced by changes in the landscape which constrict flow. Wetlands that discharge through constricted outlets between steep slopes may have large elevation heads, as may wetlands caused by dams. Other situations that have resulted in a wetland being topographically "perched" generally increase the elevation head. In contrast, marine and estuarine wetlands have little elevation head (except as a temporary result of tides), and hence have little recharge value (Clark and Clark 1979/*).

Pressure head is influenced in a downward direction by the mass of water and in an upward direction by confined ground water flow caused by underlying strata (e.g., artesian springs). Larger wetlands (i.e., greater water volume) have greater pressure head, other factors being equal. The effects of the elevation and pressure heads combined result in a hydraulic gradient.

Hydraulic conductivity can be influenced by the physical aspects of the sediments such as porosity, transmissivity, permeability, and storage capacity. Ground water moves best through coarse sands or gravels and successively slower through fibric peats, deep sapric peats, and clays (Mitsch and Gosselink 1986:72/*). Although rooted wetland vegetation may enhance permeability of compacted bottom sediments (Eisenlohr 1966/ND:Pem), the more usual effect is that as vegetation dies, it accumulates and forms an organic layer that is less permeable to ground water exchange. This may even isolate or seal a basin from the ground water. However, Born et al. (1979/WI:L) reported significant ground water recharge from a lake with a 20-foot[1] thickness of unconsolidated organic sediments and significant discharge into a wetland with a 50-foot thickness of organic sediments. Sediments of high clay content are usually more effective seals. In such situations, most ground water exchange may be along wave-exposed shorelines where less clay is present.

---

[1] A table of factors for converting non-SI units of measurement to SI (metric) units is presented on page vii.

Direction and Location of Ground Water Movement. The water in a wetland represents the component inflows of net precipitation, upland run-off, flowing surface water, tidal inflow, and ground water discharge, balanced against outflows of ground water recharge, evapotranspiration, surface water, tidal outflow, and possibly consumption. Not all wetlands have all of these inputs and outputs. Some wetlands (seepage wetlands) are dominated by ground water inputs and/or outputs, whereas others (drainage wetlands) are surface water-dominated (Birge and Juday 1934/WI:L, Born et al. 1979/WI:L).

The role ground water plays in the hydrologic budget depends upon the physical position of the wetland with respect to the water table. Both ground water discharge and ground water recharge occur when the water table intersects the surface water of a wetland. When the wetland is perched above the water table, only ground water recharge is possible. Perched wetlands tend to be only seasonally wet (Novitzki 1979/WI).

The following situations (each occurring within the same wetland) are possible (Born et al. 1979/WI:L):

- Discharge in shallows with recharge in deeper areas.
- Recharge in shallows with discharge in deeper areas.
- Shallow flow-through (discharge at one end of the basin, recharge at the other) with discharge or recharge in deep areas.
- Deep flow-through with shallow discharge or recharge.

Ground water flow may enter or leave a wetland at any point, but in wetlands that are wider than the thickness of underlying porous sediments, most significant movement (usually discharge) will likely occur where vegetation is absent, usually along wave-scoured, unstable sandy shorelines (McBride and Pfannkuch 1975/MN:L). Locations of recharge and discharge within the basin, as well as the overall magnitude of recharge and discharge, frequently vary by season and from year to year (Williams 1968; Lissey 1971/Man:Pem; Lichtler et al. 1976/FL:L; O'Brien 1977/MA:P; McKay et al. 1979/IL:P,R; Winter and Carr 1980; Siegel 1981/MN:P; Wilcox et al. 1986). Wetlands with a shrubby fringe may "recharge" in their shallow areas during the growing season to compensate for water lost from the shoreline soils by transpiring shoreline shrubs. Such lateral movement, although ecologically important, is usually of minimal value to replenishing other wetlands and regional aquifers of interest to society for water supply.

Evapotranspiration. Evapotranspiration refers to the combined processes of evaporation and plant transpiration. The role of transpiration becomes obvious as wetland water levels increase after removal of timber and as they fluctuate under normal circumstances on a diurnal basis (Wharton 1970/*fo). Transpiration is greatest during the growing season and during daylight hours. Densely vegetated wetlands are thought to lose water faster (through transpiration) than sparsely vegetated wetlands, and consequently have less water available to augment low flows or recharge ground water (Miller 1965/NJ:P, Bay 1967/MN:P, Meyboom 1967/MAN, Hutchinson 1975/*L,[1] Verry and Boelter 1979/*P). In Florida, emergent wetlands apparently lose water at a faster rate than open water (Brown 1981/FL:Pfo). Cattails[2] and waterhyacinth have been reported to transpire two to three times the amount of water evaporated from adjacent open water.

Many studies were not based on in situ measurements, and therefore may not be reliable (Idso 1981/*, Winter 1983). Also, some recent studies indicate that in many situations wetland vegetation contributes little, if any, to water loss on a net annual basis (Gessner 1959, O'Brien 1977/MA:P, Fetter 1980/*, Idso 1981/*, Bertulli 1982/ONT:fo). In some instances where open water is prevalent (Idso 1981/*), and in regions with short growing seasons, wetland vegetation with a substantial underlying litter layer actually enhances storage (Odum 1970b/*E) on a net annual basis. It does so by reducing temperature (by shading) and wind speed, which would increase evaporation (Eisenlohr 1966/ND:Pem, Hickok et al. 1977/MN:P, Mitsch et al. 1977/IL:fo), and by intercepting snow and concentrating snowmelt.

2.1.3 Values and Interactions With Other Wetland Functions

Floodflow Alteration. Ground water recharge may help desynchronize flood peaks. However, wetlands which discharge ground water when saturated with *run-off do not significantly lessen flood peaks, and may even increase them* (Young and Klawitter 1968/*fo).

Sediment Stabilization. Ground water discharge can have a destabilizing effect on some steep shorelines (Gray 1977) and streambed sediments (Harrison and Clayton 1970/AK:R). However, because ground water in northern regions is generally warmer in winter than ambient water temperatures, discharge may also keep shorelines free of potentially erosive ice for longer periods. Also, aquatic vegetation encouraged by moisture made available by discharge in dry regions may help anchor the shoreline.

Sediment/Toxicant Retention. Ground water discharge normally has only a localized effect, possibly discouraging the settling of fine sediments. Recharge might reduce turbidity of downcurrent areas, as surface waters percolate through underlying sediments.

Nutrient Removal/Transformation. Ground water exchange may profoundly affect wetland water quality, particularly in noncontiguous basins. Ground water discharge may either improve or degrade water quality,

---

[1] For the common reed, the ratio has been reported to be as high as 7 to 1 (Hutchinson 1975/*L).

[2] Common names of organisms are used throughout this document; the corresponding scientific names are listed in Appendix A.

depending on evaporation rates, geology of underlying strata, and rate of discharge. Hull et al. (1981/FL) determined that the water quality of the Suwannee River, Florida, was controlled in the headwaters by surface flow from wetlands, and in downstream reaches by ground water discharge. Ground water discharge may aggravate salinity problems in prairie pothole wetlands, and transport nutrients into lakes (Loeb and Goldman 1979/CA:L, Winter 1983) and estuaries (Sewell 1982/AU:E, Johannes and Hearn 1985/AU:E). Removal of nutrients may also occur, however, via denitrification and filtration as wetland waters pass through the ground water system.

Production Export. By potentially extending the growing season in normally ice-covered waters, and increasing the availability of soil moisture and dependability of flows downstream, ground water discharge might increase net annual primary productivity and the sustained export of nutrients. Where ground water exchange with a wetland is great, seasonal stabilization of water levels may benefit some wetland plants. However, primary productivity may decline in some systems dependent upon regular fluctuations (e.g., bottomland hardwoods), resulting in reduced production export. Fluctuating water levels are more characteristic of ground water recharge basins (Sloan 1979/ND: Pem). Ground water discharges, being more alkaline than surface waters in most regions, may also increase the capacity of surface waters to buffer the effects of acid precipitation, and as a result help maintain productivity.

Aquatic Diversity/Abundance, Wildlife Diversity/Abundance. Discharge is important not only for maintaining low flows essential to fisheries, but also for maintaining vegetation and drinking water for wildlife. Seepage discharged through gravel is essential to spawning and rearing of salmonid fishes (Scarnecchia 1981, Bilby 1984). In summer, springs provide a cool refugium for salmonid fish; this is particularly important in wide rivers and developing watersheds where natural sources of shade are limited. Discharging springs often keep important northern wetlands free of ice for long periods in winter, increasing their use by waterfowl. Discharge in areas such as the prairie potholes region maintains the quality of vital waterfowl breeding areas (Swanson et al. 1983/ND:Pem). Elevated water hardness levels associated with discharge reduce the toxicity of many chemicals. A disproportionate number of rare plant species is found in ground water discharge wetlands (e.g., springs, seeps, fens), especially in calcareous regions (Williams and Dodd 1979).

Ground water discharge in coastal estuaries may be a key regulator of salinity at the microscale. Salinity determines where many fish larvae will survive and grow. Discharge of ground water is generally less variable on a yearly and seasonal basis than surface run-off, and thus may be an important determinant of the nursery value of poorly flushed estuarine microhabitats (Welsh et al. 1976/*E). Ground water discharge, most entering within 300 feet of the shore, was estimated to account for 10 to 20 percent of freshwater input to a large coastal bay (Great South Bay, New York) (Bokuniewicz 1980/NY:M).

Recreation. Discharge may help maintain base flows essential to water sports.

Uniqueness/Heritage. The effects of ground water recharge or discharge on the uniqueness/heritage aspects of wetlands depend on the situation.

2.1.4 Direct Economic Significance

Recharge, measured on a net annual basis, is important for replenishing aquifers used for water supply, particularly in situations where aquifers are threatened by drought, overuse, contaminants, or saltwater. Recharge from wetlands in a 1,600-acre prairie pothole area was estimated to provide 12 acre-feet to the aquifer, enough to support 1,699 head of cattle for 1 year (Hubbard and Linder 1986/SD:P). Discharge may also be critical for maintaining soil moisture in arid agricultural regions. In scattered instances, recharge from wetlands may discolor or impart an unpleasant taste to aquifer waters, and discharge may create soil chemical conditions detrimental to cultivation.

2.2 Floodflow Alteration

2.2.1 Definition

Floodflow alteration (Figure 3) is the process by which peak flows from run-off, surface flow, ground water interflow and discharge, and precipitation enter a wetland and are stored or delayed in their downslope journey. Floodflow alteration also includes floodflow desynchronization, which is the process by which flood waters are stored in numerous wetlands within a watershed, and then gradually released in a staggered manner. This gradual release usually results in more persistent flow peaks downstream. Storage may be measured in seconds or in months, reduction of high-water levels associated with peak flows in fractions of an inch or in feet, and diminution of flooded area in square feet or square miles. Not included in this Section is flood damage caused by waves or storm surges. These topics are addressed in Section 2.3 (Sediment Stabilization).

Anywhere a depression of any size occurs in the landscape, some quantity of water can potentially be stored on the surface and/or in underlying sediments. Because many depressions in the landscape will contain wetlands (at least periodically), wetlands not already filled to capacity with surface water are usually effective for flood storage (Carter et al. 1979/*, Clark and Clark 1979/*, Novitzki 1979/WI). However, many wetlands quickly become saturated and filled to capacity, especially in developed watersheds, in the lower reaches of watersheds, and in watersheds with little wetland acreage (see Chapter 3). Undeveloped terrestrial environments, despite their steeper slope, occasionally have greater flood storage value (i.e., greater recharge and less runoff) than wetlands, particularly where the permeable and unsaturated nature of terrestrial soils is sufficient to offset their greater slope (e.g., Young and Klawitter 1968/*fo, Chamberlain 1982/*, Schwan 1985/*), or where their vegetative cover is sufficiently extensive to desynchronize run-off by frictional drag and dissipate flood waters by evapotranspiration.

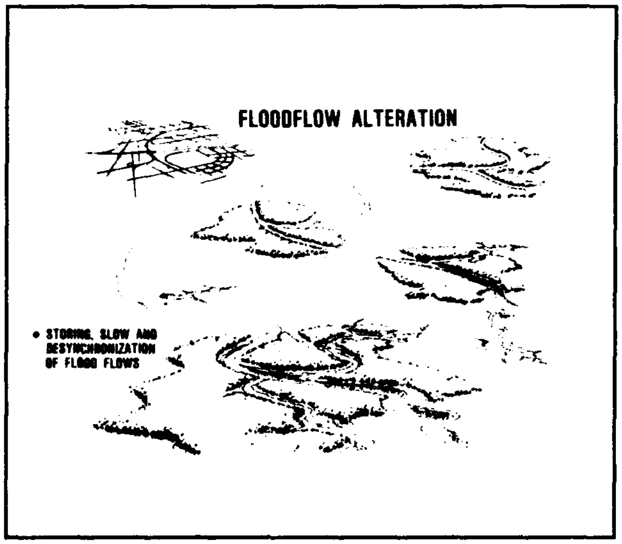


Figure 3. Schematic representation of floodflow alteration by wetland areas

The evidence for a significant floodflow alteration capacity of wetlands is not unequivocal, and the relative effectiveness of wetlands for this function may be region-specific. A simulation analysis of the lower Ohio River revealed no change in flood height even when floodplain storage was eliminated and roughness was decreased by 20 percent (Johnson and Senter 1977). Empirical analyses conducted in several states have found that upstream storage in lakes, wetlands, and floodplains is a statistically significant but relatively weak predictor of downstream surface hydrology (e.g., Spear and Gamble 1964, 1976; Price 1977; Whetstone 1982; Eli and Rauch 1982/WV:P).

However, several other empirical studies (e.g., Conger 1971/WI, Kloet 1971/ND, Moore and Larson 1979/MN:Pem, Novitzki 1979/WI, Rannie 1980, Ammon et al. 1981/FL:P, Brun et al. 1981/ND) and computer simulation analyses (e.g., Campbell and Johnson 1975, Bedient et al. 1976/FL, Dybvig and Hart 1977, Moore and Larson 1979/MN:Pem, Flores et al. 1981, Brown and Sullivan 1988/FL:P) have supported the importance of wetlands (or wetlands plus lakes) for altering floodflows. Some of these have indicated that the consequences of wetland loss are most severe if wetland filling occurs in situations where other wetlands/lakes comprise less than about 10 percent of the watershed area above the point of flooding.

One of the geographically broadest, landscape-level studies of the significance of surface storage was that of Thomas and Benson (1970). Decreased storage (i.e., decreasing percent of total watershed occupied by "lakes and swamps") was determined to be a significant correlative of the following stream hydrologic conditions:

- **In the Eastern United States** (n = 41 watersheds, Potomac River Basin only):
  - increased annual peak discharge recurring at 10-, 25-, and 50-year intervals

- ➪ decreased average streamflow in November, December, January, and July
- ➪ decreased year-to-year variability of July streamflow

- **In the Southern United States** (n = 42 watersheds, mainly Louisiana Pine Hill region)
  - ➪ increased annual peak discharge recurring at 1.2-, 2-, and 5-year intervals
  - ➪ increased magnitude of 7-day scouring streamflow that occurs at a 2-year interval
  - ➪ increased year-to-year variability of January streamflow

- **In the Western United States** (n = 44 watersheds, California's Central Valley only)
  - ➪ decreased magnitude of 7-day scouring streamflows that recur at intervals of 2, 10, and 20 years
  - ➪ decreased magnitude of 3-day scouring streamflows that recur at intervals of 10 and 20 years
  - ➪ decreased mean monthly streamflow
  - ➪ decreased mean streamflow in June and July
  - ➪ decreased year-to-year variability of June and July flow

- **In the Central United States** (n = 41 watersheds, mostly Kansas; the storage variable in this case was decreased "alluvial soils area")
  - ➪ increased mean streamflow in March and April
  - ➪ increased year-to-year variability of annual flows

Storage capacity of prairie wetlands was estimated at 12 inches per acre (Kloet 1971/ND) and sometimes can be many times that amount (Hubbard and Linder 1986/SD:P), depending on prior degree of saturation of wetland soils and type of underlying ground water connection. Another study in the pothole region found that wetlands collectively could store 72 percent of the total run-off volume from a 2-year frequency run-off event and 4 percent from a 100-year event (Ludden et al. 1983/ND:P).

In New England, a simulation of the Connecticut River indicated that a 10-percent reduction in floodplain storage would increase flood height by 1 foot; a 20-percent reduction would cause a 7-toot increase (Dewey and Kropper Engineers 1964). For the Charles River in Massachusetts, a simulation model by the Corps of Engineers indicated that downstream flooding could be more effectively reduced by preventing floodplain encroachments than by constructing control works.

The social significance of these figures depends on site-specific conditions, such as proximity and type of dwellings along the shoreline, and the size and frequency of flood that can be expected. In Florida, storage of 166 acre-feet per square mile of development was adequate to reduce flood peaks to predevelopment levels (Wycoff and Pyne 1975/FL).

Nevertheless, in most instances, wetlands are more effective than developed terrestrial environments for flood storage and desynchronization (Clark and Clark 1979/*, Novitzki 1979/WI). Comparisons of watersheds before and after wetland drainage (Brun et al. 1981/ND) and region-wide studies of multiple watersheds with drained versus undrained wetland acreage (Moore and Larson 1979/MN:Pem) both strongly suggest the importance of wetlands for desynchronization of peak flows.

Of course, storing water almost inevitably means that unflooded areas adjacent to the storage wetland will be flooded at the expense of protecting downstream areas. This reduces the potential for cropland and other uses adjacent to the wetland. However, the economic consequences of slightly flooding each small storage wetland are usually proportionately less than if flows are combined and their cumulative effect is allowed to occur downstream.

Ultimately, the question arises as to whether the amount of water stored by wetlands is significant in terms of minimizing downstream property damage. Most estimates suggest that few wetland watersheds, and fewer individual wetlands, are capable of significantly desynchronizing flows from very severe (e.g., 50- or 100-year probability) floods, which are responsible for most property damage (Clark and Clark 1979/*). A single study from the lower portion of a southern watershed with extensive forested wetlands suggested that storm-flow flooding was more severe and flashy (i.e., greater hourly fluctuations) than in a watershed with less wetland acreage (Young and Klawitter 1968/*fo). Nevertheless, by being poor sites for development, wetlands inhibit other land uses which would be susceptible to flood damage.

Flood storage, as defined in this manual, is significant only in palustrine, lacustrine, and upper riverine wetlands. Marine, estuarine, and lower riverine wetlands sometimes reduce coastal flooding by desynchronizing runoff or storm surges, but they could aggravate it if tidal conditions are right (Clark and Clark 1979/*, Carter et al. 1979/*). Wetlands also dissipate the energy of waves and currents which cause flooding. This is discussed in Section 2.3.

2.2.2 Processes

Major processes or factors that affect floodflow alteration are:

- Magnitude and duration of storms.
- Run-off from upslope areas.
- Above-ground storage capacity.
- Frictional resistance.

- Below-ground storage capacity.
- Position of wetland in the watershed.

Magnitude and Duration of Storms. The intensity and duration of storm events directly influence the amount of run-off from a wetland. Wetlands in watersheds characterized by high-intensity, short-duration storm events (often termed "flashy" watersheds) are more likely to receive flood waters than wetlands in watersheds characterized by more attenuated run-off (e.g., extensive snow-melt, rainfall of long duration).

Run-off from Upslope Areas. Flood waters enter the wetland as run-off and ground water discharge. Run-off is most likely to reach wetlands in substantial quantities when drainage areas are large and have steep slopes, shallow or impervious soils (e.g., clay or frozen), sparse vegetative cover (less opportunity for evapotranspirative loss), channelized tributaries, and landuse practices that do not incorporate conservation practices. Wetlands can store peak flows most effectively if the flows enter gradually (e.g.,where the drainage area has gentle, vegetated slopes and deep permeable soils), and if the drainage area is small relative to the size of the wetland. Watershed development is associated not only with wetland drainage, but also with increased coverage by impervious surfaces, which results in increased run-off. For example, peak flows increased "significantly" in the Oregon coastal range when roads (unpaved) occupied over 12 percent of the watershed area (Harr et al. 1975/OR:fo). In Florida, storage of 166 acre-feet per square mile of development was needed to reduce flood peaks to predevelopment values (Wycoff and Pyne 1975/FL).

Above-Ground Storage Capacity. Storage (as surface water) of flood waters depends primarily on the morphology of the wetland. The volume of water that can be stored depends on the type and location of any outlets. Of course, wetlands without outlets must store all incoming water, often resulting in considerable surface expansion. If an outlet is present, one that is constricted and located high in elevation relative to the wetland will store more water.

Frictional Resistance. Desynchronization of peak flows depends on their being slowed not only by channel constrictions, but also by frictional resistance from the bottom. Frictional drag and the potential for desynchronization will be greatest where: (1) the wetland is wide enough to intercept most of the flow, (2) the density of wetland vegetation or other obstructions (e.g., boulders, logs, hummocks) is great, (3) the rigidity of obstructions is adequate to resist flood velocity, and (4) vegetation or obstructions are not deeply submerged by flood waters. Desynchronization of run-off, as partly determined by frictional resistance of wetland vegetation, may be most capable of mediating downstream floodflows when it occurs in headwater wetlands (Flores et al. 1981), whereas in downstream wetlands the storage process may be more important than any desynchronization effect (Ogawa and Male 1983/*MA). In low-relief landscapes, headwater areas represent 30 to 50 percent of the landscape, and wetlands (mostly noncontiguous ones) comprise at least 30 percent of the total watershed; in high-relief watersheds, headwaters are proportionately small, and wetlands (mostly contiguous and riverine ones) comprise less than 10 percent of the total watershed (Brown and Sullivan 1988/FL:P).

Below-Ground Storage Capacity. Sediments underlying wetlands normally are saturated or impervious, and therefore have little capacity to store flood waters. However, if large areas are exposed to frequent water-level fluctuations (e.g., due to artificial manipulation or "flashy" runoff events), sediments may be unsaturated when floodwaters rise, and some below-ground storage capacity may exist. Under such conditions, and especially if water-retentive vegetation predominates (e.g., unsaturated moss wetlands), the wetland may act for short periods like a sponge. In most situations, below-ground storage capacity is less than above-ground capacity.

Position of Wetland in the Watershed. If wetlands "high" in the watershed have been drained, detention of floodwaters by wetlands along the main stem "low" in the watershed might, at least theoretically, aggravate flooding by helping synchronize local run-off with surface flows arriving from higher in the watershed. Simulation of a hypothetical 10-square mile watershed indicated that detention basin networks are more effective if located in the upper 40 to 80 percent of a watershed than in areas farther downstream (Flores et al. 1981).

However, wetlands along streams low in the watershed (fifth-order streams) were found by Ogawa and Male's (1983/MA:P) simulation studies to reduce flooding over a greater downstream area (exceeding 8 miles) than wetlands associated with first- through third-order streams, which reduced downstream flooding significantly only over an approximately 2-mile reach. Further, wetlands low in the watershed were important regardless of the total amount of other storage available in the watershed, while individual wetlands high in the watershed (stream order 1 and 2) ceased to play a major role in floodflow attentuation as soon the acreage of other wetlands above them in the watershed exceeded 7 percent of the total (Ogawa and Male 1983/*MA).

2.2.3 Interactions With Other Wetland Functions

Ground Water Recharge, Ground Water Discharge. Flood storage increases the probability for ground water recharge. However, recharge to economically and ecologically important aquifers is less likely when floodflows are retained only briefly within a wetland. Storage of water in upstream wetlands may result in recharge, with the possibility for subsequent discharge to wetlands downslope. However, if trapping of fine sediments also occurs with flood storage, recharge to underlying aquifers may be reduced.

Sediment Stabilization. Flood storage reduces the need for shoreline anchoring in downstream areas. However, within the wetland where flood waters are being stored, a significant need for this function may still exist.

Sediment/Toxicant Retention. Flood storage greatly enhances the immediate sediment trapping capability of a wetland. However, over long periods of time, sediment trapping in upper watershed wetlands may reduce the sediment storage capacity of these wetlands, while maintaining storage capacity in the lower watershed where it is needed less. This may lead to synchronization of drainage from the upper watershed with peak run-off from the lower watershed, thus increasing the flood peaks in the lower watershed.

Nutrient Removal/Transformation. Flood storage enhances nutrient retention by providing the opportunity for sediments and associated nutrients to settle and be deposited in the wetland.

Production Export. Flood storage and production export seem to be incompatible because when flood waters are retained, so are nutrients (at least those in particulate form). Desynchronization and attenuation of peak flows limit scouring of downstream areas, which is necessary for effective nutrient export to downstream food chains. On the other hand, by desynchronizing and prolonging flows and the organic material they carry, upstream wetlands allow detrital foods to be processed by a wider variety of organisms, and ensure freshwater flows to estuaries over a longer portion of the year. Both of these consequences could result in increased downstream value of production export.

Aquatic Diversity/Abundance, Wildlife Diversity/Abundance. The effect of this function is variable. Flood storage provides feeding and nesting areas for aquatic furbearers, waterfowl, and wading birds. However, for a wetland to be most effective for flood storage, its permanent pool must be very small, and this, coupled with major fluctuations in water level, may be undesirable for fish and wildlife. Nonetheless, gradual release of stored water is usually more beneficial to fish and wildlife downstream than sudden and large peak flows, although such peak flows may be necessary for dispersal and germination of some wildlife plant foods, upstream migration of fishes, flushing of silt from spawning gravels of some fishes, and enrichment of riparian soils. Peak freshwater flows to estuaries may actually increase the inflows of saline waters along the bottom, and can enhance the transport of larval and juvenile fishes and crustaceans (Weinstein et al. 1980/NC:E).

Recreation, Uniqueness/Heritage. The interaction of floodflow alteration with this function will depend on the situation, although under most circumstances they will probably be compatible. For example, canoing through a bottomland hardwood forest can only be accomplished when the wetland is flooded. However, storage of flood waters may make some sites in wetlands temporarily unavailable to recreationists, scientists, educators, and tourists.

2.2.4 Direct Economic Significance

Government payments for flood-related disaster assistance total millions of dollars annually, and several billion more have been invested in structural measures to control floods. In North Dakota alone, annual flood damage to highways, railroads, crops, and property averages $100 million (Cernohous 1979/ND). The economic significance of flood storage depends on site-specific conditions, such as proximity and type of dwellings along the shoreline, and the size and frequency of floods.

2.3 Sediment Stabilization

2.3.1 Definitions

Sediment stabilization (Figure 4) consists both of shoreline anchoring and dissipation of erosive forces. Shoreline anchoring is the stabilization of soil at the water's edge or in shallow water by roots and other plant parts. Dissipation of erosive forces is the lessening of energy associated with waves, currents, ice, water-level fluctuations, or ground water flow. Our definition of sediment stabilization includes any wetland-caused decrease of erosive energy or increase of shoreline anchoring whether or not erosion is significantly reduced.

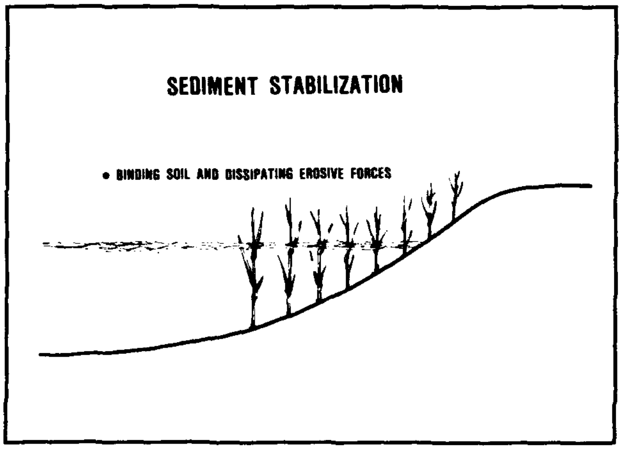


Figure 4. Schematic representation of sediment stabilization by a wetland area

Shoreline Anchoring. Wetland plants, like all plants, bind soil with their root systems. Many areas may persist longer due to the anchoring function of wetland vegetation, and in erosional environments the necessity of constructing costly protective structures may be reduced if wetland vegetation is present (Allen 1979/*L,R). For example, unplanted shores exposed to the same incoming wave intensities as cordgrass-planted shores can erode 4 times faster than vegetated shores (Benner et al. 1982/NC:Eem). Seasonal erosional effects (of ice, waves) on a shoreline can be less pronounced where a salt marsh is well established (Richards 1978/NY:Eem). Wetlands wider than 30 feet can reduce wave energy by 88 percent (Knutson et al. 1982/VA:Eem).

Dissipation of Erosive Forces. Wetlands reduce wave and current energies, as would almost any shallow, undeveloped space adjacent to uplands. Researchers do not agree on how effective wave dissipation of vegetated wetlands is compared with nonvegetated wetlands or adjacent uplands. Nor is there agreement that wetlands in general reduce wave and current velocities sufficiently to prevent damage to other resources. Prevention of damage depends largely on the magnitude of incoming waves or currents and the sensitivity of the resources. If wetlands are established and anchored with extensive, persistent (especially woody) vegetation, they may grow outward into a bay, channel, or delta and provide temporary protection from waves or currents associated with infrequent, large storms that would otherwise penetrate deep into adjacent areas.

Few studies describe the degree to which various vegetation types dissipate waves. Wayne (1976/Eem,ab) found that emergent plants reduced the height and energy of small waves by 71 and 92 percent, respectively; rooted vascular aquatic-bed plants decreased wave height and energy by 42 and 66 percent, respectively. A formula for calculating the expected percentage reduction in wave height was given by Dean (1979/*), using field measurements of stem density, spacing of emergent or woody plants, water depth, width of the vegetated wetland, and incident wave height (the last factor can be estimated indirectly by measuring the wetland's fetch, see US Army Corps of Engineers 1977).

2.3.2 Processes

The principal processes affecting this function are as follows:

- Energy associated with erosive forces.
- Frictional resistance offered by the wetland.
- Position of the wetland relative to the upland and incoming erosive forces.
- Ability of wetland plants to anchor the soil.
- Erodibility of uplands being protected.

These are discussed briefly below. More detailed documentation for each is presented in Chapter 3.

Energy Associated with Erosive Forces. Incoming waves, currents, water-level fluctuations, ice, and ground water discharge are capable of creating erosive conditions. In general, the potential ability of water to displace and transport soil particles increases with increasing flow velocity.(Figure 5). The major factors that influence the energy of incoming waves or currents are wind velocity, fetch, wetland gradient and bottom roughness, presence of man-made protective structures, tides, waves from boats, and run-off. Erosiveness depends on the frequency, period, amplitude, and seasonal timing of water-level fluctuations. The erosiveness of ice increases with its duration, instability (e.g., frequent freeze-thaw), and depth of penetration.

The erosive energy of water also depends on the suspended solids concentration of the water. Relatively sediment-free waters are more erosive than those with a heavy sediment load (Nunnally and Keller 1979:35,43/*R) because suspended sediment tends to reduce turbulence, which is responsible for transporting sediment (Ritter 1986:211/*). Relatively sediment-free waters are

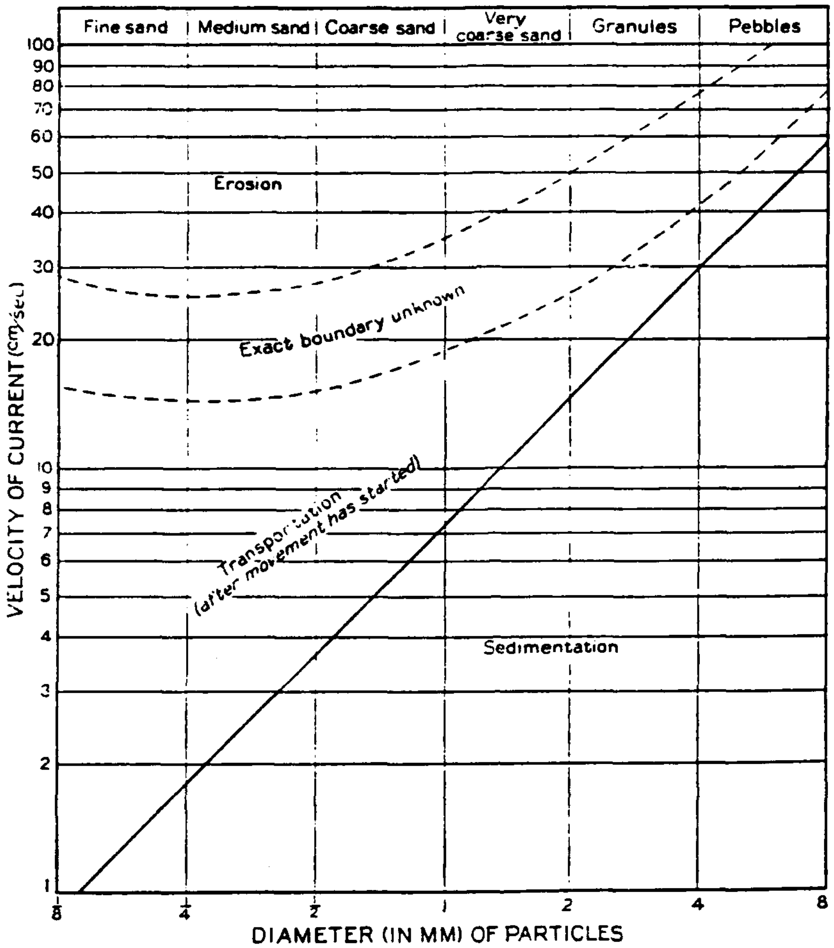


Figure 5. Erosion, transportation, and sedimentation of well-sorted sediment in relation to current velocity and particle size (modified from Rubey 1938)

likely to occur in some wetlands with substantial ground water input, below dams, or when upstream wetlands significantly alter the sediment load equilibrium. Conversely, if the wetland is located in a depositional environment, such as near a river delta or along a channel where adjoining waters are sediment laden, then shorelines are less likely to experience net erosion, and may even accrete, depending on other factors such as wave and current energies. The energy associated with ground water discharge increases with discharge rate and spatial extent.

Frictional Resistance Offered by the Wetland. The frictional drag on incoming waves or currents increases with the width of the wetland, the density of its vegetation or other obstructions (e.g., boulders, logs, hummocks), the rigidity of these obstructions, and the degree to which they extend above the incoming waves or currents. Wetlands with persistent (year-round) or perennial vegetation are more effective on a net annual basis than are wetlands with nonpersistent vegetation or those dominated by annuals.

Position of the Wetland. Wetlands that project outward into open water, due to accretion and expansive anchoring by rooted vegetation, are better able to intercept and deflect waves that otherwise might directly impinge upon adjacent shorelines. However, such wetlands exist because their environment is basically depositional, and seldom are exposed to such waves. Thus, this function may be realized only during storms approaching from atypical directions or of unusual magnitude. Moreover, if wetlands project too far into a channel, flows may be gradually constricted, with a subsequent reordering of the erosion-deposition balance (Zimmerman et al. 1967).

Ability of Wetland Plants to Anchor the Soil. The depth and degree of root branching influence the ability of plants to anchor soil. Sod-forming ability is species specific (Garbisch 1977/*). Even if plants are not particularly effective at producing frictional drag for incoming waters because they are not rigid or they are fully submersed, some, such as the mosses, may still help anchor the sediments and prevent scour (Pfankuch 1975/*R).

Erodibility of Uplands Being Protected. The intrinsic erodibility of shorelines is greater with increased sand and silt content of the soil (Dunne and Leopold 1978:527), bank slope, ice action, ground water seepage, run-off from farther upslope, and nearness of heavy objects to the top of the bank slope. Often, much of the erosion along channel banks occurs as bank failure following a rapid decrease in water level (Nunnally and Keller 1979/*R). Bank failure results from the weight and pressure of water left in the banks after the stream recedes (Nunnally and Keller 1979:48/*R; Ritter 1986:218/*).

2.3.3 Interactions With Other Wetland Functions

Ground Water Recharge, Ground Water Discharge. Eroded fine sediments may be redeposited in portions of a wetland where ground water exchange occurs, and eventually slow the rate of ground water recharge or discharge. Shoreline anchoring by wetlands may minimize such erosion.

Floodflow Alteration. Large waves may themselves cause flooding. Erosion may enlarge the storage capacity of upcurrent wetlands, but redeposition can shrink the capacity of downcurrent areas, perhaps including flood storage reservoirs.

Sediment/Toxicant Retention. This function is closely interrelated with shoreline anchoring. Dampening of wave and current energies leads to deposition of sediment, and subsequent anchoring of this sediment by plant roots ensures longer retention.

Nutrient Removal/Transformation. One process critical to nutrient removal is the long-term burial of nutrients in sediments, a process that may be facilitated by plants through sediment trapping. At the same time, other plants can translocate nutrients from the bottom sediments to the water column. Also, by dissipating oxygenating waves and currents and by contributing decaying organic matter, wetlands influence the redox potential which partly regulates nutrient removal/transformation.

Production Export. Wetland plants associated with shoreline anchoring might reduce flushing capacity by inhibiting circulation. They may also tie up nutrients in their root systems for indefinite periods, or release nutrients into the water column. Through the process of enlarging outlet constrictions, shoreline erosion may enhance the flushing capacity of a wetland and, consequently, its export capability. However, if these exported sediments are redeposited in constricted areas nearby, flushing capacity and nutrient-export efficiency may be reduced.

Aquatic Diversity/Abundance, Wildlife Diversity/Abundance. Suspended sediment from eroded shorelines may adversely impact growth and survival of aquatic organisms and submersed plants, reducing habitat suitability for waterfowl and wading birds. Erosion may also directly threaten streamside wildlife habitats. However, moderate erosion may create shelter for fish (e.g., undercut banks) and augment bank habitats for wildlife such as kingfishers, bank swallows, muskrat, and beaver. If eroded sediments are redeposited over a wider area, they may provide a substrate for the establishment of a more productive plant community.

Recreation, Uniqueness/Heritage. Shoreline erosion is seldom aesthetically appealing, and erosion of beaches may halt associated recreation. Erosion may also threaten areas considered unique or important to our heritage.

2.3.4 Direct Economic Significance

Millions of dollars are spent annually for construction of jetties, bulkheads, and other structures intended to inhibit shoreline erosion by waves and currents. Such erosion may destroy dwellings, eliminate harvestable timber and peat, remove fertile soil, and alter local land uses. Costly dredging is required when eroded sediments that are redeposited in channels and reservoirs impede navigation and reduce the capacity of flood-control basins.

2.4 Sediment/Toxicant Retention

2.4.1 Definitions

Sediment/toxicant retention (Figure 6) is the process by which suspended solids and chemical contaminants such as pesticides and heavy metals adsorbed to them are retained and deposited within a wetland. Deposition of sediments can ultimately lead to removal of toxicants through burial, chemical breakdown, or temporary assimilation into plant tissues (Boto and Patrick 1979/*). Sediment/toxicant retention may involve retention of run-off-borne contaminants before they move into the deep water of a wetland or ground water aquifers, as well as interception and retention of sediment borne by surface waters before it is carried downstream or offshore.

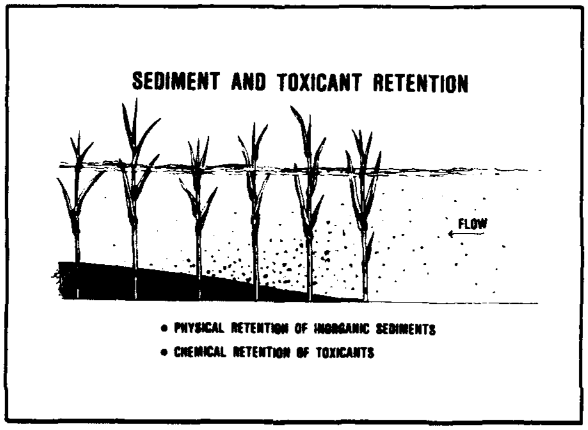


Figure 6. Schematic representation of sediment/toxicant retention in a wetland

Toxicants, as addressed in this document, include heavy metals, pesticides, and other potentially toxic organics. They are included in discussions of the sediment retention function for two reasons. First, sediment deposition is used as a surrogate for toxicants because many, but not all, toxicants adsorb to suspended or deposited sediment, particularly its associated organic matter (Hunt and Lee 1976/*, Odum and Drifmeyer 1978/*, Sharom et al. 1980, Martin and Hartman 1987/ND,SD) and its clay fraction (Richardson and Epstein 1971, Pionke and Chesters 1973, Karickhoff et al. 1979, Brown 1985). Results of a nationwide river and lake sampling program, for example, indicated that almost 20 percent of the sediment samples had one or more pesticides present (Gillion 1985/US). Second, no better surrogates of toxicant retention potential are as easily estimable as fine sediment accretion.

Most vegetated wetlands are excellent sediment traps, at least in the short term. Few wetlands export more inorganic sediment than they import.

Turbidity may be greater downstream from some wetlands, but this is usually due to algal productivity and export of suspended organic substances.

The length of time sediments and toxicants are retained depends on the hydrologic and chemical characteristics of the specific wetland. Sediment retention times are least in riverine wetlands. Moderate frequency floods, occurring at periods of one to several years, are responsible for most erosion of riverine wetland sediments (Meade 1982/US:R). Contaminated sediments can be resuspended over decades, posing a chronic threat to aquatic life (Marron 1987/SD:R).

Wetland vegetation can help trap and retain suspended sediment by anchoring the shoreline, reducing resuspension of bottom muds by wind mixing, increasing the length of the flow path, contributing organic matter, and slowing water velocities (Jackson and Starrett 1959/IL:L; DeLaune et al. 1978/LA:Eem; Richards 1978/NY:Eem; Boto and Patrick 1979/*; Phillips 1980/*Eab; Kenworthy et al. 1982/NC:Eab; Morris and Paulson 1982/NV:Rem; Johnston et al. 1984/WI:Lem; Lambou 1985/LA:fo) (see also Section 2.3, Sediment Stabilization). Even aquatic bed vegetation, which typically provides less resistance to water flow than emergent or woody vegetation, may reduce wave or current energies sufficiently to induce sediment settling (Fonseca et al. 1982/NC:Eab; Bulthuis et al. 1984; Short and Short 1984/FL:E; Ward et al. 1984/E).

The sediment-trapping ability of wetlands will decline if they fill in or if flooding kills their vegetation, especially in tidal waters, very windy wetlands, and other dynamic environments. Physical processes (e.g., gradient, tidal factors) in some cases may be more important for trapping sediment than the actual presence of wetland plants (Gersberg et al. 1986/CA:Pem).

Compared with other landscape components, wetlands can be disproportionately important for sediment retention. Over 20 percent of the permanent sediment deposition in one North Carolina watershed occurred in forested wetlands occupying only 11 percent of the area; a nearby watershed with one-sixth the wetland acreage (but a similar amount of cultivated land) had much less deposition (Cooper et al. 1986/NC:fo). A simulation study of one Mississippi watershed indicated that loss of half the watershed's bottomland hardwood wetlands would double the sediment loss (Molinas et al. 1988/MS:fo).

Regression analyses of many watersheds in northern Wisconsin indicated that sediment loads were approximately 90 percent lower in watersheds with 40 percent wetland and lake acreage than in watersheds with none (Hindall 1975/WI). Nearly 70 percent of the sediment was retained by 5 percent of the wetlands. Similar regression studies in Minnesota by Oberts (1981) and Brown (1988/MN:L) indicated that maintaining about 10 percent of a watershed in wetlands optimized sediment retention; maintaining larger acreages yielded a detectable but minimal additional improvement. Watersheds with a large proportionate acreage as ponds and reservoirs, and with upland soils containing little clay, have low concentrations of riverine suspended sediment (Simmons 1976).

Sediment retention in wetlands averages about 30 percent of the total entering, with a maximum retention of about 95 percent (e.g., Dendy 1974; Hickok et al. 1977/MN:P; Rausch and Schreiber 1977; Fetter et al. 1978/WI:Pem; Novitzki 1978/WI:P; MacCrimmon 1980/ONT:P; Stumpf 1983/DE:Eem; Brown 1985; Striegl 1987; Wolaver et al. 1988/SC:Eem).

A different expression of the sediment trapping capacity of wetlands is their vertical accretion rates. Marine shorelines accrete at a worldwide average of about 0.02 inch/year (Rusnak 1967/*E), and estuarine shorelines at about 0.08 to 0.16 inch/year (Shepard and Moore 1960/*TX), although Richards (1978/NY:Eem) reported a low marsh and an estuarine mud flat accreting at 1.5 and 1.8 inch/year, respectively. Riverine-associated palustrine wetlands usually have accretion rates between 0.3 and 1.1 inch/year, with reported rates ranging as high as 3.3 inches/year and as low as 0.04 inch/year (Eckblad et al. 1977; Bridge and Leeder 1979/*R; Mitsch et al. 1979/IL:fo; Nanson 1980/BC:R; Cooper et al. 1986/NC:fo). Bogs, which probably trap sediments for longer periods, accrete roughly 0.04 inch/year (Walker 1970, Kadlec and Hammer 1988/MI:P). Highest rates are reported from human-altered environments. For example, Turk et al. (1980/NS:E) reported sediment accumulations of 66 inches/year for an estuarine mud flat next to a causeway. Wetlands receiving stormwater for treatment accreted 0.78 inch/year (Striegl 1987).

In summary, most studies of sediment retention by wetlands have technical drawbacks. Commonly, such studies fail to document: (1) the extent to which sediment is exported from the wetland during severe, infrequent (e.g., 25-year probability) storms, (2) the degree to which sediment retention can be attributed to the wetland itself, rather than to the depression in which it is situated, and (3) the extent of long-term biological changes induced by retention or catastrophic export.

Due to accretion and other processes, many wetlands effectively retain heavy metals, preventing or delaying their transport into adjoining water bodies, aquifers, and food chains (Wigington et al. 1986, Striegl 1987). Retention of some metals in wetlands can essentially be permanent if they become organically bound in complexes with peat (Wieder and Lang 1982). This seems particularly likely to occur in isolated palustrine wetlands in cool climates. However, metal retention shows considerable variation among wetlands.

Metals may be mobilized and made available to food chains if (1) there are changes in loading rates (Laxen and Harrison 1977/*), (2) sediments and associated organic matter are resuspended by wind, waves, erosion, or disturbance from invertebrates, fish, and birds; (3) sediment organic matter becomes aerated as a result of altered hydroperiods, plant community type change, or wind mixing (van Hassel et al. 1980), or (4) sediment-bound metals are chelated as the result of organic matter added by wetland plants. Bioaccumulation is important to the retentive function of wetlands because if most of a wetland's fauna is migratory or its detritus is subject to widespread dispersal, bioaccumulation may be a major dilution mechanism for contaminants, whereas if the fauna is basically resident and highly bioaccumulative (e.g., some filter-feeding invertebrates), retention within the wetland as a whole may be increased over what might occur strictly from physicochemical processes in the sediment and water. Accumulation and uptake of metals by plants is generally greatest in roots and litter (Gallagher and Kibby 1980/OR:Eem), but sediment processes are often of greater long-term importance than plant uptake. Accumulation in animals is discussed by Olsen (1984/*), Reynoldson (1987), and Willford et al. (1987). The relative capacities of various wetland plant and animal species for retaining toxic metals have not been adequately quantified.

Lead and mercury normally are retained longer in wetland sediments than are zinc and perhaps chromium and copper (Teal et al. 1982/MA:Eem). However, cadmium and mercury can easily be mobilized from sediments, either by physicochemical disturbance or accumulation by mobile sediment organisms, and probably few regularly flooded wetlands are long-term sinks for these elements (Nixon 1980/*; Teal et al. 1982/MA:Eem; Simpson et al. 1983/NJ:tem). Some tidal freshwater wetlands appear capable of retaining moderate loadings of copper for long periods, as well as nickel, lead, zinc, and perhaps chromium (Simpson et al. 1983/NJ:tem; Dubinski et al. 1986/NJ:Ptem). For copper, algae appear to influence retention more than the presence of adsorbing sediments; the converse may be true of cadmium and zinc (Hart 1982/*).

Organic matter associated with wetlands generally adsorbs metals more effectively than does clay sediment (Hart 1982/*). Mercury concentrations in sediment of tidal (Teal et al. 1982/MA:Eem) and noncontiguous wetlands are strongly and positively correlated with organic matter, but in riverine wetlands other factors are more influential (Martin and Hartman 1987/ND,SD). Cadmium is also associated with sediment organic matter (Martin and Hartman 1987/ND,SD), although to a lesser degree than mercury, copper, and lead (Hart 1982). Cadmium concentration is negatively correlated with sediment sand content (Martin and Hartman 1987/ND,SD).

In other instances (e.g., Beck et al. 1974/US:R), wetlands appear to assist mobilization of some metals. Sulfide concentrations appear to be a key controlling factor, at least in tidal marshes (Teal et al. 1982/MA:Eem).

Wetlands may also be effective for retaining and detoxifying some pesticides and other hazardous organics. Several studies (Guenzi and Beard 1968, Ko and Lockwood 1968, Guenzi et al. 1971, Spencer et al. 1974, Parr and Smith 1976) have shown that organochloride pesticides are readily degraded in anaerobic, sulfide-rich, fine-sediment, sunlight-saturated, organic environments. Such environments typify many wetlands, and tidal wetlands in particular (Seidel 1976/*, Wolverton and McKown 1976, Renwick and Ashley 1984, Giblin 1985/MA:Eem).

Degradation is also enhanced by the proliferation of algae and microbes (Hart 1982) that, due to the abundance of organic matter and complexity of vegetation substrates, are often present at consistently higher densities in wetlands. However, the high levels of humic materials in some wetlands may depress microbial activity. Also, the high concentrations of dissolved organic matter can mobilize a few of the most insoluble toxicants, such as polychlorinated biphenyls, and the ready availability of natural organic material can divert microbial attention from synthetics that might otherwise be broken down faster (Smith et al. 1988/*).

Bioaccumulation (the uptake of contaminants by biota from water or food) also occurs disproportionately in wetlands (e.g., Birmingham and Colman 1983, Biddinger and Gloss 1984/*, Ohlendorf et al. 1986). However, the existence of true biomagnification (the accumulation of contaminants through trophic transfer) involving currently used pesticides and other hazardous organics is debatable (see contrasting viewpoints of Reynoldson 1987 and Smith et al. 1988/*).

Bioaccumulation occurs when organisms with high tissue lipid content are exposed to relatively insoluble, stable organics (Smith et al. 1988/*), or when heavy metals become concentrated in areas of low alkalinity and limited organic matter, which could otherwise offset their effect (e.g., Hargeby and Petersen 1988). In sandy estuarine sediments exposed to heavy metal pollution, total organic matter in the range 0.7 to 1.0 percent supported the most diverse benthic invertebrate community, which presumably reflects the lowest toxicity (Franz and Harris 1988/NY:E).

Among potentially toxic organics, the chlorinated phenols, chlorinated benzenes, polycyclic aromatic hydrocarbons, and phthalate esters appear most capable of bioaccumulating, especially in the organic-rich sediments of wetlands; most herbicides and carbamate and organophosphorus insecticides do not bioaccumulate (Smith et al. 1988/*) but can impact biota if applied directly or chronically near wetlands. Acute toxicity is generally greatest for the organophosphorus insecticides and anilide herbicides. Chronic toxicity is greatest for organochloride insecticides and triazine herbicides (Maas et al. 1984/*).

2.4.2 Processes

Major processes or factors that affect this function are the following:

- Amount of incoming sediment.
- Particle size and density of suspended sediment.
- Difference in energy levels of suspending forces within the wetland versus upcurrent areas.
- Vertical layering caused by salinity and temperature in waters bearing the sediment.
- Flocculation, agglomeration, and precipitation.
- Bioturbation and mobilization.
- Storage capacity of the wetland.

Each of these is discussed briefly below.

Amount of Incoming Sediment. Sediment delivery usually increases with increased drainage area size, acreage of cleared land, precipitation, and lack of soil management measures (Karr and Schlosser 1978/IL:R, Cooper et al. 1986/NC:fo). Important soil characteristics that relate to erodibility include soil particle size, aggregate stability, permeability, water-holding capacity, and infiltration rate. Sedimentation is more likely to occur when waters are carrying a heavy sediment load. Conversely, soil particles generally go into suspension more readily when their contiguous waters are relatively sediment-free (see Section 2.3.2).

Particle Size and Density of Suspended Sediment. Heavier particles naturally settle faster than lighter particles of the same size. It is for this reason that inorganic particles usually settle faster than organic particles (Boto and Patrick 1979/*). The particle size distribution of suspended sediments is also important to the sedimentation rate, with colloidal particles such as clays having the slowest sedimentation rate (Boto and Patrick 1979/*).

Difference in Energy Levels. Sediment deposition (e.g., shoaling) occurs where water velocity rapidly slows, usually as a result of increased cross-sectional area or bottom roughness. Sediment deposition is also enhanced by channel obstructions and other factors, such as wetland vegetation, which reduce wave or current velocity or extend the overland flow path. Sediment retention in wetlands without outlets approaches 100 percent because flow is totally stopped. In general, increased residence time results in more effective sediment removal.

Vertical Layering Caused by Salinity and/or Temperature. Intense stratification associated with salinity or temperature-caused differences in water density may counteract the natural settling tendencies of sediment by restricting vertical mixing across the layers (Schubel and Carter 1984). This factor is generally of more concern where flow is constricted or impeded and where the effects of tidal asymmetry (i.e., differing energy levels between incoming and outgoing tides) are prominent.

Flocculation, Agglomeration, and Precipitation. Sharp interfaces between freshwater and saltwater cause fine sediments to flocculate and settle, especially where flow is very slow. Flocculation affects only certain types of clays, and is suspected to be maximized at or around 5 ppt salinity (Rochford 1953/AU:E). Kranck (1984/E) reported that flocculation is an essential process for sediment trapping in estuaries, although Boto and Patrick (1979) questioned the significance of this process. Flocculation and agglomeration processes occur following chance collisions among particles, which occur most frequently within the turbidity maximum (Kranck 1984/E). Organic particles also promote settling by adhering to inorganic particles (Kranck 1984/E). Filter-feeding organisms may also repackage particulate matter into fecal pellets, which may account for up to 40 percent of particle transfer to the bottom of some coastal waters (Kranck 1984/E). Another process that may contribute to sediment trapping in some areas is the formation of carbonate-sediment complexes with subsequent precipitation as marl.

Bioturbation and Mobilization. Sediments and/or their associated toxicants can be measurably resuspended as a result of the activity of bottom-feeding fishes (e.g., carp), aquatic invertebrates, and birds. This is termed bioturbation. Contaminants associated with deposited sediment can be reintroduced into the water column either by this process or by uptake by aquatic plants and subsequent release during decay. Bioturbation may so significantly alter the substrate that soil drainage characteristics and plant growth are altered (Sharma et al. 1987/SC:Eem).

Storage Capacity of the Wetland. A wetland's sediment trapping efficiency depends on its depth, volume, and surface area (Schubel and Carter 1984). Where sediment deposition occurs, some wetlands gradually fill in. As the wetland surface is built up by sediment deposition, flooding frequency decreases, resulting in decreased sediment input (Mitsch and Gosselink 1986:167/*). However, many wetlands may become filled only over periods of hundreds of years. Others may be periodically flushed of stored sediment by large storms, or by herbivore activity and long-term precipitation cycles (Mitsch and Gosselink 1986:169/*).

2.4.3 Interactions With Other Functions

Ground Water Recharge, Ground Water Discharge. Trapping of fine sediments is often incompatible with these functions if they occur in the same wetland. Most ground water exchange occurs either in very shallow or very deep portions of the wetland. Reduced mixing in deepwater areas and the influence of vegetation in shallow water also make these locations favorable for sediment retention. Trapped fine sediments may slow the rate of ground water recharge or discharge. However, recharge is more likely to occur at higher elevations where usually coarser, more permeable sediments and smaller total quantities of sediment are deposited (Maxey 1968/NV, Winter 1977/:L). Potential leaching of toxicants into an aquifer via ground water recharge would also be undesirable.

Floodflow Alteration. Sediment trapping is sometimes incompatible with this function. Unless compacted, dredged, or flushed out by peak flows, trapped sediment reduces the capacity of flood storage depressions. However, sediment deposited and compacted along river banks may, under some circumstances, form natural levees that can reduce local flood damage.

Sediment Stabilization. Sediment retention usually enhances this function. Some sediment trapping usually occurs before plants become established and contribute to sediment stabilization. Sediment deposited as offshore bars may also dissipate waves and currents. However, wetlands might, by filtering out sediment, occasionally contribute to increased downstream erosion by increasing the sediment-carrying capacity of the water (Turner 1980/*L).

Nutrient Removal/Transformation. Sediment retention plays a significant role in nutrient removal/transformation through the retention of organic sediments and nutrients adsorbed to sediment particles. Sediment retention also contributes to nutrient removal through burial of nutrients in the substrate. Extreme levels of contaminants can reduce nutrient cycling rates, in part by inducing anoxia and depressing bioturbation.

Production Export. Areas that retain sediments generally retain particulate aquatic food substances and toxicants as well. Thus, retention of sediment and toxicants in one wetland may alter the fauna immediately downstream. This phenomenon is commonly observed below dams.

Aquatic Diversity/Abundance, Wildlife Diversity/Abundance. Sediment deposition in one wetland may benefit aquatic flora and fauna in another via reduced turbidity and increased photosynthesis. However, some sediment export may be essential to maintaining suitable substrate and nutrient conditions in downcurrent wetlands. In wetlands where sediment and toxicant buildup is occurring, excessive quantities of sediment may alter the plant community or smother aquatic vegetation and kill invertebrates and fish larvae, or at least modify their habitat (Cooper 1987/MS:R). The capacity of wetlands for bioaccumulation is discussed above in Section 2.4.2. Wetland communities exposed to sediment or toxicant run-off do not inevitably experience bioaccumulation or altered community structure (Chutter 1969, Hansen 1971/IA:R, Barton 1977/ONT:R, Piest and Sowls 1985/AZ:P, Warwick 1988), and basic ecological processes are not invariably altered (Schindler 1987/MAN:L).

Recreation. Sediment/toxicant retention usually conflicts mildly with recreation when the two occur in the same wetland. However, sediment/toxicant retention in one wetland would benefit recreation in downcurrent wetlands.

Uniqueness/Heritage. The effect of sediment/toxicant retention on this function depends on the particular resource of concern.

2.4.4 Direct Economic Significance

Sediments or toxicants deposited in undesirable locations often require increased expenditures for dredging and water treatment. Retention of sediments and toxicants in wetlands would decrease the need for such expenditures.

2.5 Nutrient Removal/Transformation

2.5.1 Definition

Nutrient removal/transformation includes the storage of nutrients within the sediment or plant substrate; the transformation of inorganic nutrients to their organic forms; and the transformation and subsequent removal of one nutrient (nitrogen) as a gas. Nutrient removal/transformation involves trapping of nutrients before they reach deep water, are carried downstream, or are transported to underlying aquifers. Particular attention is focused on processes involving nitrogen and phosphorus, as these nutrients are usually of greatest importance to wetland systems (Kadlec and Kadlec 1979/*). Thus, throughout this document, reference to "nutrients" applies specifically to nitrogen and phosphorus.

There are divergent opinions regarding the role of wetland plants in water quality. Wetlands have been called nutrient sinks, nutrient sources, and pass-through systems. Wetlands are at least temporary sinks for nitrogen and phosphorus, under both natural and nutrient-enriched conditions (Toth 1972; Valiela et al. 1973/Eem, 1975/MA:Eem; Klopatek 1975/WI:Pem, 1978; Broome et al. 1975/NC:Eem; Nedwell 1975/Efo; Axelrod et al. 1976/LA:Eem; de Jong 1976; Greij 1976/MI:Pem, McNabb 1976/*ab; Patrick et al. 1976/LA:E; Spangler et al. 1976/WI:R,P; Wolverton et al. 1976/MI:P; Hickok et al. 1977/MN:P; Mitsch et al. 1977/IL:fo; Rausch and Schreiber 1977; Dolan et al. 1978/FL:Pem; Fetter et al. 1978/WI:Pem; Fritz and Helle 1978/FL:Pfo; Loucks and Watson 1978/WI:L; Novitzki 1978/WI:P; Prentki et al. 1978/WI:L; Simpson et al. 1978/*; Mudroch and Capobianco 1979; Turner et al. 1979/GA:E; Mulholland 1981/NC:fo; Nichols 1983/*; Nixon and Lee 1986:221/*).

In freshwater environments, up to 91 percent of the phosphorus (Mitsch et al. 1977/IL:fo) and 94 percent of the nitrate nitrogen (Brinson et al. 1981b/NC:fo) may be retained. Kuenzler and Craig (1986/NC,VA) found that forested wetlands removed 37 and 2 kg/ha/year of N and P, respectively. Wetlands as disparate as New England salt marshes (Valiela et al. 1976/MA:Eem), southeastern salt marshes (Wolaver and Spurrier 1988b/SC:Eem), Arizona riverine wetlands (Grimm 1987/AZ:R), and southeastern bottomland hardwood wetlands (Yarbro 1979/NC:fo) reduced inflowing levels of inorganic nutrients. Freshwater systems as a whole are more effective for nutrient removal than are estuarine or marine systems, partly because the latter have less capacity for retaining carbon in sufficient amounts for supporting nutrient removal processes (Nixon 1988/*L,E,M).

Region-wide regression studies also indicate that surface-water concentrations of nitrate, and to a lesser extent phosphorus, are less in watersheds having a large proportion of wetland acreage, especially in urban watersheds (Oberts 1981). In an Iowa agricultural watershed, the presence of 2,200 ha of wetlands was estimated to reduce annual nitrate loading by 4,000 kg (Jones et al. 1976/IA:Eem).

However, Mitsch and Gosselink (1986:125/*) state that "there is a growing consensus that not all wetlands are nutrient sinks nor are the patterns consistent from season to season or year to year." Terrestrial soils, because of their generally higher cation concentrations, are probably more efficient in removing and retaining phosphorus in runoff (Richardson 1985/P,R, Jordan et al. 1986/MD) and perhaps in removing nitrogen (Ehrenfeld 1987/NJ:P). Instances have been reported of summertime dissolved-phosphorus loadings from lacustrine wetlands exceeding loadings from other parts of a landscape (Carpenter 1980). Forested wetlands, when enriched, were only mildly effective in retaining added phosphorus (Brinson et al. 1984/NC:fo).

Still, the strategic position of wetlands on the landscape, coupled with the ability of wetland vascular plants (and their attached algal and microbial communities) to remove nutrients from waters and sediments during the growing season and release them later when light or temperatures will not support profuse algal growth (Smith and Horne 1988/CA:Eab), seems to be a general phenomenon, and one important in maintaining the water quality of adjoining systems (Toth 1972; Lee et al. 1975/*; Mason and Bryant 1975; Spangler et al. 1976/WI:R,P; Kibby 1978/*; Post and de la Cruz 1978; Dawson 1980; Howard-Williams 1981/*Lab; Nichols 1983; Simpson et al. 1983/NJ:tem; Johnston et al. 1984/WI:Lem; Lowrance et al. 1984/GA:fo; Reddy and Sutton 1984/FL:em; Reddy et al. 1984/FL; Heckman 1986/EU:Rem).

Huge or long-term applications of nutrients may not be assimilated without severely altering wetland vegetation, polluting downstream areas, or being associated with bioaccumulation of toxicants in food chains (e.g., Steward and Ornes 1975/FL; Valiela et al. 1975/MA:Eem; de Jong 1976; Ewel 1976/FL:Pfo; Whigham and Simpson 1976/NJ:tem; Zoltec et al. 1979/FL:Pem; Kuenzler et al. 1980/NC:fo; Schwartz and Gruendling 1985/VT:Lem; Nixon and Lee 1986/*). However, this outcome does not seem *inevitable* (e.g., Whigham and Simpson 1976/NJ:tem; Piest and Sowls 1985/AZ:P, Sanville 1988/AK:P). Nitrogen saturation is probably less a problem than phosphorus saturation (Boyt et al. 1976/FL:P, DeBusk and Reddy 1987/FL:Pfo).

Wetlands can retain and subsequently release nutrients in many ways. As noted above, nutrients can be taken up and stored by wetland vegetation on a short-term basis. On a long-term basis, wetland vegetation may effectively remove nutrients from biological cycling by assisting burial below the zone of biological activity, usually to depths greater than 1 meter (Prentki et al. 1978/WI:L, Fisher et al. 1982b/NC:E). The content of amorphous, extractable aluminum in a wetland's sediments is one of the most important factors influencing the ability of a wetland to assimilate phosphorus over long periods (Richardson 1985/P,R).

Nitrogen nutrients can be permanently removed through conversion to gaseous form by denitrification and ammonia volatilization, and seasonally removed as emerging insect biomass. These processes are much more prevalent in wetlands than in uplands. This is because anaerobic, organic-rich soils (which typify wetlands) favor these processes. The same is true of deeply rooted plants in forested wetlands which can effectively use several forms of nitrogen for the denitrification process (Ehrenfeld 1987/NJ:P). However, some wetlands also contribute to the pool of available nitrogen through nitrogen fixation, especially when ratios of nitrogen to phosphorus are less than 10.

2.5.2 Processes

Assessing a wetland's efficiency for nutrient removal/transformation is complicated by the fact that nutrients are involved in both biotic and abiotic cycles, and are continuously being exchanged between the substrate, water, vegetation, and atmosphere (for nitrogen and carbon). Nongaseous nitrogen most commonly exists in five forms: particulate organic nitrogen, dissolved organic nitrogen, ammonia, nitrate, and nitrite. Phosphorus exists in any of three forms: particulate organic phosphorus, dissolved organic phosphorus, and phosphate.

Major processes or factors that affect nutrient removal/transformation are

- Biological uptake and processing.
- Sedimentation and accumulation of organic matter in the substrate.
- Adsorption and nutrient interactions with sediments.
- Chemical and microbial processes including denitrification, nitrogen fixation, and ammonia volatilization.

All of these processes are enhanced, to varying degrees, by the increased retention times commonly associated with wetlands.

Biological Uptake and Processing. As described above, wetland vascular plant communities can significantly regulate nutrient concentrations in adjoining waters, at least seasonally. Free-floating vascular plants (as well as phytoplankton and attached algae) take up nutrients directly and exclusively from the water and thus directly influence nutrient concentrations within the water column. Rooted submersed species derive their nutrients from the sediments (Barko and Smart 1980) and directly from the water (Steward and Ornes 1975/FL, Boyt et al. 1976/FL:P, McNabb 1976/*ab, Klopatek 1978/*Rem, Thayer et al. 1984/*Eab). Recent evidence also indicates that they may shift between uptake from the water and sediments depending upon supply (Boynton et al. 1980/MD:E, Davis and Brinson 1980/*ab, Day et al. 1981). However, under natural conditions, the sediments are usually the primary source of nutrients for most emergent and submersed species (Banoub 1975/EU:Lem; Klopatek 1975, 1978/WI:Pem; Brinson and Davis 1976/NC:R; Kibby 1978; Prentki et al. 1978/WI:L; Barko and Smart 1980; Hopkinson and Schubauer 1984/LA:tem).

The water quality impact of species that remove nutrients only from the sediments may not be immediate (Klopatek 1975/WI:Pem, Boyt et al. 1976/FL:P, Day et al. 1980/LA:fo), but such uptake may encourage a reverse movement of nutrients from the water to underlying sediments to maintain equilibrium between sediments and the water (Klopatek 1978/*Rem). Conversely, uptake from the water may upset the equilibrium, causing release of nutrients from the sediments (Farnworth et al. 1979/*).

Plants with persistent, woody tissues retain for long periods more nutrients than they lose through leaf-drop, decay, or leaching (e.g., Lowrance et al. 1984/GA:fo). Such species conserve nutrients by translocating them to persistent tissues prior to leaf fall (Klopatek 1975/WI:Pem, Prentki et al. 1978/WI:L, Simpson et al. 1978/*, van der Valk et al. 1979/*, Morris 1982/MA:Eem, Kistritz et al. 1983/BC:Eem). However, woody plants generally are less effective than emergent and aquatic bed vegetation at taking up nutrients (Richardson et al. 1978/MI:Pem, Ehrenfeld 1987/NJ:P) and, despite translocation, much is lost with leaf fall.

With enrichment, vegetation may retain additional nutrients by increasing biomass. Primary production often increases with enrichment, especially in low-alkalinity or oligotrophic systems (Broome et al. 1975/NC:Eem; Hartland-Rowe and Wright 1975/*; Valiela et al. 1975/MA:Eem; Boyt et al. 1976/FL:P; Dolan et al. 1978/FL:Pem; Mudroch and Capobianco 1979; DeLaune and Patrick 1980/LA:Eem; Sanville 1988/AK:P). The ability of plants to accumulate nutrients depends on their life stage at the time nutrients are most plentiful (Boyd 1978/*; Kadlec and Kadlec 1979/*; Heliotis and DeWitt 1983/MI:P; Kistritz et al. 1983/BC:Eem; Bernard et al. 1988/*L,Pem). Thus, although standing crop is approximately related to nutrient content (Peverly 1985/NY, Bernard et al. 1988/*L,Pem), maximum nutrient accumulations do not necessarily occur when biomass is at its seasonal peak (Klopatek 1978/*Rem; Kistritz et al. 1983/BC:Eem). Also, excessive accumulations of peat may inhibit biomass accumulation by existing plants and prevent establishment of new plants, thus reducing the nutrient retentive ability of a wetland (Wetzel 1979, Barko and Smart 1983/L, Day et al. 1988, Kadlec and Hammer 1988/MI:P). Such accumulations may result from increased nutrient additions or altered hydrology (Day et al. 1988, Kadlec and Hammer 1988/MI:P).

Maximum amounts of nutrients accrued in standing crops of emergent macrophytes, including above- and below-ground structures, average 17.6 g/m^2 for nitrogen and 3.7 g/m^2 for phosphorus (Klopatek 1978/*Rem). Increased nutrient availability may lead to uptake in excess of current needs (Steward and Ornes 1975/FL; Klopatek 1978/*Rem; Kadlec and Kadlec 1979/*; Barko and Smart 1980; Heliotis and DeWitt 1983/MI:P; Kistritz et al. 1983/BC:Eem).

Wetland vascular plants can remove nutrients only as long as the plants (or their attached epiphytes and microbes) are accumulating biomass (Yarbro 1979/NC:fo). Biomass accumulation (primary productivity) and thus nutrient uptake is controlled by a host of other environmental conditions (e.g., current, shading, turbidity, hydroperiod, and sediment oxygen and organic content), so uptake may depend more on site-specific characteristics than on plant species. Salt marsh cordgrass retains nitrogen but is less effective for retaining added phosphorus (Valiela et al. 1973/Eem, Patrick et al. 1976/LA:E, DeLaune and Patrick 1980/LA:Eem). Generally, plant growth is more likely to be limited by nitrogen in saltwater and by phosphorus in freshwater (Farnworth et al. 1979/*), although some exceptions are noted in Section 2.6 (Production Export).

Nutrient concentrations in living plant tissue are usually 10 to 100 times those in surrounding waters (Nixon and Lee 1986/*) and may be up to 10,000 times greater (Peverly 1985/NY). However, the relative magnitude of nutrient storage in plants appears to vary greatly. For example, less than 20 percent of the phosphorus was tied up by vascular plants in a Wisconsin marsh (Sloey et al. 1978/*), whereas in Florida lakes, 20 to 96 percent was contained in submersed macrophytes (Canfield et al. 1983/Lab). Less than 5 percent of the nutrient uptake was attributed to plants in a coastal wetland (Turner et al. 1979/GA:E) and about 3 percent in a Michigan fen (Richardson et al. 1978/MI:Pem). However, more than 90 percent of the nitrogen and phosphorus added to a hardwood swamp in Florida was removed year round, with no evidence of accumulation in the sediments, suggesting that the vegetation was the major nutrient sink (Boyt et al. 1976/FL:P).

Nutrient retention in wetlands is also enhanced by algae and microorganisms, especially fungi and yeasts (Richardson and Marshall 1986/MI:Pem), that are attached to vascular plants and their entrained litter. Epiphytes and microbes enhance nutrient retention by rapidly taking up nutrients released by wetland vascular plants (Wetzel 1983/*), from the water column (Atchue et al. 1983/VA:Pfo, Heliotis and DeWitt 1983/MI:P, Grimm 1987/AZ:R), or from litter (Davis and van der Valk 1978/Pem, Stephenson et al. 1981/*, Marinucci and Bartha 1982/NJ:Eem, Richardson and Marshall 1986/MI:Pem). Although nutrients are readily leached from fresh litter, nutrient concentrations of the litter compartment as a whole may not decline (Heliotis and DeWitt 1983/MI:P, Sollins et al. 1985/OR:fo).

However, the relative importance of the nonmacrophyte biota in phosphorus cycling is controversial (Carpenter and Lodge 1986/*ab, Kadlec 1987/*). Epiphytes and microbes living on plant surfaces, as well as invertebrates hosted by wetland plant roots (Fukuhara and Sakamoto 1987), in some situations may actually speed nutrient release (Spangler et al. 1976/WI:R,P, Anderson and Sedell 1979), rather than enhance retention.

Most rooted wetland vegetation translocates oxygen to the sediments, especially in oligotrophic environments. This has been reported either to reduce nutrient retention by accelerating decomposition (Hackney 1987/NC:Eem), or to increase nutrient retention by phosphorus immobilization within the sediments (Jaynes and Carpenter 1980/MI:L) and by denitrification (Gersberg et al. 1986/CA:Pem, Kadlec and Hammer 1988/MI:P). However, vascular plants may also inhibit nitrogen removal that would occur from denitrification (e.g., Kaplan et al. 1979/MA:Eem; Buresh and Patrick 1981) and from ammonia volatilization (Buresh and Patrick 1981).

Sedimentation and Accumulation of Organic Matter in the Substrate. In addition to taking up nutrients directly or indirectly, vegetation enhances retention of sediment (see Section 2.4, and Predictor 12 in Section 3.5) and, consequently, of nutrients by physically slowing currents and allowing nutrient-laden sediment to be deposited. Nutrients are commonly adsorbed to the sediments deposited by the slowed current regime (Copeland and Dickens

1969/*E, Kaplan et al. 1974, Karr and Schlosser 1977/*, Gosselink and Turner 1978/*, Boto and Patrick 1979/*). The amount of phosphorus attached to suspended sediment is often 2 to 5 times that available in dissolved forms (Froelich 1988).

Large reserves of nitrogen (Brown 1985, Smith et al. 1985/LA:E, DeLaune et al. 1986/LA:Eem) and, to a much larger extent, phosphorus (Herron et al. 1984/UT:Pem, Bayley et al. 1985/Man:Pem, Kadlec and Hammer 1988/MI:P) can be carried into wetlands during high run-off. Unless resuspended and flushed out by wind mixing, currents, and biochemical reactions, these nutrients are deposited, and become stored in wetland sediment and as peat deposits (Boto and Patrick 1979/*, Mitsch et al. 1979/IL:fo, Jordan et al. 1986/MD, Maret et al. 1987/WY:P). Peat accumulation may also enhance nutrient removal through ammonium adsorption (Gersberg et al. 1984/CA:Pem). In one watershed, 20 percent of the sediment deposition (and 40 percent of the phosphorus associated with it) occurred in forested wetlands (Cooper et al. 1986/NC:fo).

The time lag between nutrient release, immobilization within the sediments or peat, and peak flows largely determines whether nutrients are flushed from the system, recycled within it, or buried in sediments or as peat (Davis and van der Valk 1978/Pem, Prentki et al. 1978/WI:L, Livingston and Loucks 1979/*, van der Valk et al. 1979/*, Odum and Smith 1981/*Ptem, Turner et al. 1983/FL:L, Mattraw and Elder 1984/FL:fo, Elder 1985/FL:fo, Nixon 1988/*L,E,M). This is discussed further, with reference to carbon, in Section 2.6.2.

Normally, most nutrients have only an ephemeral existence in the surface waters, and rapidly return to the sediments or are stored in peat (Klopatek 1975, 1978/WI:Pem; Bernard and Solsky 1977; Kadlec and Kadlec 1979/*). Generally, the internal nutrient transfers between underlying organic soils and live plants are greater than external transfers related to surface water inputs and outputs (Kadlec and Hammer 1988/MI:P).

Hydrologic conditions in late autumn and early winter are probably most critical, as most nutrients are leached within the first few days or weeks after plants die (Lee et al. 1975/*, Mason and Bryant 1975, Odum and Heald 1975/*Efo, Brinson and Davis 1976/NC:R, Chamie and Richardson 1978/*p, Davis and van der Valk 1978/Pem, Odum and Heywood 1978/*Ptem, Prentki et al. 1978/WI:L, Simpson et al. 1978/*, Kadlec and Kadlec 1979/*, van der Valk et al. 1979/*, Mulholland 1981/NC:fo, Pieczynska 1986/EU:L). Most plant tissues lose 5 to 30 percent of their dry weight shortly after death by leaching, with the highest percentages being from soft herbaceous leaves and the lowest percentages from woody tissues (Cummins 1974, Heal and French 1974/AK:Pem, Klopatek 1975/WI:Pem, Mulholland 1981/NC:fo, Brock 1984, Polunin 1984, Peverly 1985/NY).

Nonhydrologic factors influencing the rate of peat accumulation include temperature, pH, moisture, salinity, oxygen concentration, and characteristics of the plant materials being decomposed (Gallagher 1978/*, Godshalk and Wetzel 1978/*L) (see Chapter 3). Temperature is the primary factor controlling the rate of litter decomposition, with decomposition proceeding faster at higher temperatures (Mitsch and Gosselink 1986:224/*). Since the rate of decomposition is generally slower under anaerobic conditions than when oxygen is present (Cook and Powers 1958/NY:Pem, Chamie and Richardson 1978/*P, Gallagher 1978/*, Gosselink and Turner 1978/*, Klopatek 1978/*Rem, Mathias and Barica 1980, Phillips 1980/*Eab), wetlands generally have lower rates of decomposition than upland areas. Optimum conditions for decomposition are found in moist aerobic environments (Mitsch and Gosselink 1986:224/*).

The rate of decomposition is also influenced by the kind of plant tissue involved. In general, decomposition slows with increasing fiber content (Chamie and Richardson 1978/*P, Gallagher 1978/*, Godshalk and Wetzel 1978/*L) and decreasing nutrient content, especially nitrogen.

Mature wetlands with deep peat layers may recycle proportionately more N and P than young wetlands with shallow sediments (Morris and Bowden 1986/MA:tem). Typical concentrations of nitrogen in peat range from 1.0 to 2.6 percent (Nichols 1983/*), while those for phosphorus typically range from 0.05 to 0.12 percent (Richardson 1985/P,R). Nutrients may be stored on a long-term basis in peat, e.g., 3,000 years in a salt marsh (Redfield 1972/MA), 11,700 years in a Minnesota bog (Heinselman 1970/MN:P), and 600 years in a Wisconsin bog-lake (Friedman and DeWitt 1978/*). Accumulation is slow, especially in temperate climates. Vertical accretion rates of 0.20 to 0.75 centimeter per year (or 10 to 300 g/m^2/year dry matter) generally occur (Nichols 1983*, Richardson 1985/P,R). In one Florida wetland, waterhyacinth wet biomass of more than 35 kg/m^2 resulted in net accumulation of organic matter (Reddy and Sutton 1984/FL:em). Although the below-ground rates of plant decomposition are believed by some to be at least as important as above-ground processes (Gallagher 1978/*), peat accumulation usually represents only a small portion of the production (Richardson and Marshall 1986/MI:Pem).

Nutrient additions to wetlands seldom accelerate the rate of peat accumulation or nutrient uptake by microbes (Richardson and Marshall 1986/MI:Pem), because increases in productivity are offset by increases in decomposition and mineralization (Sampou 1985). Thus, although wetland soils rich in organic compounds may naturally hold large quantities of nitrogen and phosphorus, their capacity to retain additional nutrients through peat accumulation is very limited.

Nutrient Interactions with the Sediments. Within the sediments, nutrients may exist either dissolved in interstitial waters or in association with solid particles. Nutrients may be associated with solid particles as ions or precipitates, or they may be bound within organic or inorganic particles (Kadlec and Kadlec 1979/*). Nutrients may enter the sediments either directly or by complexing with dissolved or suspended particulate matter and precipitating (Darnell et al. 1976/*, Windom 1977). Metal ions, especially aluminum and iron, readily immobilize phosphorus by adsorption and precipitation (Windom 1977, Sewell 1982/AU:E, Callender and Hammond 1982/MD:E, Nichols

1983/*, Heliotis and DeWitt 1983/MI:P, Brinson et al. 1984/NC:fo, Richardson 1985/P,R). Richardson (1985/P,R) found that the phosphorus adsorption capacity of wetland soils could best be predicted from their extractable aluminum content. These reactions predominate under acid to neutral pH (Nichols 1983/*). Although humic and fulvic acids can complex with phosphorus in the presence of iron (Richardson et al. 1978/MI:Pem, Tilton and Kadlec 1979/MI:P), under alkaline conditions (Lee et al. 1975/*) organic matter has little capacity to retain phosphorus (Nichols 1983/*). Thus, fine mineral soils, because they usually have higher concentrations of aluminum and iron, typically have much higher capacities to retain phosphorus (Richardson 1985/P,R). Phosphorus can also be precipitated with calcium mainly under alkaline conditions (Lee et al. 1975/*, Fetter et al. 1978/WI:Pem, Nichols 1983/*).

The phosphorus retention capacity of many wetland soils is limited, and decreases with time as chemical adsorption sites become saturated (Heliotis and DeWitt 1983/MI:P, Nichols 1983/*, Brinson et al. 1984/NC;fo, Richardson 1985/P,R). Saturation often occurs at concentrations of phosphate around 1.0 mg/l (Nichols 1983/*). At higher concentrations, phosphorus may be held less tightly by physical adsorption (Nichols 1983/*).

Adsorption and precipitation of phosphorus does not necessarily result in permanent retention. Phosphorus can be desorbed, if there is a sufficient concentration gradient (Kadlec and Kadlec 1979/*, Fisher et al. 1982b/NC:E, Nichols 1983/*). Thus, wetland soils may act as phosphorus buffers, regulating concentrations in the water column (Nichols 1983/*). For example, Dierberg and Brezonik (1983/FL:fo) observed that, after sewage additions to a cypress dome were stopped, inorganic nutrients were released to the waters from the sediments and vegetation for 20 months. Moderate sediment concentrations of phosphorus in streams do not, however, appear to be readily desorbed and released (Prairie and Kalff 1988a/QUE:R).

Oxygen conditions within or above a wetland's substrate may be important in controlling phosphorus fluxes. A thin oxidizing layer at the sediment surface, if present in a wetland, may help keep phosphorus trapped within the sediments (Darnell et al. 1976/*, Kadlec and Kadlec 1979/*, Avnimelech and McHenry 1984). Breakdown of this layer under anoxic conditions associated with excessive detritus deposition or peat accumulation (particularly during low seasonal flows) can result in phosphorus release (Syers et al. 1973, Correll 1978*E, Crow and MacDonald 1978/*, Klopatek 1978/*Rem, van der Valk et al. 1979/*, Day et al. 1980/LA:fo, Callender and Hammond 1982/MD:E, Doremus and Cleseri 1982). The released phosphorus can subsequently be flushed into adjoining waters if the wetland's anoxic waters are mixed by wind or currents (Mathias and Barica 1980), and denitrification may be inhibited by the anoxia, thus diminishing the nitrogen removal function of the wetland.

Anoxic conditions in the water column may be created by the presence of wetland plants (e.g., duckweed) that diminish mixing and reduce oxygen as they decay (Godshalk and Wetzel 1978/*L, Carpenter and Adams 1979/*Lab, Carpenter and Greenlee 1981, Reddy 1981/FL). Anoxic sediments not only release more phosphorus to waters low in phosphate, they also adsorb more phosphorus from waters high in phosphate (Heliotis and DeWitt 1983/MI:P). However, in soils where aluminum is the major retention mechanism for phosphorus, the adsorption potential should not be greatly affected by aerobic/anaerobic changes because aluminum is stable under varying redox conditions (Knight et al. 1984/NC:P). The equilibrium point between sediments and overlying waters may also vary seasonally with temperature (Stow et al. 1985/LA:L).

Although the typically extensive decomposition that occurs in wetlands may discourage sediment phosphorus retention, it may enhance nitrogen removal (Reddy and Sutton 1984/FL:em). Abiotic processes may not be as important in nitrogen removal as they are for phosphorus retention (Heliotis and DeWitt 1983/MI:P, Brinson et al. 1983/NC:fo, Nixon and Lee 1986/*). Although total nitrogen concentrations may be high in the sediments (Smith et al. 1985/LA:E), they are usually in equilibrium with water concentrations (Boynton et al. 1980/MD:E, Day et al. 1981). The oxidized layer, which typically occurs at the surface of wetland soils, does not act as a trap for nitrogen as it does for phosphorus. However, it may encourage conservation of some forms of nitrogen (Bowden 1986/MA:tem), and the roots of some wetland plants may encourage denitrification by creating oxidized conditions at depth (Gersberg et al. 1986/CA:Pem). Concentrations of micronutrients, iron, clay, and organic matter may further affect nitrogen retention in soils (Darnell et al. 1976/*, Klopatek 1978/*Rem).

Denitrification, Nitrogen Fixation, and Ammonia Volatilization. Denitrification is frequently a critical process in wetlands because it results in nutrient removal rather than retention. Denitrification is the microbial conversion of nitrate to gaseous nitrogen, resulting in a permanent loss of nitrogen from a wetland. This process occurs under anoxic or near-anoxic conditions. Prerequisites include sufficient organic matter (Tilton and Schwegler 1978/*; van Kessel 1978, Kadlec and Kadlec 1979/*, van der Valk et al. 1979/*, Gersberg et al. 1984/CA:Pem, Richardson and Nichols 1985/*, Lorenz and Biesboer 1987/MN:Pem) and moisture (Knowles 1982, Hussey et al. 1985/WY:fo), but these probably are seldom so scarce in wetlands as to be limiting (Brinson et al. 1984/NC:fo). Even 2-year-old artificial wetlands with limited organic matter accumulation were found to be capable of denitrification (Stengel et al. 1987/EU:P).

There are two sources of nitrate for denitrification: diffusion from the water (referred to as direct denitrification) and nitrification (coupled denitrification). Nitrification, the microbial conversion of ammonia to nitrate, occurs under aerobic conditions. Due to this coupling of nitrification and denitrification, the rate of denitrification usually proceeds most rapidly with fluctuations between, or in proximity to, aerobic and anaerobic conditions (Reddy and Patrick 1975/FL, Phung and Knipling 1976, Bowmer 1987/AU:P). Thus, within wetlands, the upper sediment stratum is the primary site of denitrification (Gersberg et al. 1986/CA:Pem).

Nitrogen fixation is the opposite process. Gaseous nitrogen is converted or fixed, usually into organic forms of nitrogen, by bacteria and blue-green algae. Although fixation is typically an anaerobic process, nitrogen may be fixed by some blue-green algae in the presence of oxygen (Capone and Taylor 1980/*Eab). Also, several wetland vascular plant species (e.g., speckled alder, sweet gale, and bayberry, and some species of the genera *Azolla, Lemna*, and *Juncus*) host nitrogen-fixing bacteria.

Rates of nitrogen fixation in various aquatic systems range from 0.1 to 3 kg/ha/day (Jones 1974; Wiebe et al. 1975; Burris 1976/AU:E; Hanson 1977/GA:Eem; Bohlool and Wiebe 1978; Teal et al. 1979/*; Casselman et al. 1981/LA:Eem,ab; Capone 1982/NY:Eab; Chapman and Hemond 1982/MA:P; Nixon 1982/*Eem; Varshney and Mandhan 1982; Bahr et al. 1983/*LA,MS; Hanson 1983/GA:Eem; Jordan et al. 1983; Josselyn 1983/*CA:tem; Ogan 1983; Owens and Stewart 1983/UK:E; Triska et al. 1984/OR:R; Stow et al. 1985/LA:L). For wetlands, rates of nitrogen fixation were summarized by Nichols (1983/*) and include 4.5 to 15 g N/m^2/year for salt marshes (DeLaune et al. 1976/LA:Eem; Kaplan et al. 1979/MA:EEm; Casselman et al. 1981/LA:Eem,ab) and 0.2 to 0.39 g N/m^2/year for forested wetlands (Dierberg and Brezonik 1981/FL:fo). The highest rates measured from salt marshes were along the southern Atlantic and Gulf coasts (Nixon and Lee i986/*). Input of nitrogen through nitrogen fixation can represent up to 50 percent of the annual loading (DeLaune et al. 1981/LA:Eem) and can be a major nutrient subsidy (e.g., Casselman et al. 1981/LA:Eem,ab). However, measurements of nitrogen fixation should be reviewed with caution, since measurement techniques have many sources of error, yielding both exceptionally high and exceptionally low values (Smith 1980/*E). Nitrogen fixation was found to be greater on submerged wood of beaver pond wetlands than on similar wood in streams (Francis et al. 1985/QUE:R).

Nitrogen may also be removed from wetlands by ammonia volatilization. This process occurs at high temperatures and at pH values greater than 7.5 (Isirimah 1972/WI:L). Ammonia disappearance rates of 20 to 30 mg N/l/hr have been measured in hypertrophic lakes (Murphy and Brownlee 1981), and 5.7 and 3.2 mg N/m^2/day from salt and brackish marshes, respectively (Smith and DeLaune 1983/LA:tem). For a palustrine bog, Hemond (1983/MA:P) estimated 90 mg N/m^2/year. The rate of ammonia volatilization is positively related to ammonia concentrations and pH (Smith and DeLaune 1983?LA:tem, Gersberg et al. 1986/CA:Pem).

A third mechanism for nitrogen removal is the biotic pathway. Seasonal emergence of aquatic insects and consumption of nutrient-rich aquatic plants by migrating waterfowl may represent seasonal, and sometimes permanent, losses of nitrogen from a wetland. Biotic losses from an Arizona riverine system were 0.05 mg N/m^2/day (Grimm 1987/AZ:R). Data from lakes (e.g., Vallentyne 1952/L) suggest this is not a major removal mechanism, but the situation in shallow, noncontiguous, eutrophic wetlands may be different from that of lakes.

Denitrification generally exceeds nitrogen fixation in aquatic systems (Seitzinger 1988/*). Reductions in nitrogen concentration have been attributed to denitrification in many wetlands (Isirimah and Keeney 1973/WI:L; Kitchens et al. 1975/SC:fo; Klopatek 1975/WI:Pem, 1978; Nedwell 1975/Efo; Lee et al. 1975/*; DeLaune et al. 1976/LA:Eem; Jones et al. 1976/IA:Eem; Patrick et al. 1976/LA:E; Windom 1977; Fetter et al. 1978/WI:Pem; Sloey et al. 1978/*; Hill 1979; Kaplan et al. 1979/MA:Eem; Tilton and Kadlec 1979/MI:P; Turner et al. 1979/GA:E; Boynton et al. 1980/MD:E; Rabalais 1980/*E; Brinson et al. 1981b/NC:fo; George and Antoine 1982; Klapwijk and Snodgrass 1982/ONT:L; Nishio et al. 1982, 1983/E; Brinson et al. 1983/NC:fo; Gersberg et al. 1983/CA:Pem, 1984, 1986; Smith and DeLaune 1983/LA:tem; Seitzinger et al. 1984/RI:M; Hill and Sanmugadas 1985; Jacobs and Gilliam 1985a,b/NC:P; Smith et al. 1985/LA:E; Lindau et al. 1988/LA:fo). For example, from less than 1 to 80 percent of the annual nitrogen loading may be lost via denitrification (Isirimah and Keeney 1973/WI:L, Boynton et al. 1980/MD:E, Smith et al. 1985/LA:E), with estuarine systems generally having higher rates than freshwater systems (Seitzinger 1988/*). However, in the case of freshwater systems, denitrification usually accounts for a larger portion of total nitrogen lost (Seitzinger 1988/*). Among freshwater systems, the greatest denitrification removals are probably from shallow, eutrophic lakes.

Research in salt marshes has not yielded consistent results. Salt marshes generally remove nitrate from overlying waters, which supports the hypothesis that marshes are sites of active denitrification (Nixon 1980/*). In one estuarine wetland, approximately 24 percent of the total particulate nitrogen and 35 percent of the total nitrogen inputs were lost by denitrification (Boynton et al. 1980/MD:E); for another estuary, 50 percent were lost (Smith et al. 1985/LA:E). Several other salt marsh studies have indicated that denitrification exceeds nitrogen fixation, with a net export of nitrogen ranging from 2 to 12 g/m^2/year (DeLaune et al. 1976/LA:Eem; Haines et al. 1977/GA:Eem; Valiela and Teal 1979/MA:Eem; Kaplan et al. 1979/MA:Eem). However, measurements of exchanges between marshes and surrounding waters show a net export of nitrogen from marshes, suggesting that measurements of denitrification are too high (Nixon 1980/*).

In a small lacustrine wetland, the predominance of denitrification (versus fixation) resulted in a net loss of 20 g N/m^2/year (Anderson 1974/EU:L). In a palustrine wetland, denitrification removed less than 1 percent of the annual loading, and nitrogen fixation was not even detectable (Isirimah and Keeney 1973/WI:L). In some palustrine bogs, nitrogen fixation is a minor input, at least in comparison to atmospheric deposition. In other bogs (Hemond 1983/MA:P) and perhaps in some cypress dome wetlands (Howarth et al. 1988/*), fixation seems more important, exceeding denitrification by as much as 10 times.

The wide variability in reported rates of denitrification and nitrogen fixation, and their relative importance in wetland ecosystems, appears to be a result, at least in part, of the degree of nitrogen enrichment. Increases in nitrogen concentration, especially inorganic forms, appear to favor denitrification, especially when hydraulic detention times are long (Nedwell 1975/Efo; Patrick et al. 1976/LA:E; Valiela et al. 1976/MA:Eem; Capone and Taylor 1980/*Eab; Hemond 1983/MA:P; Smith and DeLaune 1983/LA:tem; Smith et al. 1985/LA:E; Bertani et al. 1987; Brodrick et al. 1988). The addition of organic forms of nitrogen, in contrast, can inhibit denitrification while encouraging nitrogen fixation (Capone 1982/NY:Eab, Gersberg et al. 1983/CA:Pem), although there is some evidence to the contrary (Brodrick et al. 1988). Fixation can also be encouraged by addition of inorganic forms of nitrogen (Valiela et al. 1973/Eem; Capone and Taylor 1980/*Eab; DeLaune and Patrick 1980/LA:Eem; Capone 1982/NY:Eab; Ogan 1983). Also, nitrogen fixation can be encouraged by phosphorus enrichment; high concentrations of phosphorus (resulting in a nitrogen-to-phosphorus ratio of 10 or less) can favor nitrogen fixation (Nichols 1983/*, Ogan 1983) and, perhaps in a few cases, offset nitrogen losses via denitrification.

In general, the rate of nitrogen fixation tends to be greatest at intermediate levels of enrichment (Farnworth et al. 1979/*). If the degree of nitrogen enrichment is a major factor controlling rates of nitrogen fixation and denitrification, then wetlands that buffer nitrogen concentrations may have a significant value in water quality maintenance or improvement. However, some researchers believe that neither of the nitrogen-cycling processes is necessarily driven by the amount or type of available nitrogen (Capone and Taylor 1980/*Eab; Hanson 1983/GA:Eem; Nichols 1983/*), or at least that they are driven by sediment-bound nutrients, rather than water column concentrations (Seitzinger 1988/*).

Summary of Processes. Reviews of mass balance studies show that wetlands do generally act as sinks for nitrogen and phosphorus both under nutrient-enriched and natural conditions (Nichols 1983/*, Nixon and Lee 1986/*). The amount of these nutrients retained varies widely among wetlands and is not clearly related to input (Nixon and Lee 1986/*). Generally, removal efficiency is greater with longer retention times, earlier plant community successional stages (Grimm 1987/AZ:R), and lower loading rates (Nichols 1983/*). With time, the capacity of some wetlands to retain or remove additional phosphorus declines, and the role of the principal biochemical phosphorus remover, may shift from the microbe-litter complex (Herron 1985/UT:Pem) to wetland vascular plants, and finally to the sediment (Richardson and Marshall 1986/MI:Pem), whose ultimate ability to assimilate phosphorus is related to its extractable aluminum content (Richardson 1985/P,R).

However, there is probably considerable site-to-site and year-to-year variability in this process. Regardless of which biochemical processes dominate, physical processes (e.g., scouring, detention, burial) frequently have an overriding effect on the ultimate nutrient balance. Although some wetlands have been shown to be effective nutrient sinks for long periods, our predictive ability is currently limited. For critical decisions, continuous monitoring of sediment and peat accretion rates, hydraulic detention times, redox potential, pH, and extractable aluminum may give an initial estimate of assimilative capacity. Based on extensive studies of a palustrine (fen) wetland, Richardson and Marshall (1986/MI:Pem) estimated that such a wetland could assimilate no more than 10 to 15 kg/ha/year of phosphorus.

2.5.3 Interactions With Other Wetland Functions

Ground Water Recharge, Ground Water Discharge, Floodflow Alteration, Sediment Stabilization. Nutrient removal/transformation normally has no significant influence on these functions, except to the extent to which it may regulate plant growth, which in turn may affect these functions via changes in soil bulk density and evapotranspiration.

Sediment/Toxicant Retention. These functions are highly compatible. Wetlands that are efficient at nutrient retention generally occur in low-energy areas that favor sedimentation. Similarly, areas with rapid rates of sedimentation tend to trap nutrients within the substrate by burial. Moderately enriched wetlands might also help detoxify some contaminants, because nutrient additions stimulate the decomposition and denitrification processes responsible for converting insoluble organics into harmless gases (e.g., Major et al. 1988).

Production Export. Although nutrient removal/transformation and nutrient export would seem to be incompatible, this is not always the case. For example, wetlands that maintain water quality by retaining inorganic nutrients during a critical period (generally the growing season) may export nutrients (nitrogen, phosphorus, and carbon) at other times of the year. Also, wetlands that retain some nutrients on an annual basis may still play a valuable role in production export by changing the form of nutrients that are exported (e.g., changing inorganic nutrients to organic nutrients, see Elder 1985/FL:fo). The effects of enrichment on production and decomposition are described in Section 2.5.2.

Aquatic Diversity/Abundance, Wildlife Diversity/Abundance. By altering the nutrient regime, nutrient retention may have either positive or negative effects on fish and wildlife. For moderately enriched wetlands not susceptible to anoxia and fish kills, nutrient retention in sediments or macrophytes should have a neutral or beneficial effect on the in-basin fishery resource. If nutrients are tied up in plants and not released until the end of the growing season, lower light intensities and cooler temperatures then limit growth and decay of oxygen-depleting nuisance algae. Nutrients can also mitigate the effects of moderate levels of contamination (Fairchild et al. 1984).

Recreation. Nutrient retention can enhance this function by preventing downstream nuisance algal blooms and associated fish kills downstream, but not necessarily within the wetland itself.

Uniqueness/Heritage. The effects of nutrient removal/transformation on this function depend on the situation.

2.5.4 Direct Economic Significance

Nutrient retention in wetlands, by limiting eutrophication of surface waters, may help maintain fisheries and other economically important wildlife (e.g., furbearers, migratory waterfowl) and recreation. It may also reduce the necessity for construction of costly waste treatment facilities (depending on the desired use).

2.6 Production Export

2.6.1 Definition

Production export (Figure 7) refers to the flushing of relatively large amounts of organic material (specifically, carbon from net annual primary and secondary productivity) from the wetland to downstream or adjacent deeper waters. The relationship between exported production and its eventual utilization by consumers in the food chain is not easily predicted. This is due in part to the fact that the location of production and the utilization are often spatially and temporally separated. Other mechanisms of production export, such as insect emergence and consumption by wide-ranging vertebrates, further confound the relationship between production and consumption within aquatic systems. At best, in large systems (e.g., estuaries), secondary productivity can be shown to be only very loosely correlated with primary production and export from wetland systems (Nixon 1980/*, Livingston 1984/*FLE:E, Odum 1984/*e), although in more confined aquatic systems where wetlands dominated the landscape (e.g., Prairie Pothole Region), the correlation is stronger (Carpenter and Lodge 1986/*ab). In some systems, production may be augmented by chemosynthetic bacteria, but the magnitude of this interaction is unknown (Valiela 1984/MA:E).

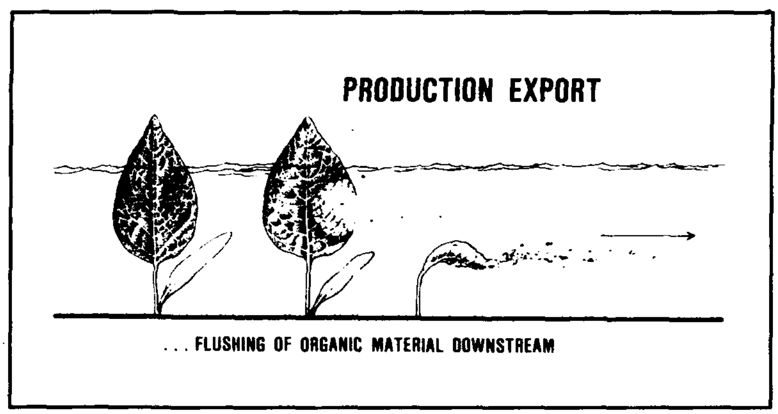


Figure 7. Schematic representation of production export

This section examines the linkages between wetlands and downstream or adjacent waters. It focuses on the production of organic foods within the wetland and the utilization of the exported production by fish and aquatic invertebrates. The release of nutrients through decomposition, although an important factor in maintaining the productivity of both wetlands and adjacent waters, is not extensively considered here, but is addressed in part in Section 2.5.

The high value for production export commonly ascribed to wetlands is based on two attributes. First, wetlands have relatively high rates of primary production. Indeed, primary production values for some wetlands (e.g., salt marshes, mangroves, and freshwater marshes) are among the highest in the world (Thayer et al. 1979/*Eab, Zieman 1982/*FL:ab). Second, wetlands are often hydrologically linked to deepwater systems so the potential for export is higher than for terrestrial systems. Flushing by currents and waves controls the aquatic export of organic material from wetlands.

Net carbon export has been reported from many wetlands, particularly the export of dissolved carbon from tidal wetlands (e.g., Teal 1962/GA:Eem; Cameron 1972/*Eem; Day et al. 1973/LA:Eem; Nixon and Oviatt 1973/RI:Eem; Moore 1974/VA:Eem; Eilers 1975/OR:Eem; Axelrod et al. 1976/LA:Eem; Heinle and Flemer 1976; Settlemyre and Gardner 1977; Valiela et al. 1978/MA:Eem; Phillips 1980/*Eab; Wolaver and Spurrier 1988a/SC:Eem). Further, high fishery production in areas adjacent to estuarine wetlands provides correlative support to the hypothesis that wetlands export particulate organic matter, which supports detrital food chains and high fishery production (Turner 1977/*Eem; Bahr et al. 1983/*LA,MS). The value perhaps lies not so much in the quantity of the carbon export, but rather in the efficiency with which it is converted to fish biomass (Nixon 1988/*L,E,M).

The primary productivity of wetlands and the link between wetlands and deep water is well documented; however, the fate of wetland production is still largely unknown. Some materials are leached from living and senescent plants as dissolved organic matter. This material may be utilized by microbes and other organisms within the wetland or exported (see Section 2.6). A large amount of the production in wetlands is consumed by microbes within the wetland. Most is mineralized, but some is converted into microbial biomass that in turn may be converted into fishery production via trophic intermediaries.

In situations where the wetland actually exports organic material, two questions must be addressed to determine whether there is a relationship between exported organic material and fishery production. First, do fish and invertebrates actually use this resource? Second, does the organic material that originated in the wetland significantly increase fishery productivity over what it would be without this resource.

Although wetlands definitely have the potential to export organic materials, the chemical and physical processes involved are very complex and poorly understood. Wetlands are clearly important in supporting fisheries by providing habitat (Turner 1977/*Eem, Boesch and Turner 1984/*Eem), but there is no clear correlation between wetland area or production and total fishery yields (Nixon 1980/*). The wetland evaluation in Volume II makes no attempt to evaluate the quality or value of exported production to consumer organisms, nor to estimate its spatial and temporal distribution.

2.6.2 Processes

Major processes affecting production export are the following (Odum and Heald 1975/*Efo, de la Cruz 1979/*, Livingston and Loucks 1979/*):

- Productivity of potential food sources (macroscopic and microscopic).
- Nitrogen-fixing ability of potential food sources.
- Dispersal and cycling of potential food sources.

These are discussed below. Factors that control these processes are documented in Chapter 3.

Productivity of Potential Food Sources. All aquatic organisms are ultimately supported by at least one of the following food (energy) sources and their associated microbial populations:

- Phytoplankton.
- Dissolved and particulate organic run-off from uplands (allochthonous plant and animal material blown or washed into the waters).
- Benthic and epiphytic attached microalgae.
- Decaying or undecayed vascular wetland plants (macrophytes).
- Macroalgae.

Only the last three of these groups are restricted to wetlands, as defined in this manual.

Average productivity ranges of potential aquatic food sources in wetlands are as follows:

- Phytoplankton: 100-2,000 $g/m^2/yr$ (Correll 1978/*E)
  Maximum = 9,000 $g/m^2/yr$ (Westlake 1963/*)
- Benthic microalgae: 180-300 $g/m^2/yr$ (Correll 1978/*E)
- Macrophytes: 560-1,980 $g/m^2/yr$ (Mitsch and Gosselink 1986:168/*)
- Nonwoody emergents: 1,370-1,980 $g/m^2/yr$ (Mitsch and Gosselink 1986:168/*)
  Maximum = 8,500 $g/m^2/yr$ (Westlake 1963/*)

- Aquatic bed: 340-940 $g/m^2/yr$ (Adamus and Stockwell 1983/*:23-33)
  Maximum = 5,000 $g/m^2/yr$ (Westlake 1963/*)
- Scrub-shrub: 575-980 $g/m^2/yr$ (Brinson et al. 1981a/*fo)
- Forested: 870-1,040/$g/m^2/yr$ (Mitsch and Gosselink 1986:168/*)

Holding equal many factors, greater primary productivity usually implies greater potential for supporting downstream or offshore food chains (McCormick and Somes 1982/MD). On the other hand, in rare instances where excessive plant growth blocks channels and raises the wetland surface, water circulation and the flushing of potential food sources may be reduced at least seasonally (Fisher and Carpenter 1976).

Wetland morphology, and ultimately geology, is a major factor in determining which potential food source (macrophytes, phytoplankton, etc.) is the most plentiful in the wetland. If a wetland is shallow, sheltered, soft-bottomed, and unshaded, then macrophytes rather than benthic algae or phytoplankton are often the most abundant potential food source. However, the various food sources help to sustain one another. Nutrients released during the die-off of one plant form can be essential to the productivity of another (i.e., the remineralization cycle). For example, the nutrient cycling interdependence of salt marshes and mud flats (Welsh 1980/NY:Eem, Wolaver et al. 1980/SC:E) as well as the biogeochemical interactions between tall and short *Spartina* zones (Chrzanowski and Spurrier 1987/SC:Eem) have been documented. On a larger scale (Chesapeake Bay), Jordan et al. (1986/MD) noted:

> The tidal marshes had very little effect on the net flow of nutrients through the tidal headwaters. In addition, the net effects of the low and high marshes were opposite and tended to cancel each other. In contrast, the subtidal area trapped large amounts of nutrients. As with the freshwater swamp, much of the phosphorus trapping followed a severe storm. Net nitrogen flux in the subtidal area varied from year to year...However, the trapping of phosphorus occurred consistently from year-to-year.

Despite much attention given to wetland macrophytes, many unobtrusive plants, including some epiphytic algae, phytoplankton, and benthic algae, may be at least as productive in some regions and wetland types as macrophytes (Correll 1978/*E, Diaz et al. 1982b/*E, Jones 1984/WI:Lab), and more valuable in food chains (Sculthorpe 1967/*, Naiman and Sedell 1980). The epiphyte production in one shallow eutrophic lake was 48 to 79 percent of vascular aquatic production (Jones 1984/WI:Lab).

Macrophytes increase the productivity of epiphytic algae by providing surfaces for attachment (Allanson 1973/UK:Lab, Cattaneo and Kalff 1980/QUE:ab, Nelson and Kadlec 1984/*Pem, Thayer et al. 1984/*Eab), and stands of macrophytes may even support higher phytoplankton production than does open water (Rabe and Gibson 1984/Lab). However, heavy epiphyte growth can reduce the productivity of the "host" (Phillips et al. 1978, Bulthius and Woelkerling 1983/AU:Eab, Thayer et al. 1984/*Eab).

Macrophytes also intercept floating detritus and microbes, causing them to be deposited at least seasonally within the wetland (Dawson 1980, Adams and Prentki 1982, Bulthius et al. 1984, Chrzanowski and Spurrier 1987/SC:Eem, Wolaver et al. 1988/SC:Eem). Substantial retention by salt marshes of eelgrass and algal detritus from adjacent systems has also been reported (Seliskar and Gallagher 1983/*Eem). Paradoxically, in some riverine systems, the lowest export of detritus often occurs where living macrophyte biomass is the greatest (Fisher and Carpenter 1976), due to the ability of living macrophytes to trap detritus arriving from upstream. However, others have pointed to large rafts of flotsam as evidence of export (Zedler 1982/*CA:Eem, Josselyn 1983/*CA:tem, Cranford et al. 1987/NS:Eem). Organic matter entrained and deposited within a wetland may also be flushed out during infrequent storms (see discussion below).

Phytoplankton community productivity in some wetlands is less seasonally variable than macrophyte productivity, with a diversity of species responding in sequence to changing environmental conditions (Correll 1978/*E). However, currents and turbidity often limit phytoplankton production more severely than they do emergent plants (Hargrave et al. 1983/NS:M,E; Gordon et al. 1985/NS:Eem). Whatever production occurs, however, may be more easily dispersed to consumer organisms.

Macroalgae also represent a significant portion of the productivity of some wetlands (Thorne-Miller et al. 1983/RI:Eab), particularly in tidal areas. Although benthic algae are productive year-round (Odum et al. 1984/*Ptem), benthic microalgae are more productive in spring and fall than in summer when macrophyte shading limits microalgae productivity (Nixon 1982/*Eem). Systems with lower vascular plant productivity and more open canopies generally have greater algal growth (Zedler 1982/*CA:Eem; Nelson and Kadlec 1984/*Pem). The contribution of benthic microalgae and epiphytes to wetland productivity varies from less than 1 percent in tidal freshwater (Whigham and Simpson 1976/NJ:tem) and irregularly flooded salt (Nixon 1982/*Eem) marshes on the East Coast to as much as 60 percent in hypersaline salt marshes on the West Coast (Zedler 1982/*CA:Eem).

Although various wetland plant species have characteristic (if somewhat overlapping) genetic capacities for primary production, the primary productivity of a wetland cannot be predicted solely by identifying its plant species or noting their apparent physical vigor. Primary productivity of macrophytes is most often a reflection of nutrient availability and other environmental conditions rather than of species-specific genetic characteristics. Productivity among macrophytes within the same wetland class is similar when each species is present in a favorable environment for its growth. Factors that usually increase macrophyte productivity are moderate increases in current, an aerobic substrate, seasonal flooding, nutrient enrichment, longer growing season, and increased solar energy (e.g., decreased turbidity or reduction of canopy shading). These are documented and discussed further in Chapter 3.

The effect of upland runoff on the productivity of estuaries has received much attention (Cross and Williams 1981/*). However, freshwater input does not guarantee a marsh's productivity. The strength of the salt-wedge counter current (Boon 1975/VA:E, Tilley and Dawson) and mixing by tides and wind are probably more important in making nutrients available for primary production than is the amount of freshwater input (Nixon 1980, 1981/*). However, the definition of the saltwater wedge can be enhanced by increased freshwater (riverine) discharge, and a few estuaries do show correlations between fishery harvest and freshwater discharge (Sutcliffe 1972, Nixon 1980/*).

Nitrogen-fixing Ability of Potential Food Sources. Whereas carbon is added to the wetland environment by all plants, nitrogen is added by specific nitrogen-fixing organisms (bacteria and blue-green algae), which can occur in any wetland class or system. Commonly, they are present as floating or benthic mats, or (at least in freshwater wetlands) in a symbiotic relationship with certain macrophytes (e.g., speckled alder, sweet gale, bayberry). In tidal wetlands, microbes are often the predominant nitrogen fixers (Carpenter et al. 1978/MA:Eem; Teal et al. 1979/*). Wetlands dominated by effective nitrogen fixers might be just as important to production export in nitrogen-limited systems as wetlands with highly productive plant species. Nitrogen fixation by wetlands was addressed in Section 2.5.2.

Dispersal and Cycling of Potential Food Sources. The productivity and nitrogen-fixing ability of wetland plants is usually less important to production export than is the widespread dispersal of fixed nitrogen and carbon throughout the wetland and to contiguous areas downstream (Murphy 1962, Nixon 1980/*). This is suggested by the fact that flushing capacity varies to a much greater degree among wetlands than does primary productivity, which varies by a factor of about 10 in estuaries (Nixon 1980/*) and by a factor of 100 in lacustrine wetlands (Carpenter and Lodge 1986/*ab). Production may be exported in the form of dissolved organic substances, particulate organic matter, or even whole, mobile organisms such as finfish and birds.

Physical dispersal (export) of potential food sources from contiguous wetlands is probably due mainly to flushing during severe storm and tidal events (Nixon and Oviatt 1973/RI:Eem; Pickral and Odum 1977/VA:E; Livingston and Loucks 1979/*; Odum et al. 1979/*Ptem; Odum 1980/*Eem; Hackney and Bishop 1981/MS:Eem; Casey and Farr 1982/UK:R; Elder and Mattraw 1982/FL:fo; Verhoff et al. 1982; Borey et al. 1983/TX:Eem; Simenstad 1983/*E; Tate and Meyer 1983/GA:R; Livingston 1984/*FL:E; Mattraw and Elder 1984/FL:fo; McPherson and Sonntag 1984/FL:E; Thayer et al. 1984/*Eab; Wolaver et al. 1984/SC:E, 1988; Nixon 1988/*L,E,M). Major export can occur during just one or a few storms. Seasonal or annual pulses of exporting energy (e.g., spring tides, storms, ice scour, mixing winds) may be at least as important for long-term productivity and for food availability as are the seasonal pulses or long-term trends in food production itself. Furthermore, these pulses need not be frequent. Rarely flooded wetlands may be nearly as important to production export as permanently flooded wetlands if foods accumulating in the wetland between floods are not lost through denitrification, burial, or peat formation. Thus, the hydroperiod is more likely to control the form of the exported nutrients, and ultimately their value to aquatic life, than the absolute amount of nutrients (Elder 1985/FL:fo).

Other factors that influence dispersal include vertical mixing currents (Imberger et al. 1983/GA:E); uprooting by herbivores, e.g., geese, nutria (Smith and Odum 1981); bioturbation by invertebrates (Gallep 1979); interconnectedness of underlying aquifers; shape of the wetland (Odum et al. 1979/*fo, Odum 1980/*Eem); and, in estuaries, by strong salt-wedge countercurrents (Ketchum 1967/*, Correll 1978/*E, Nixon 1980/*). The shape, density, and decomposition rate of the material being dispersed also play a role.

It is commonly assumed that the most visible or easily measured material leaving a wetland, usually particulate organic matter (e.g., seston and detritus from macrophytes or riparian vegetation), is also the material exported in greatest quantity. In reality, export of dissolved organic matter (leached from both living and dead tissues of any alga or vascular plant) is the major form of export from wetlands (Mickle and Wetzel 1978/Lab; Bilby and Likens 1979; Mulholland and Kuenzler 1979/NC:fo; Nixon 1980/*; Brinson et al. 1981a/*fo; Day et al. 1981; Zedler 1982/*CA:Eem; Borey et al. 1983/TX:Eem; Imberger et al. 1983/GA:E; Jordan et al. 1983; Pregnall 1983; Valiela 1984/*; Mann 1988/*E; McDowell and Likens 1988/NH:R; Wolaver and Spurrier 1988a/SC:Eem).

The popularized role of tidal vegetated wetlands as major nutrient exporters is not entirely certain either. Nutrients derived from wetlands have been shown to enrich areas within 10 miles of the shore in Florida (Livingston and Loucks 1979/*), from Florida to South Carolina (Turner et al. 1979/GA:E), and in Louisiana (Ho and Barrett 1977/LA:E, Day et al. 1980/LA:fo). The enrichment of coastal water by wetland-derived detritus and nutrients is less certain on the West Coast (Onuf et al. 1979/*CA:Eem), Chesapeake Bay (Correll 1978/*E), Narragansett Bay (Nixon and Oviatt 1973/*L,E,M), and other New England and Maritime areas (Odum 1980/*Eem, Cranford et al. 1987/NS:Eem). In fact, for the nation as a whole, levels of organic and inorganic nutrients in both the water column and deepwater sediments do not appear to be substantially higher near estuarine areas with much emergent wetland vegetation than in areas with proportionally little (Nixon 1980/*).

If wetlands as a whole conserve rather than export nutrients and organic matter (Kistritz et al. 1983/NC:Eem), what then is the fate of the enormous productivity of wetlands? Generally, less than 10 percent of macrophyte biomass is directly consumed. However, on a local basis, herbivores such as geese, insects, snails, crayfish, muskrats, nutria, fish, and sea turtles may consume up to half the peak plant biomass (Odum and de la Cruz 1967/GA:Eem; Odum and Heald 1975/*Efo; Thayer et al. 1975/*Eab; Anderson and Low/MAN:Pab 1976; Jupp and Spence 1977/UK:Lem; Crow and MacDonald 1978/*;

Kiorboe 1980/EV:ab; Pomeroy and Wiegert 1981; Smith and Odum 1981; Zieman 1982/*FL:ab; Murray and Hodson 1984/GA:P; Odum et al. 1984/*Ptem; Lodge 1985/IN:L; Lodge et al. 1985/IN:L). Such consumption probably hastens export, both aerially and hydraulically (Watson et al. 1984/AU:Eab; Stuart et al. 1985). Wetland plant detritus may also be heavily consumed by commercially important species (e.g., Lewis and Peters 1984/NC:Eem; Prouse 1986/NS:E,M).

Although some production is translocated downward to the roots, the storage capacity of roots varies widely with species. In salt marshes, at least, the below-ground production does not seem great (Hackney 1987/NC:Eem; Valiela et al. 1984/MA:E), comprising perhaps only 5 percent of the total production (Nixon 1982/*Eem).

The fate of below-ground production and how it varies from that of above-ground production are also not known. Wetlands with deeply rooted macrophytes that store most of their nutrients in roots are less likely to export nutrients, unless the wetlands are deeply scoured or inhabited by an abundance of burrowing organisms, or efficient herbivores (Smith and Odum 1981/*). Factors that influence remobilization of below-ground carbon via decomposition include temperature, moisture, and root tissue chemistry (Clymo 1983/*). Anderson and Hargrave (1984/NS:Eem) found that of organic material (primarily *Spartina* spp.) buried in a salt marsh, approximately 2 percent was leached to the water as dissolved organic carbon, 78 percent was decomposed anaerobically (primarily sulfate reduction), and 20 percent was decomposed by aerobic processes.

For above-ground plant production, the form assumed by exported plant material depends mostly on its decomposition rate. This is influenced by: (1) the inherent species-specific physical and chemical makeup of the plant, especially the fiber content (Benner et al. 1985/GA:em); (2) the physical, chemical, and biological (detritivore) environment through which it travels as it decomposes; (3) the length of time elapsed since death; and (4) loading rates of other organic matter to the wetland. Presumably, faster decomposition rates promote faster cycling of nutrients, higher net primary productivity, and thus a greater quantity of material available for export, other factors being equal. Algae and submerged (aquatic bed) plants are usually more readily decomposed than emergent macrophytes (Chamie and Richardson 1978/*p, Gallagher 1978/*, Godshalk and Wetzel 1978/*L), whereas woody plants, particularly evergreen species, are the slowest to decay (Sidle 1986). Decay rates of specific freshwater plants are summarized by Odum and Heywood (1978/*Ptem), and for saltwater plants by Kenworthy and Thayer (1984/NC:Eab), Valiela et al. (1985/MA:Eem), and Wilson et al. (1987a). Rates range from about 25 days to several years for woody vegetation. Decay is also enhanced by environments with low salinity, high dissolved oxygen (Yates and Day 1983/VA:fo; Avnimelech and McHenry 1984), moderate current velocity, warm temperature, fluctuating water levels, and large populations of fish or benthic invertebrates (Hendricks et al. 1984/VA:L, Taylor and Hendricks 1987/VA:L). Breakdown rates also are generally higher in rivers than in lakes (Hanlon 1982). These factors are discussed further in Section 3.7.

The point at which decomposition yields products most useful to production export cannot be specified, as not enough is known about what types and proportions of the various by-products are most useful. Initial indications are that the more soluble compounds—those released during the first few days of decay—may be most nutritionally desirable (Godshalk and Wetzel 1978/*L, Rice and Tenore 1981, Rice 1982, Valiela 1984/*). If wetland plants reduce water column mixing (Morris and Barker 1977) and too much organic matter accumulates, anoxic conditions may inhibit decomposition or make its products less useful to food chains (Fenchel and Blackburn 1979/*). On the other hand, if too little organic matter is present (e.g., flushing rate is great), thresholds essential to the establishment of decomposing bacteria might be reached more slowly. Further discussion of food availability is provided in Section 2.7.2.

Another major pathway for export of carbon from wetlands is by mobile organisms (Herke 1971/LA:Eem; Hackney 1977/MS:Eem; Knudsen et al. 1977/LA:Eem; Weinstein 1979/NC:Eem; Pomeroy and Wiegert 1981; Weinstein and Brooks 1983/NC:Eem; Thayer et al. 1984/*Eab). Release of gametes from both plants and animals may represent a significant input to food webs in some systems (Jordan and Valiela 1982/MA:Eem). Export of benthic microflora (in amorphous aggregates) may also be important (Ribelin and Collier 1979/FL:Eem).

2.6.3 Interactions With Other Wetland Functions

Ground Water Recharge, Ground Water Discharge, Floodflow Alteration, Sediment Stabilization, Sediment/Toxicant Retention. Production export rates have minimal direct effect on these functions, but may affect bulk density of sediments (an important hydrologic concern) and the retention of metals.

Nutrient Removal/Transformation. Nutrient retention and export of organic material may be compatible because these functions involve different nutrient forms. Nutrient retention may involve the storage of both organic and inorganic forms, but retention of inorganic forms is more important in reducing nuisance plant and algae blooms downstream. Exported production, as defined herein, considers only organic compounds (including organically bound nutrients) that are of potential value in food chains. Because wetlands act as nutrient transformers, they could conceivably (and may generally) accumulate nutrients on an annual basis and still export organic materials.

Aquatic Diversity/Abundance, Wildlife Diversity/Abundance. Depending upon the quantity, quality, and sequencing of exported nutrients, as well as dependence of species upon them, export of production from wetlands may be extremely important in supporting food chains of many terrestrial and aquatic species. Export of organic matter from wetlands may also prevent stagnation resulting from the large biological oxygen demand that could otherwise reduce faunal diversity and abundance. Production export (or peat accumulation) can strongly influence vegetation diversity in riverine wetlands (Day et al. 1988). Detritus from plant production also reduces the bioavailability of some toxicants while increasing that of others (see Section 2.4.2).

Recreation, Uniqueness/Heritage. Production export may cause nutrient levels to become favorable (increased fish production) or unfavorable (algal blooms and anoxia).

2.6.4 Direct Economic Significance

Some regional economies depend almost exclusively on fisheries that in turn may depend heavily on wetlands for exported production. On the other hand, exported nutrients may aggravate downstream pollution problems caused by eutrophication.

2.7 Aquatic Diversity/Abundance

2.7.1 Definitions

Aquatic diversity/abundance (Figure 8) is the support of a notably great on-site diversity and/or abundance of fish or invertebrates that are mainly confined to the water and saturated soils. Fisheries, as defined in this manual, are the finfish and shellfish resources harvested commercially or for sport within the interior or coastal United States. Habitat includes those biological, physical, and chemical factors that support larval, juvenile, or adult forms of aquatic organisms. Examples of habitat factors are food, salinity, temperature, substrate, cover, current velocity, and dissolved oxygen.

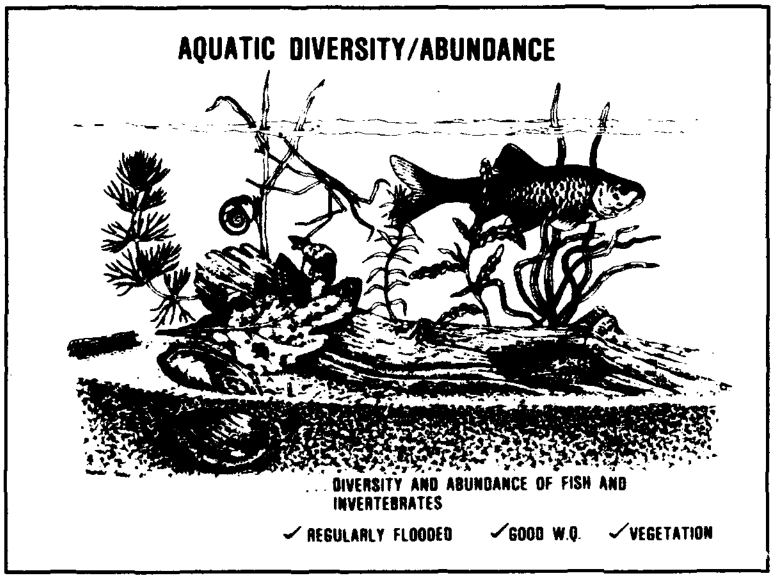


Figure 8. Schematic representation of aquatic diversity/ abundance

The procedure in Volume II permits the assessment of a wetland's importance for general Aquatic Diversity/ Abundance, and its habitat suitability for four Freshwater Fish Species Groups, 90 individual species of freshwater fish, and 133 individual species of saltwater fish, anadromous fish, and invertebrates. Freshwater Fish Species Groups include Warmwater Fish, Coldwater Fish, Coldwater Riverine Fish, and Northern Lake Fish of sport or commercial value.

Both diversity and abundance are addressed as one function. Although diverse aquatic communities are not always productive and vice versa (Whiteside 1970/EU:L, McNaughton and Wolf 1973/*), aquatic diversity and abundance are often positively correlated (e.g., Franz and Harris 1988/NY:E).

Nearly all freshwater fish and many saltwater fish require shallow water provided by wetlands at some stage of their lives for spawning, predator avoidance, shelter from extreme environmental conditions, and feeding. Perhaps two thirds of the commercial finfish harvest on the Atlantic coast consists of estuarine (including deepwater) dependent species (McHugh 1966/*). However, not all wetlands serve as vital fish habitats. For example, many isolated wetlands are too susceptible to lethal oxygen stagnation to support a sustained fishery, and many acidic moss wetlands, hypersaline wetlands, and very shallow wetlands are unfavorable as well.

Vegetation increases the attractiveness of wetlands to most fishes (Hall and Werner 1977/MI:L, Thayer et al. 1979/*Eab, Stoner 1983, Zimmerman and Minello 1984/TX:Eem), but can discourage use by some species or size classes (Vince et al. 1976/MA:E; Crowder and Cooper 1982; McIvor et al./VA:tem, in press). Some 58 (44 percent) of the 133 fishes and invertebrates of commercial or sport value that use saltwater wetlands of the United States show a strong dependence on vegetated wetlands for habitat. About 27 (30 percent) of the 90 freshwater species of commercial or sport value show such a dependence. In both cases, many other species depend on wetlands occasionally for habitat needs, and, as discussed in Section 2.6 (Production Export), some wetlands are broadly beneficial to fisheries through their ability to recycle and export nutrients (Turner et al. 1979/GA:E, Yoder et al. 1981/M, Odum 1984/*e).

The dependence of specific marine and estuarine fishes on vegetated wetlands for habitat or nutritional needs can be inferred from harvest statistics. The total catch of some individual marsh-dependent organisms such as penaeid shrimp is proportionally greater along coastlines with a high proportion of salt marsh (Turner 1977/*E, 1978/LA:Eem). Thirteenfold decreases in shrimp abundance have been reported as a result of bulkheading a marsh (Mock 1967/TX:E). However, the total fishery catch is not always greater in salt marsh-dominated regions, whereas regions not dominated by salt marshes can have rather high catches of estuarine-dependent fishes per unit area (Nixon 1980/*).

When harvest statistics indicate a large catch, it is seldom possible to determine whether this is due to shoreline habitat, nutrient cycling and export capacity (as discussed in Section 2.6), freshwater input, phytoplankton availability, unreliable catch data, or some combination of these factors. Nationwide or statewide estimates of fish habitat abundance are few. One survey (Carlander et al. 1978/*) indicated that the percentage of total lake, impoundment, and stream area inhabited in North America is 54 percent for northern pike, 32 percent for walleyes, 26 percent for yellow perch, 10 percent for saugers, and 1 percent for muskellunge. More data on aquatic resource use (e.g., Judy et al. 1984/*) and relative abundance are necessary to assess the significance of various wetlands for particular species.

2.7.2 Processes

Specific factors affecting this function include:

- Water quality (physical and chemical).
- Water quantity (hydroperiod, flow, and depth).
- Cover, substrate, and interspersion.
- Availability and quality of food sources.

These factors can be highly interrelated and are important at every life stage.

Water Quality. Salinity and temperature are the water quality parameters normally most important to marine and estuarine fishes, with dissolved oxygen and turbidity occasionally playing an important role. Freshwater fishes are most often influenced by temperature and dissolved oxygen; turbidity, alkalinity, and pH are sometimes important. All water quality factors can act on fishes either directly (physiologically and behaviorally) or indirectly (by affecting food supply).

Despite some popular notions to the contrary, estuaries are more important as nursery areas than as spawning areas, at least in the Southeast (Odum and Smith 1981/*Ptem). The most important nursery areas of most fisheries species, at least on the south Atlantic and Gulf coasts, appear to be the fresher (Rozas and Hackney 1983/NC:tem, 1984/NC:tem; Rogers et al. 1984/GA:Eem; Smith et al. 1984/VA:tem), more turbid, shallower (but regularly flooded), upper reaches of salt marshes and tidal creeks (Herke 1971/LA:Eem; Weinstein 1979/NC:Eem; Rozas and Odum 1987b/VA:tem), where primary productivity per unit volume (but not per unit area) is also highest (Herman et al. 1968/MD:E). Postlarval shrimp as well as finfish apparently seek out such areas (Weinstein 1979/NC:Eem). Individuals then move seaward as they grow (Yakupzack et al. 1977/LA:Eem).

Temperature variation within an estuary may be great, particularly on a vertical scale. No particular temperature range is optimal for fisheries species; it varies by species, life stage, region, and duration of the acclimation period. In fresh water, temperature is largely influenced by shading from vegetation and relief, water depth and turbidity, ground water inflow, and mixing by wind and currents. As long as dissolved oxygen remains adequate, most fish can tolerate moderately elevated temperatures.

Coldwater species are more sensitive to dissolved oxygen concentration than warmwater or saltwater species. In general, the most diverse and productive freshwater and saltwater fishery usually occurs at concentrations of over 5 mg/l and at least 80 percent saturation, especially for coldwater species (Bell 1973/IL:fo). Concentrations of dissolved oxygen are most often lowered by rising water temperatures, decomposing organic matter, plant respiration, and input from deoxygenated ground water.

Alkalinity and pH, and related measures such as hardness and total dissolved solids, normally do not affect fish directly (physiologically), but often influence productivity of algal and macrophytic plants that form the basis of the food chain. This relationship may be less certain in riverine wetlands (Hynes 1970/*R) than in lacustrine (Jenkins and Morais 1971/L) and palustrine (Moyle 1945/MN) wetlands. Circumneutral to slightly alkaline pH and alkalinities of at least 25 mg/l are believed to be optimal for productivity of freshwater fishes (Wesche and Rechard 1980/*R). Alkalinity and pH are largely related to wetland geology, climate, and amount of organic matter present.

Turbidity and related measures, such as total suspended solids and light refraction, may occasionally be important limiting factors in both freshwater and saltwater systems. However, because nutrient concentrations often are correlated with turbidity, these effects are difficult to separate. Larval and juvenile stages of some fishes benefit from turbid water because it affords them protection from predators (Cyrus and Blaber 1987a, 1987b; Minello et al. 1987/TX:Eem). The direct effects of suspended sediment on fish (clogged gills, etc.) are usually much less significant than indirect effects on spawning areas, fish movements, and the food chain (from reductions in plant productivity due to blocking of solar radiation) (Farnworth et al. 1979/*). However, even these effects are not always significant. Instances where sediment additions had little effect on aquatic invertebrate density have been noted by Chutter (1969), and Barton (1977/ONT:R). Minimal effects of sediment on species composition have been noted by Hansen (1971/IA:R). Lack of impact in some of these instances may have been due to the timing of sedimentation events or to rapid flushing of the added sediment.

Although direct tolerance to turbidity is species-specific, estuarine and marine fishes tend to be more sediment-tolerant than freshwater species, warmwater species more tolerant than coldwater species (Muncy et al. 1979/*), and adult stages more tolerant than egg or larval stages. Many interacting water quality parameters influence the exact threshold for sediment damage, and percent departure from background levels (compared to natural variation) is probably a better measure of impact in some situations than are actual concentrations of suspended solids.

Water Quantity. Water quantity includes factors such as depth, volume, velocity, width, and hydroperiod. It directly affects the amount of living space available to fish and their food organisms. It also affects fish passage, cycling of nutrients, territoriality (Hynes 1970/*R), and light penetration crucial to photosynthesis and respiration.

Within the usual range of wetland depths (0 to 6.6 ft), no particular depth range is generally better than another for aquatic diversity/abundance. Productivity of fish foods is often greater in shallower areas, both because such areas receive more solar radiation and because they are more frequently subject to dewatering, which (so long as plants are not scoured) increases nutrient availability. Shallow areas also tend to support larger growths of macrophytes, which provide cover and harbor dense concentrations of invertebrate foods (Menzie 1980/NY:Rab; Rozas and Odum 1987a/VA:tem). However, deeper areas of wetlands are usually more stable and allow space for freer movement. For adult fish, the minimum depth threshold is usually determined by the size of the fish and the need for physical movement. Although most adult finfish have difficulty navigating through shallow water, many commercially important aquatic invertebrates that are sessile in their adult stage (e.g., sandworms) can survive without continual surface water flooding. Larval and juvenile fishes are also quite capable of movement in very shallow water. In fact, such life stages probably select shallow areas because these provide food and afford protection from large piscivorous fishes (Power and Matthews 1983/OK:R, Schlosser 1987/IL:R, McIvor and Odum 1988/VA:tem). The maximum depth threshold is usually related to factors correlated with depth, such as temperature, light penetration, and availability of macrophytes for cover. The ratio of total dissolved solids to mean depth has frequently been correlated with fish productivity in lacustrine wetlands.

Water velocity influences the movement and respiration of fish and, in the case of filter-feeders, food availability. No single velocity range is best for fish productivity, as both flowing and nonflowing waters may produce high standing crops. For warmwater and saltwater species, relatively little is known about optimum or maximum current velocities, but in general, thresholds are probably lower than those for coldwater species.

Water velocity must also be considered in terms of other habitat needs. Velocities or depths that normally exceed the preference range of a particular species may be necessary within part of its home range to maintain habitat suitability. For example, high flow velocities or increased depth may be necessary to flush fine silt from spawning gravels, to reduce excessive growth of wetland vegetation (which otherwise restricts access to invertebrate foods), or to promote germination of wetland plants which require periodic flooding.

To support a fish community, isolated wetlands must have a permanently flooded portion, but other wetlands need not. In fact, seasonally or irregularly flooded wetlands may be critical to some riverine and palustrine species that feed in such areas when flooded or that use seasonally flooded wetlands for spawning and nursery habitat (Guillory 1979, Welcomme 1979/*, Baker and Ross 1981, Pollard et al. 1983/LA:fo, Walker and Sniffen 1985). Productivity and diversity of most warmwater fishes in riverine systems tend to be greater where water levels are seasonally variable, although beyond some unknown threshold, fluctuations become too frequent and aquatic vegetation essential as cover cannot become established.

Cover, Substrate, and Interspersion. Cover includes areas used for protection from predators and environmental extremes, or as substrate for feeding and reproduction. Cover for fishes may be provided by low overhanging trees, logs, boulders, vegetation, bottom sediments, undercut banks, water turbulence, and turbidity. Requirements vary greatly among species and life stages, but in general, partly submersed wetland plants provide suitable cover for most aquatic invertebrates and juvenile fishes (Turner 1977/*Eem; Boesch and Turner 1984/*Eem). For adult trout, overhead riparian cover may be more important than cover provided by deepwater pools or boulders (Wesche et al. 1987/WY:fo). Cover needs may be greatest in larger wetlands and lower portions of riverine systems, because these habitats have proportionately less shoreline than other wetland areas.

Some data concerning substrate preferences are available, particularly for coldwater riverine species, but these are closely interrelated with velocity preferences. No single substrate type is uniformly important to fishery diversity and productivity. Sand substrates are most likely to be unproductive for the largest number of freshwater (Hynes 1970/*R) and saltwater invertebrates, but many exceptions exist. In estuarine systems where dissolved oxygen conditions are adequate, loosely compacted (Turk et al. 1980/NS:E) and organic substrates may be highly productive as fish nursery habitats. Invertebrates that support riverine fishes are most numerous where canopy vegetation allows considerable input of terrestrial insects, or where aquatic bed or emergent vegetation is present in moderate, interspersed amounts. Channels through vegetation are essential to daily and seasonal movements of most species of adult fish.

Availability and Quality of Food. The general importance of macrophytic detritus in wetland food chains has been suggested by studies using radiotracers, analysis of gut contents (Darnell 1961/LA:E,L), and various other techniques (Odum 1984/*e). Multiple stable isotope studies of nutritional pathways in estuaries show that numerous species depend on vegetated wetlands for their nutritional requirements (Peterson et al. 1985/MA:Eem; Heaton 1986/*; Peterson and Fry 1987). Organisms that rely on salt marsh cordgrass detritus as an ultimate food source include mummichogs, mud snails, menhaden, hard clams, blue crabs, and winter flounder. The value of detritus in feeding experiments and stable isotope studies varied with plant species (Findlay and Tenore 1982/Eem, Smock and Harlowe 1983/VA:P), the age of the detritus (Kirby-Smith 1976/*E, Smock and Harlowe 1983/VA:P), and the animal species (Hughes and Sherr 1983/GA:E; Seliskar and Gallagher 1983/*Eem; Peterson et al. 1985/MA:Eem).

In most riverine systems, detritus from terrestrial and especially aquatic plants is critical to the fishery food chain (Marzolf 1978/*fo). In Montana streams, the greatest richness of stream invertebrates was associated with lake-outlet streams that exported 300 to 600 mg/m^3 of particulate organic matter (Perry and Sheldon 1986/MT:L,R), and lacustrine carbon outwelling increased stream macroinvertebrate biomass in Quebec (Morin and Peters 1988/QUE:R). However, inputs of seasonally flooded wetland plant material may reduce algal productivity and biomass in some receiving waters after an initial stimulatory effect

(Jackson and Hecky 1980, Arvola 1984/EU:L, Guildford et al. 1987/MAN:L). The inhibitory effect may be due to the humic matter's ability to tie up iron, an essential plant nutrient (Sakamoto 1971, Koenings and Hooper 1976).

As a directly consumed food source, the live tissues of benthic algae and phytoplankton are usually more digestible and perhaps more nutritious to most aquatic consumers than are tissues of vascular macrophytes (Patrick 1967/*Eab; Kirby-Smith 1976/*E; Haines 1977/GA:Eem; Zedler et al. 1980/CA:Eem; Peterson 1981/*E). Vascular macrophyte detritus, although commonly ingested by wetland fish (Werme 1981, Jeffries 1982, Lewis and Peters 1984/NC:Eem), is not necessarily assimilated (Buchsbaum et al. 1986/MA,NJ:Eem, White et al. 1986), and growth of detritivores may be slow (Prinslow et al. 1974, Tenore 1977/*, Briggs et al. 1979, White et al. 1986). The value of the ingested detritus may ultimately depend less on the nutritional value of the detritus itself than on the microbes and algae that colonize it (Newell 1965; Darnell 1967/*; Fenchel 1970/FL:Mab, 1972; Odum 1970a; Gallagher et al. 1976/GA:Eem).

Value may also depend on the ability of the detritus to serve as a durable substrate, capable of being reprocessed repeatedly (Montague 1987/*), because nutritional value is greatest when detritus is fine and well processed (Naiman and Sedell 1979). Some evidence of the food value of vascular plant detritus has been based on the assumption that a declining carbon-to-nitrogen (C:N) ratio is in fact indicative of increasing nutritional value. This has been demonstrated for decomposing estuarine emergent species (Odum and de la Cruz 1967/GA:Eem; Adams and Angelovic 1970/*Eab; de la Cruz 1975), eelgrass (Thayer et al. 1977), mangrove leaves (Odum and Heald 1975/*Efo), and a variety of freshwater species (Brinson 1977/NC:fo, Odum and Heywood 1978/*Ptem). However, other studies suggest that the C:N ratio is not necessarily correlated with nutritional value (de la Cruz 1980/*Eem; Hackney and de la Cruz 1980/MS:Eem).

Many saltwater species (e.g., oysters, clams, menhaden) may benefit at least as much from phytoplankton and other food sources as they do from detritus (Odum 1984/*e). Evidence chiefly from laboratory feeding experiments also indicates that some commercially important organisms may derive greater nourishment directly from phytoplankton (Heinle et al. 1973/MD:E, Kirby-Smith 1976/*, Haines 1977/GA:Eem, Williams 1980) or DOC (Gallagher et al. 1976/GA:Eem). Also, DOC from many plant sources may be more useful to food chains than detritus, because it is converted directly and efficiently into bacterial biomass (often forming amorphous aggregates) which is immediately available to deposit- and filter-feeding organisms such as shellfish (Gallagher et al. 1976/GA:Eem; Correll 1978/*E), amphipods, tadpoles, and fish (Baylor and Sutcliffe 1963; Bowen 1980, 1981, 1984/SA; Barlocher et al. 1988). Algae and vascular plants may also use DOC. However, other evidence suggests that much of the DOC exported from wetlands is in a form that is not readily assimilated (Wallis et al. 1981, Matson and Klotz 1983/CT:E, Imberger et al. 1983/GA:E, Hill 1985/TX:Pem,Lem).

Perhaps the most important way in which wetlands affect the quality of potential food sources is the manner in which they regulate the conversion of inorganic phosphorus and nitrates into organic compounds (Mattraw and Elder 1984/FL:fo; Elder 1985/FL:fo). This is essential because most invertebrates and fish cannot directly use these substances in their dissolved inorganic forms; only their organic by-products are useful.

The export of large quantities of a high-quality food means little unless it is available at a season when consumers are present. Aquatic organisms generally have evolved either to maximize use of specific foods predictably available at various times of the year, or to be opportunistic in their food selection. For example, the timing of shorebird migration may be keyed to the precise season of food availability in wetland sediments (Schneider and Harrington 1981/MA:E), and migration and spawning of blue crabs along the west coast of Florida is timed to take advantage of spring detritus inputs from the Apalachicola River (Livingston 1984/*FL:E).

Seasonal and life stage-related shifts in the types of food sources used have been documented for many organisms (Nixon and Oviatt 1973/RI:Eem; Haines and Montague 1979/GA:Eem; Whitlatch 1982/*E; Cudlip and Perry 1988/MN:L). Many aquatic organisms maximize use of detritus and perhaps dissolved organic matter in late fall, winter, and early spring, when other food sources may be less available (Vannote et al. 1980/*R; Mulholland 1981/NC:fo; McCormick and Somes 1982/MD; Whitlatch 1982/*E). As waters became warmer and less turbid in late spring and summer, plankton may be more extensively used because of their abundance (Livingston and Loucks 1979/*). In upper riverine systems, many fish shift from a diet of attached algae in spring and summer, to terrestrial insects that drop or are blown in during summer, to detritus-associated aquatic invertebrates in autumn and winter (Hynes 1970/*R). In early fall, aquatic macrophytes may be the major energy source (in mid-order riverine systems) when periphyton production is decreasing and allochthonous input has not yet become important (Hill and Webster 1983/*Eem). In lacustrine systems, detritivores may depend on detritus from fringe wetlands during summer and autumn, and then shift to external sources (e.g., entrained leaf litter) during winter and spring (Cudlip and Perry 1988/MN:L).

One conclusion emerges from the above: no single food source is of overwhelming importance to all fish and invertebrates in all wetland areas. The degree to which a reduction in the productivity and/or use of one source will be compensated for by an increase in the productivity and/or use of another is not known. This suggests that perhaps the best situation for stable, diverse, and sustained fish production is a diversity of food sources that provide food throughout the year.

2.7.3 Interactions With Other Wetland Functions

Ground Water Recharge, Ground Water Discharge, Floodflow Alteration. Aquatic organisms have no significant influence on these functions.

Sediment Stabilization, Sediment/Toxic Retention. Aquatic organisms normally have no significant influence on these functions. However, carp and perhaps a few other large fish may occasionally decimate aquatic vegetation and cause appreciable resuspension of sediments in shallow areas (Robel 1961).

Nutrient Removal/Transformation, Production Export. Fish kills may have dramatic effects on the nutrient dynamics of wetlands, particularly those wetlands that are poorly flushed (Kushlan 1974/FL). Less dramatic is the nutrient cycling role of fish that stir up sediments, uproot or feed on vegetation (e.g., carp), or travel large distances on a regular basis (Nakashima and Leggett 1980). Fish and aquatic organisms also may contribute to food chain support by serving as prey to other fish. Burrowing and feeding by some invertebrates may enhance the release of deeply buried nutrients, as well as contaminants.

Wildlife Diversity/Abundance. An abundance of fish and other aquatic organisms can benefit many wetland-dependent wildlife species, such as waterfowl, herons, kingfishers, mink, otter, and raccoon. However, wetlands that offer ideal habitat for aquatic organisms do not necessarily provide ideal habitat for wildlife. Predation by fish (e.g., northern pike) on young waterfowl may be significant in some localized areas. Also, wildlife may use some fish of high commercial value (Wood 1987/BC:R).

Recreation. Fish are a major focus of water-dependent recreation. Recreational boating is typically associated with fishing in many regions.

Uniqueness/Heritage. Aquatic diversity/abundance may contribute significantly to the uniqueness/heritage of an area. However, in certain situations, high populations of "rough" fish may negatively impact the uniqueness/heritage of a wetland (e.g., Robel 1961).

2.7.4 Direct Economic Significance

The contribution of fisheries to the economy varies greatly from region to region. In areas such as coastal New England, the Chesapeake Bay, and the Gulf Coast, fisheries are clearly vital to the regional economy.

2.8 Wildlife Diversity/Abundance

2.8.1 Definitions

Wildlife diversity/abundance is the support of a notably great on-site diversity and/or abundance of wetland-dependent birds. This focus on birds should in no sense imply that other wetland-dependent wildlife, such as many furbearers (e.g., muskrat, nutria, beaver, mink, otter, and raccoon), other mammals (e.g., water shrews, swamp rabbits, seals, manatees), most amphibians, and some reptiles (e.g., bog turtles, sea turtles, water snakes, alligators), are any less important or dependent on wetlands. Future revisions will incorporate other vertebrate groups.

In addition to the general Wildlife Diversity/Abundance evaluation, Volume II assesses wetlands for Breeding, Migration, and Wintering Diversity/Abundance, which evaluates the capability of wetlands to support wetland-dependent birds during each of these periods. In addition, habitat suitability may be evaluated for 14 waterfowl groups and 120 individual species of wetland-dependent birds.

In the "Social Significance" part of Volume II, higher ratings are assigned to wetland types that are located in regions of major concern for waterfowl, as well as those that support wildlife species of special importance (e.g., threatened and endangered species), with exceptionally narrow habitat requirements, or extremely limited distribution in the region. Thus, off-site as well as on-site considerations are addressed by WET in Volume II.

Both diversity and abundance are addressed as one function. Although diverse bird communities are not always productive and vice versa, in many instances diversity and abundance are positively correlated (e.g., Gauthreaux 1978/*fo). Volume II does not require that both parameters exist.

Wetland-dependent species are defined as those that: (a) normally use wetlands exclusively for food and cover throughout most of their US range and spend most of their lifetime within wetlands or (b) would be extirpated from a large region if all wetlands were to be filled. Scores of animals that use wetlands as sources of drinking water, winter cover (e.g., white-tailed deer and ring-necked pheasants), dispersal centers within urban areas (e.g., opossum), or for occasionally obtaining other life requirements are not included. The degree of dependence by any given species on wetlands often varies greatly depending on the abundance and distribution of wetlands and on suitable alternative habitats within the region. For example, urban wetlands and riparian wetlands in the arid Southwest support species that in other parts of their ranges are much less likely to inhabit wetlands.

Few of the species described in this manual can adjust to using terrestrial environments if their wetland habitat disappears. However, some appear to be adapted to the naturally dynamic character of wetlands and may travel great distances in search of replacement wetlands. Their success in doing so depends on the distribution and habitat quality of other wetlands. There are no known cases where a loss of wetland habitat resulted in a population shift to the remaining wetlands without adverse impacts on the total population (Clark and Clark 1979/*, Baines 1988/UK:Pem).

Although the wetland-upland edge is often among the most diverse and productive environments for wildlife (Brinson et al. 1981a/*fo), species richness and population densities in wetlands are sometimes lower than those of adjacent uplands. When viewed from a geographically broader perspective, however, wetlands contribute to the presence of many species that otherwise would be absent from the regional fauna. For example, monotypic moss wetlands (bogs) and tidal emergent wetlands (salt marshes) frequently have impoverished breeding bird faunas, but many of the species that occur in them (e.g., palm warbler and seaside sparrow, respectively) are highly specialized and unlikely to breed in other wetland or terrestrial environments.

The following criteria have been suggested by scientists of the International Union for the Conservation of Nature (Szijj 1972) as defining wetlands of international significance for birds:

- Regularly supports 1 percent (minimum of at least 100 individuals) of the flyway or biogeographical population of one waterfowl species.
- Regularly supports 10,000 ducks, geese, swans, or American coots, or 20,000 wading birds.
- Supports an appreciable number of endangered plant or animal species.
- Is of special value for maintaining genetic and ecological diversity because of the quality and peculiarities of its flora and fauna.
- Plays a major role in its region as habitat for plants and animals of scientific or economic importance.

In a survey of waterfowl migration/wintering habitat in the United States, Bellrose and Trudeau (1988/US) reported the following to represent at least "moderate" densities of waterfowl (number of birds per acre per day):

Flyway	Dabbling Ducks	Bay Divers	Geese
Atlantic Flyway	0.17	0.36	0.26
Mississippi Flyway	0.44	0.06	0.13
Central Flyway	0.73	0.09	0.34
Pacific Flyway	2.87	0.21	0.41

2.8.2 Processes

The major factors affecting this function are:

- Area size.
- Availability of cover
- Availability of food.
- Availability of specialized habitat needs.
- Spatial and temporal arrangement of the above factors.
- Isolation from disturbance.
- Absence of contaminants.

All of these are necessary at every season and life stage.

Area Size. The relationship between bird species diversity and the size of an area has been well documented in studies of island biogeography (e.g., MacArthur and Wilson 1967, Harris 1984/*fo). Changes in species richness generally lag behind changes in area, although a unit change in small areas will have a proportionately greater impact than a unit change in large areas. This topic is discussed in more detail under Predictors 2 and 3 in Section 3.9.

Availability of Cover. Cover consists of features used by birds for protection from predators, conspecifics, and the elements. In wetlands, cover may be provided by woody and herbaceous vegetation, large rocks, topographic relief, debris from upland sources, undercut banks, or the water itself. What constitutes adequate cover depends upon the particular species or species groups of interest. For example, a 4-inch-high growth of sedges provides poor cover for geese but may provide adequate cover for sora rails. Seaside sparrows normally remain well concealed in coastal salt marshes, whereas wintering harlequin ducks prefer marine waters off exposed peninsulas even during the fiercest storms. The cover needs of a particular species may also vary seasonally. The presence of man-induced disturbance such as recreational activities may diminish the quality of available cover for an individual species (e.g., Kaiser and Fritzell 1984/MO:R, Korschgen et al. 1985/MO:P,L, Hoy 1987/TX:L).

The availability of cover is perhaps most influenced by a wetland's hydroperiod. Some birds nest successfully in flooded wetlands (e.g., redhead ducks, prothonatary warblers). Other nesting birds may leave wetlands when flooding submerges emergent vegetation, but flooding may benefit some species as it provides new areas around the wetland periphery that formerly were dry uplands. Temporary drawdown or drought usually encourages lush growth of wetland plants, which is ultimately beneficial as cover (Weller 1978/*), but may also allow isolated islands valuable to nesting waterbirds to become temporarily accessible to predators.

Availability of Food. Wetland birds represent the full range of feeding strategies. Some are opportunists, others are specialists; some are herbivores, others are carnivores, insectivores, or omnivores; some are aerial feeders, whereas others feed on the ground, in the water column, or in sediment. Thus, it is not possible to evaluate a particular wetland as a food source without consideration of the species involved. For waterfowl, rooted vascular aquatic bed vegetation is very important. Shorebirds (e.g., sandpipers and plovers), which are common in wetlands, at least during migration, generally avoid aquatic bed vegetation and favor invertebrate-rich unconsolidated mud and sand flats.

Food habits of wetland wildlife vary both regionally and seasonally. To some extent, birds can shift among various food sources, but the limits of adaptability (i.e., the magnitude, duration, and frequency of the shift) of most species are unknown. For example, brant shifted to sea lettuce during the 1930s when eelgrass temporarily declined (Cottam et al. 1944/E). Although such flexibility may prevent species extinction, some loss to the population often results. Many species make trophic-level shifts on a seasonal basis. For example, many waterfowl species shift from being secondary consumers during the breeding season to primary consumers during the winter (Weller 1975/*). Young of many species eat animal matter during periods of rapid growth, and then shift to plant matter as they mature (Swanson 1988/*).

Availability of Specialized Habitat Needs. Habitat needs that are not directly related to food and cover may be termed specialized habitat needs. For example, bald eagles typically require very tall trees adjacent to open water for use as perches. Cliff swallows need exposed shorelines or puddles as a source of mud for their nests. Examples of nest sites for other species include tree cavities, snags, and islands of various sizes and in various stages of vegetative succession. Such requirements are usually quite inflexible.

Spatial and Temporal Arrangement of Habitat. Perhaps the most critical factor affecting the wetland bird community is the diversity and interspersion of habitat types available. Harris et al. (1983/WI:P) found that bird species diversity in freshwater marshes of Michigan was related linearly with cover type diversity, and curvilinearly with the degree of interspersion. Studies of waterfowl have indicated that breeding populations are more highly correlated with total length of wetland shoreline in a region than total acreage of wetlands (Weller 1979/*). Prairie-nesting waterfowl in particular seem to require a wide variety of closely associated wetlands within their home range (e.g., Dwyer et al. 1979/ND, Flake 1979/*), and multiple cover types are also important to shorebirds (Page et al. 1979/CA:E). Upland bird communities are also typically more diverse and productive along wetland edges (Brinson et al. 1981a/*fo).

When many wetland types are present and distributed evenly throughout a wetland (high interspersion), the "edge effect" is maximized. The amount of cover type diversity and interspersion required is species specific, but in general is inversely related to an animal's home range size (Leopold 1933/*). There are probably limits to the degree to which "edge" is beneficial. At some point, especially for animals with larger territories or requirements for isolation, habitat can become too fragmented. For example, the curvilinear relationship that Harris et al. (1983/WI:P) observed between habitat interspersion and bird species diversity suggests that species diversity decreases, or at least reaches an asymptote, as the size of individual cover types becomes very small. This threshold is presently unknown for most species (Kroodsma 1979/*), but preliminary data from nonwetland habitats, possibly applicable to forested wetlands, suggest that diversity decreases rapidly once the stand becomes smaller than about 80 acres (Thomas et al. 1979/*OR). The exact threshold may vary not only by species but also by season.

Vegetation structural diversity is particularly important for supporting diverse bird communities and has been addressed in several studies (see Predictor 12 in Section 3.9). Both vertical and horizontal variation must be considered.

In addition to spatial diversity within a wetland, it is crucial that a variety of wetland types also be present throughout a region in order to provide habitat to a wide variety of species. This is particularly true in regions subject to climatic perturbations such as drought (mostly those where evaporation exceeds precipitation) or catastrophic flooding (e.g., intense storm regions). Regional wetland diversity is also of particular importance to migratory birds (Chabreck 1979/*b, Weller 1979/*). This topic is discussed further under Predictors 3 and 38 in Section 3.9.

2.8.3 Interactions With Other Wetland Functions

Ground Water Recharge and Discharge, Floodflow Alteration. Although this interaction is usually negligible, beaver dams may have some temporary local influence.

Sediment Stabilization, Sediment/Toxicant Retention. Herbivorous waterfowl occasionally overgraze local aquatic vegetation, thus aggravating erosion and sedimentation.

Nutrient Removal/Transformation, Production Export. Colonial water birds and concentrations of feeding gulls and waterfowl may occasionally play a key role in dispersing or concentrating nutrients within a watershed, at least during certain seasons (Burton et al. 1979/FL:fo). Waterfowl that extract roots as well as above-ground portions of aquatic plants may be critical for cycling appreciable amounts of nutrients (up to 58 percent of plant biomass) that would otherwise remain buried and unavailable to aquatic food chains (Smith and Odum 1981).

Aquatic Diversity/Abundance. Birds may have a significant impact on aquatic diversity/abundance through direct consumption. For example, Wood (1987) found that predation by common merganser broods could significantly reduce Pacific salmon abundance in some streams.

Recreation. Wildlife observation is often a focus of recreational boaters, particularly canoeists.

Uniqueness/Heritage. Bird diversity/abundance may contribute significantly to the uniqueness/heritage of an area. However, in certain situations, the presence of large numbers of birds may become a nuisance or cause damage to areas considered important for uniqueness/heritage.

2.8.4 Direct Economic Significance

Birding, hunting, and other wildlife-oriented recreational activities are important to many local and a few regional economies. Dollar values from the national harvest of wetland-dependent wildlife are given by Chabreck (1979a). Depredations by wetland-associated birds may result in locally significant crop losses. Bird collisions with vehicles, especially aircraft, may result in extremely hazardous and costly damage.

2.9 Recreation

2.9.1 Definitions

Recreation includes both consumptive (e.g., sport fishing, food gathering, hunting) and nonconsumptive (e.g., swimming, canoeing, kayaking, birding) forms of recreation that are water dependent and occur in either an incidental or obligatory manner in wetlands.

Many wetlands in the United States do not regularly support recreational activities, not necessarily because they are unsuitable, but rather because they are isolated from population centers. Their importance may yet be realized as the demand for recreation increases.

2.9.2 Interactions With Other Wetland Functions

Ground Water Discharge and Recharge, Floodflow Alteration. Probably no significant interaction exists.

Sediment Stabilization, Sediment/Toxicant Retention. Use of wetlands for power boating and swimming accelerates shoreline erosion, resuspends sediments, and may provide more opportunity for trapping suspended sediments. These activities may also reduce sediment-trapping effectiveness by reducing shoreline vegetation.

Nutrient Removal/Transformation, Production Export. Domestic wastes from large boats and onshore recreational facilities may enrich wetlands and provide an opportunity for nutrient retention, nutrient removal, and perhaps production export. Blue-green algae commonly associated with enrichment are fixers of nitrogen, and thus might ultimately enhance production export, especially in estuarine and marine environments. Effectiveness of retention might decline if boats and swimmers resuspend nutrient-laden sediments otherwise destined for long-term burial, or if recreational wastes impose an oxygen demand sufficient to alter the direction of nutrient fluxes in the sediments.

Aquatic Diversity/Abundance, Wildlife Diversity/Abundance, Uniqueness/Heritage. The effects of disturbance from water sports on these functions are negative or neutral. The magnitude of these effects generally depends on the frequency of disturbance, but few thresholds have been determined. Habituation of wildlife to disturbance can have an important mitigating effect in some instances. By altering levels of nutrients and suspended solids, recreational activities can affect the abundance of wetland plants that serve as food and cover for fish and wildlife.

2.9.3 Direct Economic Significance

Wetland-based recreation is important in many local and regional economies.

2.10 Uniqueness/Heritage

2.10.1 Definitions

Uniqueness/heritage includes use of wetlands for aesthetic enjoyment, nature study, education, scientific research, open space, preservation of rare or endemic species, protection of archaeologically or geologically unique features, maintenance of historic sites, and an infinite number of other mostly intangible uses.

Whether or not one considers Uniqueness/Heritage a valid wetland value depends on one's cultural, philosophical, and theological perspectives. For instance, some groups view wetlands primarily as a major source of insect and wildlife pests or may consider protection efforts a threat to the Nation's economic survival. Others view wetlands in the same manner that our society views art as an entity that is appreciated and protected, not for any practical reason, but just because it is there, and somehow special. This view (i.e., "heritage value") may be held by millions of people, including some who have never seen a wetland.

Wetlands may be protected to preserve our options for the future. For example, harvesting the peat contained in many wetlands may be uneconomical today, but future energy needs may call for its use. Destruction of wetlands may preclude this option. Similarly, it is uncertain how much, if any, of the biological gene pool can be sacrificed through species extinctions before profound ecological disruption results. There is also uncertainty about how much of a wetland can be sacrificed before profound functional changes occur, either to the wetland or to regional resources. See Chapter 4 for a discussion of the social significance of this function.

2.10.2 Interactions With Other Functions

This function is generally compatible with all other wetland functions.

3.0 Prediction of Wetland Effectiveness and Opportunity

The purpose of this chapter is to provide the rationale, and supporting information, for each of the predictors of wetland function used in Volume II of the Wetland Evaluation Technique (WET). WET's interpretation keys use these predictors in a heuristic, hierarchical manner to evaluate wetlands with respect to their opportunity and/or effectiveness at performing individual functions (see Section 2.0). The **predictors** are variables that directly or indirectly measure the physical, chemical, and biological processes or attributes of a wetland and its surroundings.

This chapter is organized by function and predictor, with individual predictors referenced according to their corresponding question numbers in Volume II. Information provided for each predictor includes the following:

Ranking. A statement that describes how wetland types or attributes would be ranked by WET if the specified predictor were the only criterion for evaluation.

Rationale. A description of the logic used to arrive at the ranking, and information (when available) quantifying the relationship of underlying processes to the predictor, or that of the predictor to its function.

Confidence in Ranking. A subjective estimate of the relative confidence that should be placed in the stated rankings based on the quality and availability of supporting information. Factors most likely to lead to different rankings of wetland types are noted. Such factors include differences in research methods, regions, species or life stages, wetland systems, seasons, and interacting physicochemical influences.

Potential Importance to Function. A subjective estimate of the relative importance of the named predictor to the function (i.e., how integral the predictor is to the wetland process or attribute being evaluated).

Measure. A brief description of the information that must be collected to respond to the question relative to a specific predictor.

Directness of Measure. A subjective estimate of the relative directness of the relationship between the measure and its associated function. In some cases, more direct but time-consuming measures are mentioned if their use will improve predicting power.

A crucial assumption for all rankings in Chapter 3 is that all other predictors are held constant. **This assumption is not a part of the technique presented in Volume II**. In fact, the interpretation keys in Volume II were structured in a manner that attempts to account for the host of interactions among predictors in different situations. The simplifying assumption used in the present section was made so the basic structural elements of WET could be presented clearly.

3.1 Ground Water Recharge

1. Climate

Ranking: With respect to ground water recharge, climate is used in the keys as a classification variable rather than a predictor, i.e., if the wetland is located in a precipitation-deficit region, one set of predictors is used, whereas if precipitation exceeds evaporation, another set applies.

Rationale: Not applicable.

Confidence in Ranking: Not applicable.

Potential Importance to Function: Not applicable.

Measure: Identification of precipitation-deficit regions.

Directness of Measure: Not applicable.

6. Local Topography

Ranking: Ground water recharge is more likely to occur in situations where the topographic relief is characterized by a sharp downslope away from most of the wetland than in situations where terrain is generally level.

Rationale: Although many exceptions exist, the slope of the water table often parallels the topography of the land surface (Fetter 1980/*). Thus, when local topography slopes sharply away from the wetland, the water table will likely slope away as well, resulting in a hydraulic gradient favorable for movement of water into the ground water system. For example, wetlands located near the rim of a plateau or crest of a regional watershed divide (e.g., separating the watersheds of two large streams) are more likely to experience recharge (Dunne and Leopold 1978). Playa wetlands have also evidenced recharge (Wood and Osterkamp 1984).

Confidence in Ranking: Low.

Potential Importance to Function: Moderate.

Measure: Identification of special topographic features such as topographic divides, or significant elevational changes.

Directness of Measure: Low. Measurement of ground water gradients is preferred to measurement of surface topography.

8. Inlets, Outlets

Ranking: A wetland with a permanent inlet but no permanent outlet is more likely to recharge ground water than one with an outlet. A wetland with neither an outlet nor an inlet is intermediate in likelihood of recharging ground water.

Rationale: Several factors support this ranking. First, a higher hydraulic gradient will likely be present in an area with no outlet, especially if an inlet is present. Second, the longer water is retained in an area, the greater the opportunity for it to percolate through the substrate. Third, wetlands without outlets generally experience more water-level fluctuations, resulting in inundation of unsaturated soils. Unsaturated soils are more likely to transmit water than saturated soils. Finally, lack of an outlet suggests that water is being lost either through recharge or through evapotranspiration, especially if an inlet is present.

Confidence in Ranking: Moderate (humid regions) to low (arid regions).

Potential Importance to Function: Moderate.

Measure: Identification of inlets and outlets for surface flow.

Directness of Measure: Moderate. In areas of karst terrain, disappearing surface water does not necessarily indicate recharge to deep aquifers.

10. Wetland System

Ranking: Palustrine, lacustrine, and riverine systems are more likely to recharge ground water than are marine and estuarine systems.

Rationale: Marine, estuarine, and most tidal riverine systems have insufficient vertical head to recharge ground water on a net annual basis. Also, recharge with saline waters is usually not desirable. Riverine systems may have higher recharge potential than lacustrine or palustrine systems due to their greater soil diversity and seasonal inundation of coarse, unsaturated floodplain sediment (Mundorff 1950/NC:R). However, this is usually offset by their lack of sufficient vertical head on a year-round basis.

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of wetland system as described by Cowardin et al. (1979/*).

Directness of Measure: Low.

11. Fringe Wetland or Island

Ranking: Fringe or island wetlands are least likely to recharge ground water.

Rationale: Near shorelines, ground water **discharge** is more likely to occur than recharge (McBride and Pfannkuch 1975/MN:L, Winter 1977/:L). Fringe wetlands or those associated with islands have proportionately more shoreline than nonfringe or nonisland wetlands. Thus, these wetlands will be more likely to discharge, rather than recharge, ground water.

Confidence in Ranking: Low.

Potential Importance to Function: Not applicable (classification variable).

Measure: Determination of whether the wetland is a fringe wetland or is associated with an island.

Directness of Measure: Not applicable.

21. Land Cover of the Watershed

Ranking: Wetlands with watersheds of mostly impervious surfaces are more likely to recharge ground water than are wetlands in watersheds with pervious surfaces (e.g., forest, agriculture).

Rationale: Extensive paved surfaces, topographic disruptions, and the presence of wells directly tapping the aquifer are associated with developed environments, and all may lower or diversify the potentiometric contours. Lowered or diversified potentiometric contours enhance the likelihood of recharge, at least to local aquifers (Fetter 1980/*). Whitehead and Langhtee (1978) reported an instance where as few as six bounding wells were sufficient to reverse the effects of preexisting ground water movement. To a much lesser extent, potentiometric contours can be lowered by vegetation (especially woody) during the growing season, enhancing at least shallow recharge (Meyboom 1966/MAN, Sloan 1979/ND:Pem). Also, the presence of extensive paved surfaces on uplands prohibits recharge from occurring there, and runoff becomes focused on unpaved depressions such as wetlands where opportunities for recharge are thus increased.

Confidence in Ranking: Low.

Potential Importance to Function: Moderate.

Measure: Determination of the major land cover type of the wetland's watershed.

Directness of Measure: Low. Characterization of permeability should focus on developed areas known to have formerly been recharge areas.

23. Ditches/Canals/Channelization/Levees

Ranking: Wetlands **without** ditches, canals, levees, or similar artificial features (which cause surface water to leave faster than it would if these features were not present) are more likely to recharge ground water than wetlands *having such features.*

Rationale: Surface drainage alterations generally result in a low water table, lower minimum water levels, and drying of wetlands (Griswold 1978, Verry and Boelter 1979). The flattened hydrologic gradient can result in reduced ground water exchange.

Confidence in Ranking: Low. In some cases (e.g., artificial recharge pits), drainage may make the hydraulic gradient **more** favorable for recharge. The effects of any particular activity or plan depend on type of drain, ditch/canal depth and spacing, physical properties of the wetland substrate, presence or absence of vegetation, and other factors (Verry and Boelter 1979/*P, Moore and Larson 1980/MN:Pem).

Potential Importance to Function: High.

Measure: Documentation of hydrologic modifications that cause surface water to leave at a faster rate than originally occurred.

Directness of Measure: Not applicable.

24. Soils

Ranking: Wetlands with permeable soils or those located in karst (limestone) regions are most likely to recharge ground water. Wetlands located in watersheds dominated by soils with slow infiltration rates are more likely to recharge ground water than are those located in watersheds with high infiltration rates. Wetlands having soils with slow infiltration rates are less likely to recharge ground water.

Rationale: Slow infiltration by watershed soils results in delivery of more runoff to the wetland, making more water available for recharge. However, if wetland soils are impermeable or have slow infiltration rates, significant ground water movement cannot occur, no matter how much water is available. Karst regions represent a unique situation in which ground water movement is controlled primarily by the orientation of rock fractures and cavities formed by the dissolution of limestone and other soluble rock types (Ritter 1986:479/*). These fractures and solution cavities allow large amounts of water to be funneled into underground drainage systems (Ritter 1986:476/*).

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of wetland and watershed infiltration rates from soil surveys and determination of whether or not the wetland is located in a karst region. Field verification is desirable.

Directness of Measure: Low.

32. Hydroperiod (spatially dominant)

Ranking: Wetlands that are **not** permanently flooded or tidal are more likely to recharge ground water.

Rationale: Sediments are likelier to transmit water and recharge the aquifer if unsaturated (Sklash and Farvolden 1979/MN). Wetlands exhibiting recharge are less permanently inundated (Sloan 1979/ND:Pem). In the precipitation-deficit regions, ephemeral channels are a major ground water recharge mechanism (Hanks et al. 1981/AZ:R). In these areas, permanent wetlands are more likely sites of ground water discharge than recharge (Lissey 1968/MAN:Pem, 1971; McKay et al. 1979/IL:P,R; Winter and Carr 1980; Richardson and Bigler 1984/ND:Pem; Miller et al. 1985/SAS:P; Arndt and Richardson 1986/ND:Pem). Tidally influenced wetlands (e.g., marine, estuarine, and tidal riverine) usually have insufficient vertical head to recharge ground water.

Confidence in Ranking: Moderate.

Potential Importance to Function: Not applicable (used as a classification variable).

Measure: Determination of the spatially dominant hydroperiod.

Directness of Measure: Not applicable (used as a classification variable).

33. Most Permanent Hydroperiod

Ranking: Wetlands that are **not** permanently flooded or tidally influenced are more likely to recharge ground water.

Rationale: See Predictor 32 (Hydroperiod - spatially dominant). This predictor is particularly applicable in precipitation-deficit regions, where permanent water is a likely indicator of ground water discharge.

Confidence in Ranking: Not applicable (used as a classification variable).

Potential Importance to Function: Not applicable (used as a classification variable).

Measure: Determination of the wetland's most permanent hydroperiod.

Directness of Measure: Not applicable (used as a classification variable).

34. Water Level Control

Ranking: Wetlands created by a dam and located above it are more likely to recharge ground water than those located immediately downslope of a dam.

Rationale: Impoundment increases the hydraulic head, which can increase the local recharge rate at least initially after impoundment (Andrews and Anderson 1978/WI:L,P). Areas downstream of dams are likely to be dominated by ground water discharge, not recharge.

Confidence in Ranking: Low. The effect depends on the type of dam, proximity, elevation, age, wetland type, regional topography, and other factors.

Potential Importance to Function: Low.

Measure: Documentation of water control structures (including beaver dams) that impact the wetland.

Directness of Measure: Low.

35. Flooding Extent and Duration

Ranking: Wetlands with extremely variable water levels or unstable flow are more likely to be recharging ground water than those with very stable water levels.

Rationale: Wetland soils are more likely to transmit water and recharge an aquifer if unsaturated (Sklash and Farvolden 1979/MN). Water is likely to cover unsaturated soils when the wetland expands greatly during flooding events or flow is unstable (e.g., precipitation-deficit regions).

Confidence in Ranking: Low. Water-level instability can be due also to geomorphic and land cover characteristics of the contributing area.

Potential Importance to Function: Moderate.

Measure: Determination of flooding extent and duration in the wetland. Thresholds are not specific to this function and are adapted from Hendrickson et al. (1973/MI,WI,NY:R) and US Army Corps of Engineers (1980).

Directness of Measure: Low.

54. Ground Water Measurements

Ranking: Wetlands where potentiometric (water table) contours slope **sharply** away from the wetland are more likely to recharge ground water than those lacking this condition.

Rationale: See rationale for Predictor 6 (Local Topography).

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of potentiometric contours by installing piezometers.

Directness of Measure: High, when carefully interpreted by a qualified hydrologist.

59. Water Quality Anomalies

Ranking: Wetlands with water quality anomalies characterized by drastically reduced total dissolved solids, halinity, alkalinity, conductivity, and/or hardness, with increased prevalence of bicarbonates or sulfates of calcium or magnesium, are more likely to indicate recharge of ground water than those without such anomalies.

Rationale: In many arid regions, dissolved solids can be removed only by ground water outflow (recharge) or overflow. However, most potholes overflow only infrequently or not at all (Sloan 1972/ND:Pem). Also, other factors that can influence the concentration of dissolved solids, such as evapotranspiration, precipitation, and runoff, are relatively constant for potholes in a local area (Sloan 1972/ND:Pem). Dissolved solids are carried into potholes by ground water inflow (discharge). Thus, the concentration of dissolved solids in surface waters can be a reasonable index of the direction and amount of ground water flow (Toth 1971; Eisenlohr et al. 1972/ND:Pem; Sloan 1972/ND:Pem). Sloan (1972:11/ND:Pem) indicated that recharge was probable in fresh potholes, whereas discharge was likely in saline potholes. Brackish waters were influenced both by recharge and discharge. Stewart and Kantrud (1972:6/ND:Pem) indicated that, normally, freshwater areas had dissolved solid concentrations below 500 mg/l, whereas saline areas typically had dissolved solid concentrations greater than 15,000 mg/l.

Confidence in Ranking: Moderate (arid regions) to low (elsewhere).

Potential Importance to Function: Not applicable.

Measure: Comparison of water chemistry samples collected from the wetland with those from nearby areas. The 500 mg/l threshold is from Sloan (1972/ND:Pem) and Stewart and Kantrud (1972/ND:P) and is applicable mainly to wetlands in glaciated regions with small watersheds and no inlets or outlets (i.e., *prairie* potholes).

Directness of Measure: Low (if no anomalies are found) to moderate (if anomalies are found). Conductivity measurements from a single point in time are not, by themselves, a good indicator due to dilution effects of runoff, evapotranspiration, and local variability in soil chemistry.

60. Water Temperature Anomalies

Ranking: Wetlands with drastic natural thermal anomalies are more likely to be ground water discharge areas, not recharge areas.

Rationale: Ground water temperatures are more stable from season to season than are surface water temperatures. They also tend to be cooler in summer and warmer in winter than ambient conditions. Thus, discharge of ground water to wetlands may create water temperature anomalies when compared with other wetlands in the area, and these anomalies may be used as an indicator of the probability of ground water **discharge** to a wetland (Benson 1953/MI:R; Toth 1971; Andrews and Anderson 1978/WI:L,P; Hennings 1978/WI:L; Bilby 1984). Wetlands discharging ground water are not likely to be recharging as well.

Confidence in Ranking: Moderate. Anomalies also can be due to local snowmelt, water depth, turbidity, differential heating of sediments, water velocity, shading, or wind buffering. Thus, interpretation must be cautious.

Potential Importance to Function: Not applicable.

Measure: Determination of atypical thermal conditions as evidenced by locally cooler and less variable temperatures in summer and warmer or less variable temperatures in winter.

Directness of Measure: Moderate.

62. Underlying Strata

Ranking: Wetlands underlain by thick and unstratified porous materials such as cobble-gravel or sand are more likely to recharge ground water than are those underlain by stratified nonporous sediments.

Rationale: Sediment porosity is essential to movement of water between the wetland and the aquifer. The presence of even thin layers of low-permeability material, such as silts and clays, can retard recharge. Inhibition of recharge or discharge by silt has been reported by Cooke et al. (1973/OH:L), and the application of clay to sediments is used as a technique for managing seepage losses (Born et al. 1979/WI:L).

Larson (1976/*MA) gives the following probabilities for wetlands contributing large quantities of water to an underlying aquifer with respect to permeable sediment thickness:

- High where saturated sediments are greater than 50 feet thick.
- Moderate-high where saturated sediments are 25 to 30 feet thick.
- Low where saturated sediments are less than 25 feet thick.

The New York State Department of Environmental Conservation (1980) uses 10 feet as the minimum thickness of pervious sediments necessary for significant ground water interchange. Novitzki (1981/WI) noted net annual recharge occurring in a wetland where only a few feet of glacial drift overlay sandstone.

Confidence in Ranking: High.

Potential Importance to Function: Very high.

Measure: Determination of underlying material from a geologic map.

Directness of Measure: High (if good data are available and are carefully interpreted by a qualified hydrologist).

63. Discharge Differential

Ranking: Ground water recharge is more likely in those wetlands where inflow exceeds outflow (after accounting for evapotranspiration) than in wetlands where outflow exceeds inflow.

Rationale: Diminished surface water outflow (e.g., in "losing streams") suggests water was lost via recharge, or perhaps evapotranspiration (Leopold and Miller 1961/R; Vecchioli et al. 1962/NJ:fo; Harrison and Clayton 1970/AK:R; Lissey 1971/Man:Pem; Andrews and Anderson 1978/WI:L,P).

Confidence in Ranking: Moderate. In areas of karst terrain, disappearing streamflows do not necessarily indicate recharge of deep aquifers.

Potential Importance to Function: Not applicable.

Measure: Determination of the discharge differential via hydrographs.

Directness of Measure: Moderate. An apparent loss of flow between inlet and outlet could be due to evapotranspiration, shallow infiltration, or lateral flow, rather than recharge. Moreover, the error of estimate (of flow) is often greater than the expected rate of loss due to recharge (Winter 1981/*).

3.2 Ground Water Discharge

1. Climate

Ranking: With respect to ground water discharge, climate is a classification variable rather than a predictor (i.e., if the wetland is in a precipitation-deficit region, one set of predictors is used, whereas if precipitation exceeds evaporation, another set is used).

Rationale: Not applicable.

Potential Importance to Function: Not applicable.

Measure: Identification of precipitation-deficit regions.

Directness of Measure: Not applicable.

5. Wetland/Watershed Area Ratio

Ranking: A large wetland with a proportionately small watershed may indicate subsidization of its water budget by ground water disharge. The probability of ground water discharge occurring may thus increase as the wetland/watershed area ratio increases.

Rationale: Williams (1968) observed that a small wetland situated in a large watershed favored ground water **recharge**, because surface water inflow from a large watershed was sufficient to create a water mound conducive to recharge.

Confidence in Ranking: Low (lower elevations) to moderate (higher elevations).

Potential Importance to Function: Not applicable.

Measure: Determination of the percentage of the watershed acreage occupied by the wetland. The thresholds used are arbitrary.

Directness of Measure: Low. A more meaningful measure would be ratio of wetland volume to runoff volume. Apparent water surpluses (i.e., large ratio) could be attributable as well to differences in soil permeability, geomorphology, and land cover of the watersheds being compared.

6. Local Topography

Ranking: Ground water discharge is more likely to occur in areas where the topographic relief is characterized by a sharp downslope **toward** the wetland (i.e., the wetland is located at the toe of a slope) and/or the wetland is located near a geologic fault or at the base of a local horizontal gradient of decreasing soil permeability.

Rationale: As discussed for ground water recharge (Predictor 6, Local Topography), the hydraulic gradient for ground water movement is influenced by the slope of the water table with respect to the wetland. The slope of the water table usually roughly parallels the topography of the land surface (Fetter 1980/*). Thus, when local topography slopes sharply toward the wetland, the water table will also likely slope toward the wetland as well, resulting in a hydraulic gradient favorable for discharge of ground water into the wetland (Toth 1971).

Contact with ground water is likelier to occur where the water table (potentiometric) contour changes sharply at geologic, topographic, or edaphic interfaces, e.g., "breaks in slope" (Maxey 1968/NV; Winter 1977, 1981; Carter and Novitzki 1988/VA,NY:P). The more numerous, dissimilar, and spatially abrupt these interfaces are, the likelier it is that significant contact will exist between ground and surface water (Freeze and Witherspoon 1967/*). Ground water discharge is particularly likely to occur at downgradient shifts from coarser to finer textured soils or at intrusions of aquicludes such as shale layers (Meyboom 1966/MAN, Van Voast and Novitzki 1968/MN, Dunne and Leopold 1978:215-217, Pionke et al. 1986/PA:fo).

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Identification of special topographic features such as playas, topographic divides, significant elevational changes, geologic faults, or changes in hydraulic conductivity.

Directness of Measure: Low.

8. Inlets, Outlets

Ranking: A wetland with a permanent outlet and no inlet is more likely to discharge ground water than one with other combinations of inlets and outlets.

Rationale: Continuous discharge of water (i.e., permanent outlet) without surface water feeding the wetland through an inlet suggests an internal source of ground water (e.g., springs), especially in arid regions and during droughts (Born et al. 1979/WI:L, Novitzki 1979/WI).

Confidence in Ranking: Moderate.

Potential Importance to Function: Not applicable.

Measure: Identification of inlets and outlets for surface flow.

Directness of Measure: Low. Other internal sources of water (e.g., snowmelt) must first be discounted. Field inspection, rather than reliance on topographic maps, is preferred.

9. Constriction

Ranking: This predictor applies specifically to cases where no inlet is present, but a permanent outlet exists. Wetlands in which outflow originates from sources other than precipitation or snowmelt are most likely to discharge ground water.

Rationale: If precipitation or snowmelt is not the primary source of outflow, then ground water discharge must be occurring.

Confidence in Ranking: Not applicable.

Potential Importance to Function: Not applicable.

Measure: Determination of whether precipitation or internal snowmelt can be discounted as the source of outflow from the wetland.

Directness of Measure: Low.

12. Vegetation Class/Subclass (Primary)

Ranking: Riverine wetlands dominated by aquatic bryophytes (mosses and liverworts) are most likely to discharge ground water.

Rationale: Because many bryophytes prefer water rich in carbon dioxide, they sometimes proliferate near springs (Hynes 1970/*R).

Confidence in Ranking: Low.

Potential Importance to Function: Not applicable.

Measure: Determination of the dominant vegetation class and subclass of the wetland.

Directness of Measure: Low.

21. Land Cover of the Watershed

Ranking: Wetlands with unpaved watersheds are more likely to allow ground water discharge to occur.

Rationale: Extensive paved surfaces, topographic disruptions, and the presence of wells are associated with human development that lowers the potentiometric contours. Lowered or diversified potentiometric contours enhance the likelihood of **recharge,** not discharge (Fetter 1980/*).

Confidence in Ranking: Low.

Potential Importance to Function: Low.

Measure: Determination of the major land cover type of the wetland's watershed.

Directness of Measure: Low.

23. Ditches/Canals/Channelization/Levees

Ranking: Wetlands **without** ditches, canals, levees, or similar artificial features (which cause surface water to leave faster than it would if these features were not present) are more likely to discharge ground water than altered wetlands.

Rationale: Surface drainage alterations generally lower the water table, lower minimum water levels, and drain wetlands (Griswold 1978, Verry and Boelter 1979/*P). The flattened hydrologic gradient can result in reduced ground water exchange.

Confidence in Ranking: Low. In some cases (e.g., excavation of riverine borrow pits), ditching can make the hydraulic gradient **more** favorable for discharge. For example, channelization in one case increased the stream base flow (Huish and Pardue 1978/NC:fo). The effects of any particular activity or plan depend on type of drain, ditch/canal depth and spacing, physical properties of the wetland substrate, presence or absence of vegetation, and other factors (Verry and Boelter 1979/*P, Moore and Larson 1980/MN:Pem).

Potential Importance to Function: High.

Measure: Documentation of hydrologic modifications that cause surface water to leave at faster rates than originally occurred.

Directness of Measure: Low.

32. Hydroperiod (spatially dominant)

Ranking: Wetlands with the following hydroperiods are more likely to discharge ground water: permanently flooded nontidal, intermittently exposed nontidal, saturated nontidal, artificially flooded nontidal, regularly flooded tidal, and irregularly exposed tidal or subtidal.

Rationale: Permanent surface water, especially in regions having high evaporation rates, often indicates ground water discharge into the wetland. For references, see Predictor 32 in Section 3.1. Estuarine intertidal zones adjacent to open water or tidal channels temporarily store water as tides rise, and discharge water when tides fall (Harvey et al. 1987).

Confidence in Ranking: Moderate (arid regions) to very low (nonarid regions).

Potential Importance to Function: Not applicable.

Measure: Determination of the dominant flooding regime.

Directness of Measure: Moderate.

34. Water Level Control

Ranking: Wetlands influenced by upstream impoundments are more likely to discharge ground water than those not influenced by such impoundments.

Rationale: The pressure head often increases downstream from dams. Andrews and Anderson (1978/WI:L,P) reported that creation (by impoundment) of an approximately 500-acre lake with a maximum depth of 16.5 feet resulted in a sixfold increase in discharge rates to a riverine wetland 3.2 miles away. Ground water levels stabilized within a month after the lake was filled.

Confidence in Ranking: Low. The effect depends on the type of dam, proximity, elevation, age, wetland type, regional topography, and other factors.

Potential Importance to Function: Low.

Measure: Documentation of water control structures (including beaver dams) that impact the wetland.

Directness of Measure: Low.

35. Flooding Extent and Duration

Ranking: Wetlands that have stable water levels and stable flows are more likely to discharge ground water than those with unstable conditions.

Rationale: Ground water levels are usually a dampened, lagged replicate of surface runoff amplitude and timing (Fetter 1980/*). Thus, wetlands with substantial ground water discharge should have less variable water levels. Data given by Hendrickson et al. (1973/MI,WI,NY:R) suggest that, at least in Michigan, if the ratio calculated by dividing the flow velocity (cubic feet per second), that is reached or exceeded 10 percent of the year by the typical flow velocity (cfs) that is exceeded 90 percent of the year is less than 1.5, there is a strong possibility that ground water input is major.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Determination of flooding extent and duration in the wetland. Thresholds are adapted from Hendrickson et al. (1973/MI,WI,NY:R) and US Army Corps of Engineers (1980).

Directness of Measure: Moderate.

54. Ground Water Measurements

Ranking: Wetlands where potentiometric contours do not slope away from the wetland are most likely to discharge ground water.

Rationale: The hydraulic gradient is influenced by the slope of the water table with respect to the wetland. The hydraulic gradient favors recharge when the water table slopes away from the wetland, and discharge is favored when the opposite condition exists. (See also rationale for Ground Water Recharge (Section 3.1), Predictor 6 - Local Topography.)

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Measurement of potentiometric contours by installing piezometers.

Directness of Measure: High, when carefully interpreted by a qualified hydrologist.

59. Water Quality Anomalies

Ranking: Wetlands with drastic natural water quality anomalies such as elevated carbon dioxide, alkalinity, hardness, halinity, total dissolved solids, conductivity, and/or chlorides, sulfates, or bicarbonates of sodium, iron, or manganese are more likely to discharge ground water than those without such anomalies. Those with reduced levels of the above are least likely to discharge ground water.

Rationale: See Ground Water Recharge (Section 3.1), Predictor 59 -Water Quality Anomalies.

Confidence in Ranking: Moderate.

Potential Importance to Function: Not applicable.

Measure: Comparison of water chemistry of samples collected from the wetland with those from nearby areas. The 15,000 mg/l threshold for total dissolved solids is from Sloan (1972/ND:Pem) and Stewart and Kantrud (1972/ND:P) and is probably applicable to wetlands in glaciated regions with small watersheds and no inlets or outlets (i.e., prairie potholes).

Directness of Measure: Low to moderate. Measured concentrations may reflect only the time elapsed since higher or lower water levels of the wetland (i.e., concentrations generally decline with duration of inundation, Beauchamp and Kerekes 1980/NB:P).

60. Water Temperature Anomalies

Ranking: Wetlands with water temperatures that deviate drastically from ambient temperatures are more likely to discharge ground water than those not having such anomalies.

Rationale: Ground water temperatures are more stable from season to season than are surface water temperatures. Thus, discharge of ground water to wetlands may result in water temperature anomalies when compared with other wetlands in the area. Thus, water temperature anomalies may be used as an indicator of the relative magnitude of ground water discharge to a wetland (Benson 1953/MI:R; Andrews and Anderson 1978/WI:L,P; Hennings 1978/WI:L).

Confidence in Ranking: Moderate.

Potential Importance to Function: Not applicable.

Measure: Determination of atypical thermal conditions.

Directness of Measure: Low (if anomalies are absent) to moderate (if anomalies are present).

63. Discharge Differential

Ranking: Ground water discharge is most likely to occur in wetlands where outflow (after accounting for evapotranspiration) exceeds inflow.

Rationale: Sharply increasing surface water outflow (e.g., in "gaining streams") indicates that water was gained via discharge (Leopold and Miller 1961/R, Harrison and Clayton 1970/AK:R).

Confidence in Ranking: oderate.

Potential Importance to Function: Not applicable.

Measure: Determination of the discharge differential via hydrographs.

Directness of Measure: Moderate. An apparent increase in flow between inlet and outlet could be due to a sharply increasing contributing area, lateral flow, or differences in the land cover and permeability of soils that directly adjoin the wetland. Moreover, the error of estimate (of flow) is often greater than the expected rate of gain due to ground water discharge (Winter 1981/*).

Confidence in Ranking: Moderate.

Potential Importance to Function: Not applicable.

Measure: Determination of the discharge (flow) differential via hydrographs.

Directness of Measure: Moderate.

3.3 Floodflow Alteration

1. Climate (predictor for effectiveness)

Ranking: Wetlands located in precipitation-deficit regions are more likely to alter floodflows than those in more humid regions.

Rationale: Wetlands can store more flood water if the water level in the wetland is initially low. This condition is most likely to exist in precipitation-deficit regions (i.e., where evaporation exceeds precipitation).

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of whether or not the wetland is located in a precipitation-deficit region.

Directness of Measure: Low. Direct measurement of evapotranspiration and sediment saturation is preferred.

2. Acreage (predictor for effectiveness)

Ranking: Large wetlands are more likely to alter floodflows.

Rationale: The ability of a wetland to alter floodflows depends on its storage capacity and hydraulic length, which are determined by its area, depth, and sediment type.

Using Conger's (1971) regression analysis of watershed characteristics from Wisconsin watersheds, Novitzki (1979/WI) determined that wetland acreage was an important factor influencing flood storage. These analyses indicated that if a watershed has 15 percent of its area in wetlands and lakes, flood peaks will be 60 to 65 percent lower than they would be in the absence of the wetland and lake area. If wetlands or lakes occupy 30 percent of the watershed, there will be a further reduction in flood peaks up to 75 to 80 percent. However, 50 percent of the flood peak reduction results from the first 5 percent of the lake and wetland area in the watershed. Simulated peak flows increased by more than 20 percent (measured at the wetland outlet) when more than 60 percent of the wetlands above a floodable area in Massachusetts were filled (Ogawa and Male 1983/*MA).

Flood peak attenuation is also indicated by a modeling study of palustrine and lacustrine wetlands in Florida (Ammon et al. 1981/FL:P108). These authors projected that 5 percent of the emergent wetlands accounted for up to 70 percent attenuation of the mean annual flood. Although the incremental gain in terms of flood peak attenuation was less dramatic once wetland acreage exceeded 10 percent, 95-percent flood peak attenuation was indicated where only 15 percent of the watershed was wetland.

A more conservative estimate from the prairie region suggests that drainage of wetlands might result in a 7- to 20-percent increase in flood peaks (Dybvig and Hart 1977). Storage capacity of prairie wetlands can be 48 inches per acre (Cernohous 1979/ND), depending on the degree of saturation of wetland soils.

For detention depressions, Tourbier and Westmacott (1974/*) recommended the following storage capacities:

Storm Frequency years	Depth of Basin feet	Acreage Required per Acre of Development
2	5 10 15	0.16 0.08 0.05
5	5 10 15	0.21 0.10 0.06
10	5 10 15	0.26 0.13 0.13
25[1]	5 10 15	0.35 0.17 0.12
50	5 10 15	0.42 0.21 0.14
100	5 10 15	0.50 0.25 0.17

[1] Flores et al. (1981) found the 25-year storage design policy most efficient in reducing peak flows for a given storage volume.

Confidence in Ranking: High.

Potential Importance to Function: High. Position in watershed and geomorphic characteristics are also important.

Measure: Determination of wetland surface area.

Directness of Measure: Moderate. Measurement of volume, water yield, and capillary potential of sediments is preferred. Hydraulic length of a watershed can be estimated from watershed size.

5. Wetland/Watershed Area Ratio (predictor for opportunity)

Ranking: Wetlands with watersheds that contain little wetland area above the wetland of interest, or those with a large watershed in relation to the wetland, are most likely to have an opportunity for floodflow alteration.

Rationale: The more wetland area located in the watershed above the assessment area wetland, the less opportunity the assessment area wetland has to provide additional floodflow alteration. Conversely, if the watershed is small in relation to the wetland, then the watershed would yield less water and there is proportionately less opportunity or need for floodflow alteration.

Confidence in Ranking: High.

Potential Importance to Function: Moderate. Runoff depends also on soil type, land cover, and other factors.

Measure: Determination of wetland/watershed ratio as directed in Volume II.

Directness of Measure: Low.

8. Inlets, Outlets (predictor for effectiveness)

Ranking: Wetlands without outlets are more likely to alter floodflows than those with outlets.

Rationale: Wetlands with no surface outlet hold water that would otherwise contribute to surface flow (Carter et al. 1979/*, Novitzki 1981/WI).

Confidence in Ranking: High.

Potential Importance to Function: High. Wetlands lacking outlets are highly correlated with low-gradient landscapes, which on a regional level may be the more influential factor in mediating floodflows. Attenuation of flood surges increases significantly when channel slope (parallel to flow) is less than 0.001 (Grushevsky 1967/EU:R).

Measure: Identification of inlets and outlets for surface flow.

Directness of Measure: Low. Field checking is preferred because intermittent inlets and outlets are sometimes not indicated on topographic maps.

9. Constriction (predictor for effectiveness)

Ranking: Wetlands with an unconstricted inlet and a constricted outlet are more likely to alter floodflow than those with unconstricted outlets.

Rationale: Constricted outlets hold back flood water and increase the storage capacity of a wetland, thus potentially reducing downstream flood peaks (Carter et al. 1979/*).

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of presence or absence of constrictions to surface flow.

Directness of Measure: Low. Field checking is preferred because intermittent inlets and outlets are sometimes not indicated on topographic maps.

10. Wetland System (predictor for opportunity and effectiveness)

Ranking: Riverine (tidal), estuarine, and marine systems are less likely to have an opportunity for, and be effective at, altering floodflows.

Rationale: The effects of marine, estuarine, and tidal riverine wetlands are usually overshadowed by tidal conditions (Carter et al. 1979/*). Typically, their sediments are completely saturated and incapable of storing additional moisture. Also, by virtue of their position (i.e., low in the watershed, with little property downstream), whatever storage or desynchronization effect they exert is of little economic consequence.

Confidence in Ranking: High.

Potential Importance to Function: Moderate.

Measure: *Determination of the wetland classification according to* Cowardin et al. (1979/*).

Directness of Measure: Moderate.

11. Fringe Wetland or Island (predictor for effectiveness)

Ranking: Fringe wetlands, or those associated with islands, are less likely to alter floodflows.

Rationale: Because fringe wetlands are, by definition, small relative to some adjacent body of water (i.e., a channel or standing body of water), they are relatively less important in altering floodflows than nonfringe wetlands. Frictional drag and the potential for desynchronization of floodflows are likely to be less in fringe wetlands than in wetlands that are wide enough to intercept most of the flow passing through them.

Confidence in Ranking: Low.

Potential Importance to Function: Moderate.

Measure: Determination of whether the wetland is a fringe wetland or is associated with an island.

Directness of Measure: Moderate.

12. Vegetation Class/Subclass (Primary) (predictor for effectiveness)

Ranking: Wetlands with forested or scrub-shrub vegetation are more capable of altering floodflows than are aquatic bed, moss, or emergent wetlands.

Rationale: Vegetation helps desynchronize flows by increasing channel roughness. However, vegetational resistance rapidly diminishes as the water depth becomes greater than the vegetation height (Camfield 1977/em). In descending order of frictional resistance, terrestrial vegetation may be ranked as closely spaced trees, scattered trees, brush or low bush trees, dense grass, and sparse grass (inferred from Aldridge and Garrett 1973/AZ:R, Petryk and Bosmajian 1975).

One simulation study indicated that removal of woody debris occupying 16 percent of a channel would reduce flood stage at the wetland outlet by 0.1 meter (Taylor and Barclay 1985). Johnson and Senter (1977) found that a 20 percent variation in the roughness coefficient would not affect flood height in the lower Ohio River. Burkham (1976) found that the removal of woody debris from a floodplain sufficient to lower the roughness coefficient by 0.026 resulted in a 1.2-foot decrease in depth and an increase of 0.8 foot per second velocity during floods, while an increase in the roughness coefficient of only 0.008 caused a stage increase of 0.4 foot and a 0.2-fps reduction in velocity.

Riparian forests may dissipate water through evapotranspiration (Brown 1981/FL:Pfo), although they simultaneously reduce evaporation by reducing water temperature and wind. Up to 63 percent of the total water input (over twice the percentage transpired from agricultural fields) may be removed by the evapotranspiration of riparian forest (Peterjohn and Correll 1984/MD:fo). Intermediate successional stages usually experience the greatest evapotranspirative losses (Bormann and Likens 1979/NH:R). Emergent vegetation with little underlying plant litter experiences greater evapotranspirative losses than evergreen shrub, willow-birch, and moss-covered wetlands (Kadlec 1987/*, Kadlec et al. 1988/*). Soil saturation is a key determinant.

Woody vegetation may also increase bank storage and remove sediment from runoff, thus preserving the flood water storage capacity of adjoining deepwater areas. Although often offset by "fog drip," interception of incoming precipitation by the wetland vegetation's canopy also can beneficially increase the desynchronization effect. Canopies of black spruce can intercept and potentially evaporate 15 to 20 cm of precipitation per year (Boelter and Verry 1977/MN:P).

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate. Basin morphology is usually a better predictor of flow desynchronization than are vegetation roughness and evapotranspirative potential.

Measure: Determination of the dominant vegetation class and subclass of the wetland.

Directness of Measure: Moderate. Field measurement of roughness coefficients and evapotranspiration is preferred (see Daniel 1976/AL:R, Winter 1981/*, Arcement and Schneider 1984/R,P, Dolan et al. 1984/FL:Pfo).

15. Vegetation/Water Interspersion (predictor for effectiveness)

The influence of vegetation-water interspersion as a predictor for wetland functions depends upon two factors: (1) the amount and interspersion of vegetation present and (2) the type of water flow through the wetland. Therefore, vegetation-water interspersion is defined with two predictors.

15.1 Vegetation Interspersion

Ranking: Wetlands with dense stands of vegetation with little interspersed *open water are more likely to alter floodflows.*

Rationale: Channel roughness and thus the ability to retain flood waters increases with increasing vegetation density. Flood waters are also partially dissipated by evapotranspiration. For further discussion and references, see Predictor 12 above.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Determination of relative vegetation-water interspersion.

Directness of Measure: Moderate.

15.2 Sheet Versus Channel Flow

Ranking: Wetlands in which water occurs primarily as sheet flow are more likely to alter floodflow than those where water occurs primarily as channel flow.

Rationale: Frictional resistance, and thus the potential for desynchronization of floodflows, is greater where flow is mostly sheet flow (Hewlett and Hibbert 1965/*, Patton and Baker 1976/IN,TX,CA,UT). See Predictor 12 above.

Confidence in Ranking: Moderate.

Potential Importance to Function: Low. Basin morphology is usually a better predictor than is wetland vegetation.

Measure: Determination of type of flow occurring in the wetland.

Directness of Measure: Low (see Predictor 12 above).

21. Land Cover of the Watershed (predictor for opportunity)

Ranking: Wetlands in watersheds with mostly impervious land cover are more likely to have an opportunity for floodflow alteration than those in watersheds that are predominantly forested or scrub-shrub.

Rationale: Impervious surfaces associated with developed environments provide greater opportunity for floodflow alteration because impervious surfaces and hydraulically efficient drainage systems cause earlier peaking and higher outflows than what would occur on vegetated lands (Martens 1968, Wycoff and Pyne 1975/FL, Harr 1976/OR:fo, Chang and Watters 1984/TX:fo, Inman 1987/GA:R). Full urban development of a 1-square mile watershed would increase peak flows by 3 and 4 times for the 100- and 5-year frequencies, respectively. The maximum (i.e., 40 percent) increment in hydrologic impact of development would come as the watershed is converted from 50 to 75 percent urbanized (Flores et al. 1981). Urbanization (from no development to 77 percent urban) of a 100-square mile watershed was shown by simulation to increase annual runoff from 5.3 to 5.9 inches (Arnold et al. 1987).

Evapotranspirative losses presumably are greater in forested and scrub-shrub wetlands than in other vegetation types (Lee 1980:164/*), thus decreasing the amount of runoff. Croft and Monninger (1953) observed evapotranspirative losses of 20 and 10 cm for aspen cover types and herbaceous cover, respectively. However, Odum (1971:376/FL:Efo) indicated that evapotranspiration is a complex process and is not necessarily greater in forests than in grasslands.

Confidence in Ranking: Moderate. One study has indicated that, in some regions, forest cover has little influence on streamflows (e.g., Thomas and Benson 1970).

Potential Importance to Function: High. The land use and relative proportions of different cover types in the watershed and the proximity of cover types to the wetland are also important. Land cover (particularly via its effects on evapotranspiration) can overshadow other landscape components (e.g., watershed geomorphology) in its effects on water budgets (Sophocleous and McAllister 1987).

Measure: Determination of the cover type found on a majority of the wetland's watershed.

Directness of Measure: Moderate.

22. Flow, Gradient, Deposition (predictor for effectiveness)

Ranking: With respect to floodflow alteration, gradient is used in the interpretation keys as a classification variable rather than a predictor (i.e., if flow is present, one set of predictors is used, whereas if flow is absent, another set of predictors apply).

Confidence in Ranking: Not applicable.

Potential Importance to Function: Not applicable.

Measure: Documentation of indicators of flow such as a channel, presence of an outlet and inlet, tidal influence, scour lines, sediment deposition, gage data, etc.

Directness of Measure: Not applicable.

23. Ditches/Canals/Channelization/Levees (predictor for effectiveness)

Ranking: Wetlands having hydrologic alterations that cause water to leave faster than originally occurred are less likely to alter floodflows significantly.

Rationale: Channelization of distributaries or drainage increases the transport of flood waters away from the wetland and, as a result, reduces the potential for significant storage and desynchronization of floodflows (DeBoer and Johnson 1971/IA:Pem, Wycoff and Pyne 1975/FL, Huish and Pardue 1978/NC:fo). Drainage density (the length of drainage canals per unit area of watershed) can be a stronger determinant of watershed response than the condition of the floodplain, and can predict runoff volume (Bedient et al. 1976/FL). Model simulations (Moore and Larson 1980/MN:Pem) suggest that draining wetlands can increase storm runoff volumes by 50 to 590 percent for small depressional watersheds. These simulations also indicate that channelization can increase peak flows by 100 to 200 percent at the watershed outlet. In contrast, the meanders that characterize unchannelized areas can increase channel roughness by as much as 30 percent (Chow 1959/*R). In some rivers this would be the hydraulic equivalent of increasing the depth by 20 percent and decreasing the velocity by 30 percent (Burkham 1976/AZ:fo).

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of evidence of alteration of the wetland's hydrologic regime, including ditches, canals, levees, or similar artificial features that cause water to leave the area at an accelerated rate. Meandering is considered "minor" for ratios of 1.0 to 1.2, appreciable for ratios of 1.2 to 1.5, and "severe" for ratios of 1.5 and greater (Arcement and Schneider 1984/R,P).

Directness of Measure: Low.

24. Soils (predictor for effectiveness and opportunity)

Ranking: Watersheds with mostly impervious soils provide wetlands with proportionately more runoff and thus greater opportunity for floodflow alteration. Wetlands most likely to be effective at altering floodflows have soils with rapid infiltration rates.

Rationale: Watersheds with impervious soils or surfaces that impede infiltration (e.g., pavement in urban areas) produce greater surface runoff than watersheds having permeable soils. Increased runoff provides greater opportunity for wetlands to alter floodflows.

***Confidence in Ranking**: High.*

Potential Importance to Function: Low (effectiveness); high (opportunity). The effect depends also on location, soil, cover type, and other factors. Soil storage capacity is usually small compared to surface storage. However, Hewlett and Hibbert (1965/*) found that up to 50 percent of runoff from storms in excess of 6 inches was stored in unsaturated soils of 15 small Piedmont-Appalachian watersheds. Soil storage may also depend on the capillary characteristics of the soil (Heliotis and DeWitt 1987/MI:P).

Measure: Determination of watershed and wetland soil infiltration rates from soil surveys.

Directness of Measure: Low (effectiveness); moderate (opportunity).

31. Water/Vegetation Proportions (predictor for effectiveness)

Ranking: Wetlands with a high proportion of vegetation coverage will be more capable of altering floodflows.

Rationale: Vegetation slows flood waters by creating frictional drag in proportion to stem density (see discussion and references under Predictor 12). Also, wetland zones without permanent standing water (zone A as defined in

Volume II) are important not only because of the frictional resistance offered by vegetation there, but also because underlying sediments are more likely to be unsaturated and thus better able to assimilate flood water.

Confidence in Ranking: High.

Potential Importance to Function: Moderate.

Measure: Determination of the vegetation coverage in wetland zones as defined in Volume II.

Directness of Measure: Low. Although the technique in Volume II specifies emergent vegetation, the density of all robust vegetation would be a better predictor. Measurement of stem density and hydraulic roughness would give even better estimates of floodflow alteration potential (Hewlett and Hibbert 1965/*, Patton and Baker 1976/IN,TX,CA,UT).

32. Hydroperiod (spatially dominant) (predictor for effectiveness)

Ranking: Wetlands without permanent standing water, especially those with a low probability of ever having standing water, are more likely to alter floodflows than are permanently inundated wetlands.

Rationale: Greatest flood water storage occurs when no water is initially present in the wetland or sediment pores.

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of the wetland's dominant flooding regime.

Directness of Measure: Moderate.

35. Flood Extent and Duration (predictor for effectiveness)

Ranking: Wetlands capable of expanding their surface water acreage substantially and for long periods of time are most likely to alter floodflows.

Rationale: Alteration of floodflow is partially accomplished by retention of flood waters up to a geomorphically determined threshold of expansion (Bhomik and Demissie 1982/*). In some regions, the amount of water stored in wetlands during flood events may expand fivefold or more relative to dry season conditions (Rundquist et al. 1987/NE:L).

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of duration and extent of flooding in relation to threshold levels. Thresholds are not specific to this function. They were adapted in part from US Army Corps of Engineers (1980).

Directness of Measure: Low. Measurement of volume of water storage (above-ground, in sediment pores, and capillary effects) is a better predictor.

63. Discharge Differential (predictor for effectiveness)

Ranking: Wetlands that have higher inlet flood peaks than occur just below their outlets are more likely to alter floodflows.

Rationale: Lower flood peaks at wetland outlets are direct evidence for floodflow alteration.

Confidence in Ranking: High.

Potential Importance to Function: Not applicable.

Measure: Determination of discharge differential and interpretation using hydrographs.

Directness of Measure: High. More than one storm should be measured. The error of measurement may be proportionately large, especially in small wetlands.

3.4 Sediment Stabilization

7. Gradient

Ranking: Wetlands having steep gradients are more likely to have an opportunity to stabilize sediments.

Rationale: Steeper gradients imply greater velocities, greater erosiveness, and greater erosion potential.

Confidence in Ranking: High.

Potential Importance to Function: Moderate. Sediment cohesiveness, ground water influx rates, and suspended/deposited sediment equilibria are at least as important.

Measure: Comparison of the wetland's channel gradient (as calculated from topographic maps) with threshold gradients known from empirically based hydraulic equations to estimate depositional conditions.

Directness of Measure: Moderate. Field determinations of gradient, using surveying equipment, are preferred.

12. Vegetation Class/Subclass (Primary)

Ranking: Wetlands with predominantly forested and scrub-shrub vegetation are more likely to stabilize sediment than are those with predominantly aquatic bed vegetation.

Rationale: Plants dissipate erosive forces by creating frictional drag (see Section 3.3, Predictor 12, for discussion and references). Vegetational resistance rapidly diminishes as the water depth becomes greater than the vegetation height (Camfield 1977/em). Trees and scrub-shrub vegetation are important in creating frictional drag because they are rigid, persistent, and tall enough to penetrate the entire water column during seasonal flooding. Scrub-shrub vegetation can reduce flow velocities in the vicinity of a vegetated bank by as much as 50 percent (Klingeman and Bradley 1976).

Plants anchor shorelines and floodplain sediments by binding soil with their roots (Sigafoos 1964, Pfankuch 1975/*R, Allen 1979/*L,R, Nunnally and Keller 1979/*R, Sheridan and Hubbard 1987/GA:R). In this regard, trees and shrubs are very important because they have the deepest roots, layered canopies, high regenerative capacity, and a long lifespan. The length of bank protected by tree roots is about 5 times the diameter of the tree (Nunnally and Keller 1979/*R). Mosses can be important because they can sometimes blanket fine sediment substrates. Emergents are also important in binding the soil. In fact, Murgatroyd and Ternan (1983) reported instances of increased bank erosion associated with regrowth of floodplain forests that shaded out the blanketing herbaceous cover.

Confidence in Ranking: Low.

Potential Importance to Function: Moderate. The sediment erosion rate may be more closely related to watershed size and percentage of silt-clay sediment than to the presence of vegetation (Daniel 1971/IN:R; Hooke 1979/SC, 1980).

Measure: Determination of dominant vegetation class and subclass found in the wetland.

Directness of Measure: Low. Species-specific analysis is a better indicator.

15. Vegetation/Water Interspersion

The influence of vegetation-water interspersion as a predictor for wetland functions depends upon two factors: (1) the amount and interspersion of vegetation present and (2) the type of water flow through the wetland. Therefore, vegetation-water interspersion is defined with two predictors.

15.1 Vegetation Interspersion

Ranking: Wetlands with dense, extensive stands of vegetation are more likely to contribute to sediment stabilization than are those with no vegetation.

Rationale: As previously discussed, roots of wetland plants anchor wetland soils and, thus, reduce erosion. By creating frictional resistance (bottom drag), wetland plants may also reduce flow velocities and cause more rapid wave dissipation, thus reducing potentially erosive forces. Therefore, the denser and more extensive the vegetation stand (i.e., lower interspersion), the more likely the vegetation will stabilize sediments.

Confidence in Ranking: Moderate. Vegetated tidal areas may attract animals that stimulate erosion by their activities (e.g., Serodes and Troude 1984/QUE:Eem).

Potential Importance to Function: High.

Measure: Determination of relative interspersion of vegetation and water.

Directness of Measure: Low. Field measurement of stem and root volume is preferred.

15.2 Sheet Versus Channel Flow

Ranking: Wetlands where water enters in a channel and then spreads out over a wide area are most likely to stabilize sediments.

Rationale: Frictional resistance is greater, and thus the potential erosiveness lower, where flow through the wetland is mostly sheet flow (e.g., Brown 1985).

Confidence in Ranking: High.

Potential Importance to Function: Moderate.

Measure: Determination of whether or not water enters the wetland in a channel and then spreads out over a wide area. Care must be exercised to be sure that braided flow (a sign of increased erosion in some systems) is not interpreted as sheet flow.

Directness of Measure: Low.

19. Fetch/Exposure

Ranking: Wide fetch provides greater opportunities for wave generation; wetlands that intercept waves and thus protect nearby shores are more likely to stabilize sediment.

Rationale: Erosive forces such as waves and storm surges are likelier to become amplified if the fetch is great (Keddy 1983/ONT:Lem). Wetlands that intercept waves will reduce the fetch for nearby shores and, as a result, reduce their erosion potential. Opportunity for erosion of salt marshes was considered to be greatest where fetch exceeded 3 miles, and least where it was less than 2 miles (Knutson et al. 1982/VA:Eem). However, in another study, a distance of only 0.5 mile was needed to generate waves capable of resuspending wetland sediments (Carper and Bachmann 1984/IA:Pem). Wave height can be predicted given the fetch distance, depth, and wind velocity (US Army Corps of Engineers 1977). See also Section 3.5, Predictor 19, for thresholds and references.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate. Wind direction and water depth are important interrelated factors, but are difficult to assess from a single visit.

Measure: Identification based on basin shape and position of nearby areas sheltered by the wetland.

Directness of Measure: Moderate.

22. Flow, Gradient, Deposition

This predictor can be separated into two predictors relating to sediment stabilization: flow and scouring. Both of these are indicators of opportunity for sediment stabilization rather than predictors of a wetland's sediment stabilization effectiveness.

22.1 Flow

Ranking: Wetlands with flowing water are more likely to have an opportunity to stabilize sediment.

Rationale: Current velocity is a key contributor to erosion (Karr and Schlosser 1977/*).

Confidence in Ranking: High.

Potential Importance to Function: Moderate. Sediment cohesiveness and suspended/deposited equilibria are at least as important.

Measure: Determination of flow, or indications of flow, such as gage data, scour lines, sediment deposition, etc.

Directness of Measure: Moderate.

22.2 Scouring

Ranking: Wetlands that are scoured indicate greater opportunity for sediment stabilization than those that are unscoured.

Rationale: Scouring indicates high flows capable of eroding the shoreline. Thus, greater opportunity exists near scoured areas for sediment stabilization to operate (Pfankuch 1975/*R).

Confidence in Ranking: High.

Potential Importance to Function: Not applicable.

Measure: Observation of evidence of long-term erosion in the wetland.

Directness of Measure: Moderate.

23. Ditches/Canals/Channelization/Levees

Ranking: Wetlands having modified distributaries that allow surface waters to flow at a faster rate are likely to have greater opportunity to stabilize sediment.

Rationale: Flow velocity increases in channelized areas. Erosiveness increases with increasing flow velocities, thus providing greater opportunity for sediment stabilization to occur. Erosiveness is further amplified by the relatively rapid fluctuations of flows that occur in channelized areas (Laser et al. 1969/IA:R; King 1973/IA; Griswold 1978; Huish and Pardue 1978/NC:fo; Schmal and Sanders 1978/WI:R,P; Zimmer and Bachman 1978/IA:R; Stern and Stern 1980/*; Scaife et al. 1983).

Confidence in Ranking: Low.

Potential Importance to Function: Moderate.

Measure: Determination of presence of ditches, canals, levees, or similar artificial features that cause surface water to leave at a faster rate than if such features were not present.

Directness of Measure: Low.

25. Sediment Sources

Ranking: Wetlands surrounded by potentially erosive conditions are more likely to have an opportunity for sediments to be stabilized.

Rationale: Erosive conditions must be present for sediment stabilization to occur.

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Documentation of potentially erosive conditions including drastic water-level fluctuations, steep slopes, frequent boat wakes, channelized tributaries, etc., that significantly elevate suspended solid levels.

Directness of Measure: High.

31. Water/Vegetation Proportions

Ranking: Wetlands having no robust vegetation in zones where surface water occurs are less likely to stabilize sediment.

Rationale: Robust vegetation, especially if persistent, is important for sediment stabilization because it typically produces relatively extensive stands with high stem densities. Knutson et al. (1981/Eem) defined effective marshes as those covering at least 60 percent of the shoreline. Allen (1979/*L,R) discussed the advantages of several native and introduced emergent species for erosion control.

Confidence in Ranking: Moderate.

Potential Importance to Function: High.

Measure: Determination of the aerial coverage of robust vegetation in wetland zones with surface water.

Directness of Measure: Moderate. Stem and root volume is a better measure than percent coverage (Gleason et al. 1979/Eem), and water depth is also a key factor. Quantitative estimates are possible using equations of Dean (1979/*).

34. Water Level Control

Ranking: Wetlands located just downstream from large impoundments (higher than 20 feet at the outlet) are more likely to have an opportunity to stabilize sediment.

Rationale: Large impoundments are usually equipped with structures for rapidly releasing water (i.e., spillways). Rapid water-level fluctuations are more erosive than stable flow conditions, and often contribute to bank failures (Pfankuch 1975/*R, Nunnally and Keller 1979/*R, Ritter 1986/*). Also, impoundments allow sediment loads to settle out, causing disequilibrium of downstream sediments and increasing the erosiveness of water as it enters a wetland (see Section 2.3.2).

Confidence in Ranking: High.

Potential Importance to Function: Moderate. The effect depends also on impoundment type, dam size, proximity, age, and other factors.

Measure: Documentation of upstream impoundments that impact the wetland.

Directness of Measure: Low.

36. Vegetated Width

Ranking: Wetlands with wide stands of vegetation are more likely to stabilize sediments than those with narrow stands.

Rationale: Greater widths of vegetation provide more frictional resistance and greater dissipation of erosive forces (Gleason et al. 1979/Eem). Measurements by Knutson et al. (1981, 1982/Eem) and Wayne (1976/Eem,ab) indicated that 50 to 80 percent of the energy from small waves could be dissipated within 30 to 65 feet by partially flooded emergent vegetation. Silberhorn et al. (1974/VA:Eem) and Garbisch (1977/*) felt that tidal emergent wetlands wider than 2 and 10 feet, respectively, may be significant as an erosion deterrent, depending on local conditions. However, Knutson et al. (1981/Eem) considered such wetlands to be relatively ineffective in buffering waves if less than 6 feet wide, or if they comprised less than 20 percent of the reach of an erodible shoreline; wetlands wider than 30 feet reduced wave energy by 88 percent.

Confidence in Ranking: Moderate.

Potential Importance to Function: High.

Measure: Determination of average width of vegetated wetland zones.

Directness of Measure: Moderate. Proportionate width (width relative to channel size) rather than absolute width is often a better measure. Field measurement of stem density is preferred.

41. Velocity (spatially dominant)

Ranking: Wetlands with flowing water of high velocity will have greater opportunity for stabilizing sediments than those with no flow.

Rationale: Current velocity can be a key contributor to erosion (Figure 5).

Confidence in Ranking: Low. If equilibrium conditions exist, erosion may occur independently of velocity.

Potential Importance to Function: High.

Measure: Determination of the water velocity throughout most of the wetland during annual peak flow.

Directness of Measure: Moderate. Determination of equilibrium conditions is preferred.

45. Substrate Type (spatially dominant)

Ranking: Wetlands having no vegetation are likely to stabilize sediment only if they have predominantly rubble substrates. Rubble substrates are defined by Cowardin et al. (1979/*) as areas where stones and boulders larger than 10 inches in diameter together with bedrock cover more than 75 percent of the area.

Rationale: Finer sediments are more erodible, especially if uncompacted.

Confidence in Ranking: High.

Potential Importance to Function: Low.

Measure: Determination of the dominant substrate found in the upper 3 inches of the wetland.

Directness of Measure: Low.

3.5 Sediment/Toxicant Retention

1. Climate (predictor for effectiveness and opportunity)

Ranking: Areas along the Pacific coast (Patric et al. 1984/US:fo), those with high drainage density (miles of stream per square mile), and those with high rainfall erosivity are most likely to have greater transport of sediment and toxicants to wetlands, and thus provide wetlands with the greatest opportunity for retention. Nationwide studies of 800 watersheds indicate sediment yields per unit area are greatest in regions with about 2 inches of annual runoff (Dendy and Bolton 1976/U.S.). The **effectiveness** of tidal wetlands subject to frequent storms may be great, especially in regions of low tidal amplitude (Stumpf 1983/DE:Eem, Wolaver et al. 1988/SC:Eem).

Rationale: Runoff from areas located in regions where rainfall is particularly erosive will likely contain relatively high sediment loads (Novotny 1980). Thus, wetlands in such areas have a greater opportunity for retaining sediments and toxicants.

Confidence in Ranking: Low to moderate.

Potential Importance to Function: Moderate. Erosion and transport depend also on seasonality of rainfall, soil type, slope, land cover, and other factors.

Measure: Determination of whether the wetland is located in an intense storm region, an area with high rainfall erosivity factors (greater than 300), or small tidal amplitude (less than 2 meters). These thresholds are arbitrary.

Directness of Measure: Moderate for opportunity; low for effectiveness. Precipitation, soils, and tidal data from local sources are preferred.

5. Wetland/Watershed Ratio (predictor for opportunity)

Ranking: Large watersheds (relative to wetland size) are more likely to provide opportunities to wetlands for sediment/toxicant retention, especially if few wetlands are located in the watershed upstream of the wetland being assessed.

Rationale: Loadings of suspended sediment available to wetlands for retention are often correlated with watershed area (Costa 1977/MD:R, Dunne and Leopold 1978) and storm runoff volume (Sheridan and Hubbard 1987/GA:R). An equation for estimating flushing of sediment, given the wetland and watershed sizes, is given by Heinemann (1981).

Confidence in Ranking: Moderate to high. A nationwide study of 800 watersheds indicated sediment yield per unit watershed area was inversely correlated with the 0.16 power of the area (Dendy and Bolton 1976/U.S.).

Potential Importance to Function: Moderate. Other key interacting variables include soil type, climate, slope, and land cover.

Measure: Determination of the percentage of the watershed occupied by the assessment area as defined in Volume II, and the percentage of the watershed occupied by wetlands upslope of the assessment area. The thresholds are arbitrary.

Directness of Measure: Moderate.

7. Gradient, Velocity (predictor for effectiveness)

Ranking: Wetlands with gradual gradients are more likely to perform sediment/toxicant retention than those with steep gradients.

Rationale: Water velocity decreases, and thus retention time increases, with decreasing slope. Increasing retention time increases the potential for sediment/toxicant retention. The more gradual the wetland gradient, the greater the potential for sediment deposition and retention of toxicants by burial (De Jong 1976, Karr and Schlosser 1977/*, Fetter et al. 1978/WI:Pem, Boto and Patrick 1979/*, Mulholland 1981/NC:fo, Sheridan and Hubbard 1987/GA:R).

Confidence in Ranking: Moderate. Anoxic conditions associated with long retention times can mobilize some toxicants.

Potential Importance to Function: Moderate.

Measure: Comparison of the channel gradient with thresholds indicative of depositional conditions.

Directness of Measure: Moderate. On-site measurement using surveying equipment is preferred.

8. Inlets, Outlets (predictor for effectiveness and opportunity)

Ranking: Wetlands with surface water inlets are more likely to have an opportunity for sediment/toxicant retention than are those without such inlets. Wetlands without outlets are more likely to be effective than those with outlets.

Rationale: Surface waters carry sediments and toxicants adsorbed to them. Therefore, wetlands with inlets are more likely to have sediment and toxicant inputs, thus providing an opportunity for sediment/toxicant retention (ground water may carry dissolved but not suspended toxicants). Wetlands without outlets normally retain all sediment and physically bound toxicant inputs (with export possible only via ground water, wind, animals, or in gaseous form). Wetlands that occupy at least 1 percent of their watershed area and are capable of storing more than 10 percent of their average annual inflow usually trap more than 85 percent of the incoming sediment (Dendy and Bolton 1976/U.S.). Concentrations of lead, arsenic, cadmium, and selenium (but not mercury) are higher in sediments of wetlands without outlets than in riverine wetland sediments (Martin and Hartman 1987/ND,SD).

Confidence in Ranking: Moderate (opportunity); high (effectiveness).

Potential Importance to Function: Moderate (opportunity); high (effectiveness).

Measure: Identification of inlets and outlets for surface water flow.

Directness of Measure: Low (opportunity); high (effectiveness).

9. Constriction (predictor for effectiveness)

Ranking: Wetlands with constricted outlets are more likely to retain sediments and toxicants than those with unconstricted outlets.

Rationale: Average discharge volume is less in wetlands with constricted outlets. Therefore, the potential for retention of sediments and toxicants is greater there. Sheet flow is also more conducive to sediment detention than is channel flow (Kadlec and Tilton 1979/*, Tilton and Kadlec 1979/MI:P, Mulholland 1981/NC:fo, Brown 1985, Chescheir et al. 1987/NC:Pem, Kadlec 1987/*, Sheridan and Hubbard 1987/GA:R,).

Confidence in Ranking: Moderate. Opportunity may tend to be higher for wetlands with unconstricted outlets, as they tend to be associated with large volumes of discharge.

Potential Importance to Function: Moderate.

Measure: Identification of constricted outlets. Thresholds are arbitrary.

Directness of Measure: Moderate. Field measurement of cross-sectional area of inlets and outlets, relative to wetland volume, is preferred.

10. Wetland System (predictor for effectiveness and opportunity)

Ranking: Tidal riverine, estuarine, and marine wetlands are more likely to retain sediments and toxicants.

Rationale: Riverine systems typically carry large quantities of suspended sediments and associated toxicants, thus providing opportunity for sediment/toxicant retention. When conditions are suitable (e.g., flow velocity is decreased due to vegetation, pools/riffles, tidal influence, etc.), these suspended sediments are deposited. Clay particles are flocculated at the salt/fresh water interface (Boto and Patrick 1979/*, Correll 1978/*E), with intense flocculation occurring at salinities of 2 to 5 ppt (see Predictor 38). These salinities most commonly occur in estuarine systems.

Confidence in Ranking: Low.

Potential Importance to Function: Low to moderate. In estuaries, flocculation due to salinity is commonly overridden by the influence of density currents (Nelson 1960/*Pem).

Measure: Determination of the wetland classification according to Cowardin et al. (1979/*).

Directness of Measure: Low.

12. Vegetation Class/Subclass (Primary) (predictor for effectiveness)

Ranking: Wetlands dominated by forest, scrub-shrub, or persistent emergent vegetation are more likely to retain sediments and associated toxicants than are unvegetated, moss-lichen, or riverine aquatic bed wetlands.

Rationale: Persistent, multistemmed plants enhance sedimentation by offering frictional resistance to water flow (Richards 1934/UK:Eem; Jackson and Starrett 1959/IL:L; Karr and Schlosser 1977, 1978/*; DeLaune et al. 1978/LA:Eem; Richards 1978/NY:Eem; Boto and Patrick 1979/*; Dean 1979/*; Gleason et al. 1979/Eem; Phillips 1980/*Eab; Mulholland 1981/NC:fo; Kenworthy et al. 1982/NC:Eab; Morris and Paulson 1982/NV:Rem; Knight et al. 1984/NC:P; Brown 1985; Stabel and Geiger 1985; Lowrance et al. 1986/GA:fo; Sheridan and Hubbard 1987/GA:R).

Because vegetational resistance to water flow rapidly diminishes as the water depth becomes greater than vegetation height, wooded wetlands may be particularly effective. However, wooded sites in frequently flooded riverine situations are more likely to support single-stemmed, widely spaced vegetation (Klimas et al. 1981/*fo, Ehrenfeld 1987/NJ:P), which is less effective for sediment retention and toxicant removal.

Moss-lichen cover, because it generally intercepts little of the surface water mass, is less effective in slowing water movement and thus is not effective in supporting sedimentation, unless basin morphologies influence water exchange rates with adjoining waters.

Emergent vegetation may be more effective than aquatic bed vegetation for oxidizing toxic substances (Barko and Smart 1983/L).

The relative effectiveness of four seagrass species for retaining sediment is discussed by Fonseca and Fisher (1986/NC:Eab).

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate to high. Basin morphology is probably a better predictor of sedimentation than is vegetation frictional resistance. Some studies have found less sediment retention in wetlands than in adjacent deep water (Smith et al. 1985/LA:E) or subtidal areas (Jordan et al. 1986/MD).

Measure: Determination of the dominant vegetation class and subclass of the wetland.

Directness of Measure: Low to moderate. Sedimentation also depends on particle size, gradient, morphology of the wetland's basin, and other factors.

13. Vegetation Class/Subclass (Secondary) (predictor for effectiveness)

Ranking: See Predictor 12, Vegetation Class/Subclass (Primary).

Rationale: See Predictor 12, Vegetation Class/Subclass (Primary).

Confidence in Ranking: See Predictor 12, Vegetation Class/Subclass (Primary).

Potential Importance to Function: See Predictor 12, Vegetation Class/Subclass (Primary).

Measure: Determination of vegetation classes and subclasses comprising 10 percent, or at least 1 acre, of the wetland. Thresholds are arbitrary.

Directness of Measure: Low.

15. Vegetation/Water Interspersion (predictor for effectiveness)

Ranking: Wetlands with dense vegetation (low vegetation-water interspersion) are more likely to retain sediments and toxicants than those with sparse vegetation.

Rationale: Dense vegetation offers frictional resistance to water flow, promoting sedimentation and retention of toxicants via burial (see Predictor 12 above). Dense vegetation also reduces the resuspension of bottom sediments by wave action (Richards 1978/NY:Eem; Phillips 1980/Eab).

Confidence in Ranking: High.

Potential Importance to Process: Moderate.

Measure: Determination of relative degree of vegetation-water interspersion.

Directness of Measure: Low. Field measurement of stem and root volume is preferred but is time-consuming.

19. Fetch/Exposure (predictor for effectiveness)

Ranking: Unsheltered wetlands are less likely to retain sediments and toxicants, whereas those such as islands, deltas, bars, or peninsulas, which shelter adjacent areas by intercepting waves, are more likely to retain these.

Rationale: The greater the fetch, the greater the potential for wind mixing, and the greater the sediment carrying capacity of water (Settlemyre and Gardiner 1977/SC:E, Stern and Stickle 1978/*). Wetlands that shelter adjacent areas create conditions favorable for sedimentation (Settlemeyre and Gardiner 1977/SC:E, Stern and Stickle 1978/*) and possibly for toxicant retention. Deltas and bars, in addition to providing shelter for adjacent areas, are natural depositional areas and retain sediments directly on-site.

Winds of less than 16 mph will resuspend sediments in half-mile-wide basins that are shallower than 5 feet (Carper and Bachmann 1984/IA:Pem). Winds of only 10 mph can resuspend sediments in large 20-ft-deep basins (Demers et al. 1987/Que:E). Wave height (in inches) can be approximated as 0.1413 times the square root of the effective fetch (Hutchinson 1957/*).

Confidence in Ranking: Moderate. Sheltered areas may be subject to ice-related erosion.

Potential Importance to Function: Moderate. Sedimentation also depends on particle size, mass, temperature, and other factors.

Measure: Determination of whether the wetland is sheltered or unsheltered and exposed to wave action, or if the wetland is a part of an island, delta, bar, or peninsula that protects nearby shores.

Directness of Measure: Low to moderate. Direct measurement of wave size, wind velocity, and (ideally) sedimentation rate is preferred.

21. Land Cover of Watershed (predictor for opportunity)

Ranking: Wetlands with watersheds dominated by forest or scrub-shrub vegetation are less likely to have an opportunity for sediment/toxicant retention than are those with urban, agricultural, or similar land uses.

Rationale: Densely vegetated watersheds (e.g., undisturbed forest, scrub-shrub cover) stabilize soils, reduce runoff velocity, and thus export less sediment or suspended toxicants (Bormann et al. 1974, Likens and Bormann 1974/*R, Ostry 1982, Chang et al. 1983, Cooper et al. 1986/NC:fo). Because runoff on exposed soils (e.g., urban areas) is unimpeded by vegetation, sediments and associated toxicants are frequently flushed into wetlands, thus providing the wetlands with an opportunity to retain sediments and toxicants (Wolman and Schick 1967/MD:R, Yorke and Herb 1976/MD).

The long-term sediment yield from 812 forested watersheds nationwide was 0.25 ton/acre/year (Patric et al. 1984/US:fo). Urbanization of a 100-square mile watershed (77 percent urban) was shown by simulation models to increase sediment yields to between 4.1 and 4.4 tons/acre/year (Arnold et al. 1987).

Confidence in Ranking: High.

Potential Importance to Function: High. Proximity to source, watershed gradient, precipitation, and soil type are at least as important.

Measure: Determination of the type of land cover found on a majority of the wetland's watershed.

Directness of Measure: Moderate.

22. Flow, Gradient, Deposition (predictor for effectiveness)

Ranking: Unscoured wetlands that are part of an actively accreting delta are more likely to retain sediments and associated toxicants.

Rationale: Scoured wetlands indicate an erosional environment, and thus would not retain sediments or toxicants associated with them. Deltas, if stable over time, retain sediments and toxicants because they are depositional environments. Channel migration, which would result in scouring of wetlands and resuspension of their sediment, increases with increasing river size (Hooke 1980/SC, Nanson and Hickin 1986/BC:R) and decreasing silt and clay composition of the streambanks (Daniel 1971/IN:R, Hooke 1980/SC).

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Determination of whether scouring is present or if the wetland is part of an actively accreting delta. Definition of delta conditions is sometimes subjective.

Directness of Measure: High (if long-term series of aerial photos are used) to moderate (if other sources are used). In some cases, dendrochronology can also be used.

25. Sediment Sources (predictor for effectiveness and opportunity)

Ranking: Wetlands that receive runoff from watersheds with erosion-susceptible areas have the greatest opportunity to perform sediment/toxicant retention. For effectiveness, this predictor is used as a classification variable rather than a determinant (i.e., if overland flow is the primary sediment source, one set of criteria apply, whereas another set is used if channel flow is the primary sediment source). However, sediment loading rates may be used in a few cases to indicate wetland effectiveness; increasing levels of suspended sediment in the wetland are, up to a point, associated with increasing retention (Faye et al. 1980).

Rationale: Sediments (and toxicants associated with them) are more likely to wash into wetlands if watersheds have steep slopes, exposed soils, or soils susceptible to erosion (Paulet et al. 1972/L, Novotny 1980, Whipple et al. 1981/NJ:R, Chang et al. 1983, Cooper et al. 1986/NC:fo).

Confidence in Ranking: High (opportunity); not applicable (effectiveness).

Potential Importance to Function: High (opportunity); not applicable (effectiveness). Slope and degree of sediment control were the most significant factors affecting sediment yield from construction sites, explaining 59 percent of the variation in sediment yield (Yorke and Herb 1976/MD).

Measure: Determination of potential source(s) of inorganic sediment and toxicants such as storm-water outfalls, irrigation return waters, surface mines, exposed soils, erosion-prone soils, gullies, sand or gravel pits, or severely eroding stream or road banks. Numeric thresholds used in Volume II to define size of the potential input zone are derived from several studies.

Directness of Measure: Low. Direct measurement of suspended or deposited sediment is preferable. Numerous field techniques are available (e.g., Hudson 1982). If sediment sources are assumed, then coincidence with periods of maximum runoff should be considered, as some sources are associated only with seasonally intermittent activities (Shahane 1982). Models are available for quantifying the transport and fate of sediments (e.g., Universal Soil Loss Equation, ANSWERS model, some Hydrologic Engineering Center models).

27. Contaminant Sources (predictor for opportunity)

Ranking: Wetlands in proximity to potential sources of waterborne contaminants are more likely to have an opportunity for sediment/toxicant retention.

Rationale: By definition, a potential source of toxicants must be present in order to have an opportunity for retention.

Confidence in Ranking: High.

Potential Importance to Function (Opportunity): High. Atmospheric sources of metals and synthetic organics (via deposition or precipitation) may occasionally be significant as well (Lazrus et al. 1970, Rappaport et al. 1985/U.S.).

Measure: Documentation of potential sources of waterborne contaminants.

Directness of Measure: Low. Direct measurement of contaminant levels or (better yet) their biotic effects is preferable to assuming presence based on stereotypical sources. If sources are assumed, then coincidence with periods of maximum runoff should be considered, as some sources are associated only with seasonally intermittent activities. Models are available for quantifying the transport and fate of contaminants (e.g., Chui et al. 1982/*, Hoffman et al. 1985).

28. Direct Alteration (predictor for effectiveness)

Ranking: Wetlands that were tilled or filled or excavated or have had an outlet added or an inlet blocked are less likely to retain sediment and associated toxicants.

Rationale: Direct alterations that reduce a wetland's ability to receive or hold water (and suspended sediment and toxicants), or those that destroy vegetation which otherwise enhances sedimentation by slowing water movement, will diminish the wetland's ability to retain sediment and toxicants. Filling and excavation may also cause short-term increases in suspended sediment.

Confidence in Ranking: Low (fill, excavation) to moderate (tillage). Some newly created wetlands may have greater annual accretion rates than older ones (Pethick 1981/Eem).

Potential Importance to Function: Moderate.

Measure: Documentation of direct alterations such as tillage, excavation, filling, addition of an outlet, or blockage of an inlet.

Directness of Measure: Low. Measurements of the magnitude, location, and type of alteration will improve the accuracy.

31. Water/Vegetation Proportions (predictor for effectiveness)

Ranking: Wetlands with mostly open water are less likely to retain sediment and toxicants than those that are extensively vegetated.

Rationale: Vegetation creates frictional resistance to water movement, often creating a tortuous flow path, limiting resuspension by wind mixing, and thus promoting sedimentation.

Confidence in Ranking: Moderate. Mobilization of some contaminants may be encouraged by conditions commonly associated with vegetated wetlands (see Section 2.5).

Potential Importance to Function: Moderate. Water depth, particle size, and other factors may be at least as important.

Measure: Determination of relative water/vegetation proportions within each of the wetland zones as defined in Volume II. The thresholds are arbitrary.

Directness of Measure: Low. Stem and root volume is a better measure of sedimentation potential than is percent coverage.

34. Water Level Control (predictor for effectiveness and opportunity)

Ranking: Wetlands not influenced by an upslope impoundment are more likely to have an opportunity for sediment/toxicant retention, whereas wetlands that experience ponding created by a downstream dam or dike are more likely to be effective at retaining sediments and associated toxicants.

Rationale: Dams or dikes that create ponds allow sediments to settle out, but allow little opportunity for wetlands immediately downstream to retain sediments (Dendy 1974, Rausch and Schreiber 1977, Heinemann 1981, Whipple and Di-Louie 1981/NJ:R).

Confidence in Ranking: High (opportunity); high (effectiveness).

Potential Importance to Function: High (opportunity); high (effectiveness). Retention depends also on proximity to impoundment, type and size of dam, and other factors.

Measure: Documentation of upstream or downstream dams or dikes that impact the wetland.

Directness of Measure: Low.

35. Flooding Extent and Duration (predictor for effectiveness)

Ranking: Wetlands that experience seasonal flooding of long duration and great extent are more likely to retain sediment and toxicants.

Rationale: The greater the seasonal flooding duration and extent, the greater the settling time for suspended sediment/toxicant deposition. Also, because primary productivity generally increases with seasonal flooding (Conner and Day 1976/LA:fo, Darnell et al. 1976/*, Johnson et al. 1976/ND:fo, Burns 1978/FL:fo, Fredrickson 1979/MO:fo, Mitsch et al. 1979/IL:fo, Odum 1979/*fo, 1981; Day et al. 1980/LA:fo) vegetation in seasonally flooded wetlands will likely offer greater frictional resistance and thus promote sedimentation.

Confidence in Ranking: Moderate to high. As flooding duration and extent increase, so do flood water volume and energy, which reduces sediment trapping efficiency. Sediments from the intermittently flooded fringes of wetlands may (if unvegetated) be resuspended when flooded. Mobilization of some contaminants may be encouraged by anaerobic conditions, which commonly develop during long periods of flooding (see Section 2.5).

Potential Importance to Function: Moderate.

Measure: Determination of the extent and duration of wetland flooding. Thresholds are adapted from US Army Corps of Engineers (1980).

Directness of Measure: Low.

36. Vegetated Width (predictor for effectiveness)

Ranking: Wetlands in which the average width of emergent, scrub-shrub, or forest vegetation is great are more likely to retain sediment and associated toxicants than where vegetation is narrow.

Rationale: Persistent vegetation offers frictional resistance to water flow, thus enhancing sedimentation (see Predictor 12 above). The greater the width of vegetation within the wetland, the greater the frictional resistance offered. Silberhorn et al. (1974/VA:Eem) indicated that tidal emergent wetlands wider than 2 feet might be effective in filtering sediments.

Confidence in Ranking: High.

Potential Importance to Function: High. Width of vegetation **relative to channel width** is probably a better indicator.

Measure: Determination of the average width of emergent, scrub-shrub, or forest vegetation in the wetland.

Directness of Measure: Moderate.

41. Velocity (spatially dominant) (predictor for effectiveness)

Ranking: Wetlands with predominantly low water velocities during annual peak flows are more likely to retain sediments and toxicants than are those with rapid flow.

Rationale: The greater the flow velocity, the less likely that sediments and toxicants will be retained by sedimentation and burial (Kadlec and Tilton 1979/*, Brown 1985).

Confidence in Ranking: Moderate. Resuspension may be no greater in rapid channels than in slow ones if sediment equilibria exist.

Potential Importance to Function: High. Flow velocity is the single most important factor affecting sediment-trapping efficiency (Dendy 1974, Karr and Schlosser 1977/*, Turner 1980/*L). Sediment particle size, settling rate, and cohesiveness are also important.

Measure: Determination of the water velocity through most of the wetland during peak flow.

Directness of Measure: Moderate to high.

42. Velocity (Secondary) (predictor for effectiveness)

Ranking: See Predictor 41, Velocity (spatially dominant).

Rationale: See Predictor 41, Velocity (spatially dominant).

Confidence in Ranking: Low.

Potential Importance to Function: Low.

Measure: Determination of the water velocity occurring in at least 1 acre, or 10 percent of the wetland. Thresholds are arbitrary.

Directness of Measure: Low.

43. Water Depth (spatially dominant) (predictor for effectiveness)

Ranking: Shallow wetlands are more likely to retain sediments and toxicants than are deep wetlands.

Rationale: Shallow wetlands offer greater frictional resistance, both directly and as a result of their favoring rooted vegetation. The resultant velocity reduction favors sedimentation (Kadlec and Tilton 1979/*, Knight et al. 1984/NC:P).

Confidence in Ranking: Low. Wind mixing of the substrate is greater in shallow wetlands (especially if the wetland is unvegetated), thus resuspending sediments and inhibiting burial (Nolen et al. 1985/OK:L). As a result, sediment retention over the long term may be greater in large, deep wetlands due to their longer retention times and greater storage capacity (Evans and Rigler 1983, Stevenson et al. 1988).

Potential Importance to Function: Moderate. Wetland size and gradient have more effect on retention time (and thus on sedimentation) than does depth (Kadlec 1987/*).

Measure: Determination of the spatially dominant water depth of the wetland. Mean depth in some riverine wetlands can be rapidly estimated from measuring curvature of meanders on aerial photos (see Williams 1986).

Directness of Measure: Moderate.

45. Substrate Type (spatially dominant) (predictor for effectiveness)

Ranking: Wetlands with predominantly bedrock, rubble, or cobble-gravel subtrates are less likely to retain sediment and toxicants than those with mud or organic sediments.

Rationale: Mud and organic matter commonly associated with fine sediments (Ricci et al. 1983) are usually found in sheltered, depositional areas, whereas cobble and rubble are typical of high-energy, erosional areas. Thus, wetlands with muddy or organic substrates are likely to have a higher potential for, and be indicators of, sediment trapping (Silberhorn et al. 1974/VA:Eem). Toxicant retention (especially of metals and nonionic synthetic organics, e.g., PCBs) is often associated with organic soils (see Section 2.5 for references).

Confidence in Ranking: Moderate. Resuspension may be no greater among coarse-substrate wetlands than among fine-substrate (organic) ones if sediment equilibria exist. Organic sediments in some situations can encourage mobilization of contaminants (see Section 2.5). Adsorption of contaminants onto the finest particles may make them less likely to be retained in wetlands, because settling rates are slowest for the finest particles and they may be easily flushed downstream.

Potential Importance to Function: High.

Measure: Determination of the spatially dominant substrate type of the wetland.

Directness of Measure: Low.

48. Salinity and Conductivity (predictor for effectiveness)

Ranking: Wetlands with mixosaline (0.5 to 18.0 ppt) waters are more likely to retain sediment and thus its associated toxicants.

Rationale: Clay particles flocculate and settle out at the salt/fresh water interface (i.e., mixosaline waters) (Correll 1978/*E, Boto and Patrick 1979/*). Intense flocculation occurs as soon as the salinity reaches 2 ppt and is completed at a salinity of 5 ppt (Rochford 1953/AU:E).

Confidence in Ranking: Moderate. The process can sometimes be overriden by the influence of density currents (Nelson 1960/VA:R).

Potential Importance to Function: Moderate.

Measure: Determination of salinity or conductivity of the wetland.

Directness of Measure: Moderate.

49. Aquatic Habitat Features (predictor for effectiveness)

Ranking: Riverine wetlands with pools and riffles are more likely to retain sediment and toxicants. Those with straight channels are less likely to perform this function.

Rationale: Pools and riffles reduce erosive energy and sediment transporting ability, especially during low and medium flows (Stall and Yang 1972, Karr and Schlosser 1977/*, Nunnally and Keller 1979/*R). Stall and Yang (1972/IL) found that erosive energy and transporting capacity was 23 to 26 percent lower in a pool and riffle stream during medium- and low-flow conditions than in an otherwise equivalent uniform channel.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Determination of the proportion and spacing of the wetland's stream reach containing riffles, pools, backwaters, or other slow-water areas. Pool-riffle ratio thresholds are based on criteria for long-term stream stability given by Nunnally and Keller (1979/*R).

Directness of Measure: Moderate.

55. Suspended Solids (predictor for opportunity)

Ranking: Wetlands that receive runoff or other surface water with high levels of suspended solids have greater opportunity to retain sediments and toxicants than those that receive runoff or other surface water with low levels of suspended solids.

Rationale: By definition, sediment must be present as suspended solids for a wetland to have an opportunity to retain them. The greater the concentration of solids, the greater the opportunity to trap the suspended particles.

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of suspended solids or (less desirably) the Secchi disc reading of water entering the wetland.

Directness of Measure: Moderate. Where suspended solids exist as colloids, it may be impossible for them to settle.

64. Total Suspended Solids (TSS) Differential (predictor for effectiveness)

Ranking: Wetlands that show higher levels of suspended solids at inlet(s) than at outlet(s), or those with retention times sufficiently long to allow sediment deposition, are more likely to retain sediment and toxicants.

Rationale: If the TSS concentration is consistently lower at outlet(s) than at inlet(s), then the wetland may be acting as a sink for suspended solids.

Confidence in Ranking: Moderate. If the wetland has another water source (e.g., the wetland is a ground water discharge area), the level of TSS may be lowered by dilution in the absence of sediment-trapping activity. Conversely, the TSS level may increase due to additional inputs from eroding stream banks, concentration by evapotranspiration, or export of suspended organic substances, even if the wetland is actively trapping sediment.

Potential Importance to Function: High.

Measure: Determination of levels of TSS simultaneously at wetland inlet(s) and outlet(s) or measurement of retention time in the wetland with tracer dyes or morphometric analyses. Discharge at inlet and outlet should be measured concurrently with TSS, and both measurements should be made continuously over a 1- to 2-year period.

Directness of Measure: Moderate.

3.6 Nutrient Removal/Transformation

1. Climate (predictor for opportunity)

Ranking: Areas having highly erosive rainfall are more likely to encourage transport of nutrients to wetlands, and thus their wetlands have an opportunity for nutrient removal/transformation.

Rationale: The rainfall erosivity factor is an index to the erosiveness of rainfall and associated runoff. Nutrients, in addition to soil particles, are removed and transported in the erosion process. Thus, this index reflects the potential nutrient load of runoff.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate. Erosion and transport depend also on seasonality of rainfall, soil type, gradient, and land cover.

Measure: Determination of the rainfall erosivity factor for the area of interest.

Directness of Measure: Low (if from maps) to moderate (local data).

4. Location and Size (predictor for opportunity)

Ranking: Larger watersheds are more likely to encourage transport of nutrients to wetlands, and thus their wetlands have an opportunity for nutrient removal/transformation.

Rationale: The larger the watershed, the greater the source area and erosion potential for nutrients in runoff. The relationship of nutrient delivery to watershed area in agricultural watersheds is nonlinear, with nutrient delivery per unit area decreasing in larger watersheds, as can be predicted from the equations of Prairie and Kalff (1986/QUE:R).

Confidence in Ranking: High.

Potential Importance to Function: Moderate. Land cover type, proximity to wetland, slope, soil type, and other factors are also important.

Measure: Determination of the wetland's watershed size.

Directness of Measure: Moderate (high if comparing wetland watersheds with similar characteristics).

5. Wetland/Watershed Ratio (predictor for opportunity)

Ranking: Large watersheds (relative to wetland size) have a greater opportunity for nutrient removal/transformation, especially if few other wetlands are located upslope from the assessed wetland.

Rationale: Concentrations of nutrients available to wetlands are often correlated with watershed area, although in the case of agricultural watersheds the relationship is not necessarily linear (Prairie and Kalff 1986/QUE).

Confidence in Ranking: High. Soil type, climate, slope, and land cover are other important interacting variables.

Potential Importance to Function: Moderate.

Measure: Determination of the percentage of the watershed acreage occupied by the wetland. The 20-percent threshold is arbitrary.

Directness of Measure: Moderate (high if comparing wetland watersheds with similar characteristics).

7. Gradient (predictor for effectiveness)

Ranking: Wetlands with gradual gradients are more likely to retain or transform nutrients than those with steep gradients.

Rationale: Water velocity decreases, and retention time increases, with decreasing slope. Because the potential for nutrient retention increases with retention time (De Jong 1976), the more gradual the wetland gradient, the greater the potential for retention of sediment-adsorbed nutrients by burial (De Jong 1976, Karr and Schlosser 1977/*).

Confidence in Ranking: Moderate to high. Uncertainty exists because (1) anoxia associated with longer retention times (Weiler 1978/WI:L, Dahm et al. 1987) can mobilize sediment-bound phosphorus and (2) low-gradient (<3 percent) riverine wetlands may be vulnerable to rapid export of particulate phosphorus (Prairie and Kalff 1988b/QUE:R).

Potential Importance to Function: Moderate.

Measure: Determination of the wetland channel gradient relative to threshold gradients necessary to create depositional velocity conditions.

Directness of Measure: Moderate. On-site measurement using surveying equipment is preferred.

8. Inlets, Outlets (predictor for effectiveness and opportunity)

Ranking: Wetlands with permanent inlets have a greater opportunity for nutrient removal/transformation than do those without inlets, whereas those with no outlets are more effective at nutrient removal/transformation.

Rationale: Surface waters carry suspended or dissolved nutrients. Thus, wetlands with inlets are more likely to have nutrient inputs. Wetlands without outlets *would retain all nutrient inputs except those lost via ground* water, wind, freeze-out (Kadlec 1986/*), animals, or gaseous transformation (Novitzki 1978/WI:P, Carter et al. 1979/*).

Confidence in Ranking: Moderate (opportunity); high (effectiveness).

Potential Importance to Function: Moderate (opportunity); high (effectiveness).

Measure: Determination of inlets and outlets for surface water flow.

Directness of Measure: Low (opportunity); high (effectiveness).

9. Constriction (predictor for effectiveness)

Ranking: Wetlands with constricted outlets are more likely to retain and/or transform nutrients than are those with unconstricted outlets.

Rationale: Average discharge volume is likely less in wetlands with constricted outlets. Therefore, the potential for biological processing, sedimentation, and retention of nutrients is greater (e.g., Brown 1985). In any season, beaver-impounded wetlands, for example, can retain 3 to 10 times more particulate organic matter than do downstream channels (McDowell and Naiman 1986/QUE:R). Impounded tidal wetlands exported less ammonium and phosphorus than natural salt marshes (McKellar et al. 1987/SC:E,P). Wetlands with sheet flow (Question 9.2, Volume II) may experience greater nutrient removal due to greater contact between plants and runoff (see Predictor 12 below). Similarly, riverine pools that comprise only 23 percent of a stream may contribute 36 percent of the shredder organisms, which are partly responsible for nutrient transformation (Huryn and Wallace 1987/NC:R).

Confidence in Ranking: Moderate. Nutrients can be mobilized, not retained, in the anaerobic zones most commonly associated with constricted wetlands (Dahm et al. 1987). Unconstricted (e.g., fringe) wetlands may have temporarily high nutrient retention because their greater water exchange rates may encourage plant production and nutrient uptake (Hynes 1970/*R; Conner and Day 1976/LA:fo; Gosselink and Turner 1978/*; Odum 1978/*FL, 1979/*fo, 1981; Brown et al. 1979/*fo; Day et al. 1980/LA:fo; Mendelssohn and Seneca 1980/NC:Eem; Phillips 1980/*Eab; Brown 1981/FL:Pfo; Horner and Welch 1981/WA:Rab).

Potential Importance to Function: Moderate.

Measure: Determination of whether a constriction is present or not. Thresholds are arbitrary.

Directness of Measure: Moderate. Field measurement of cross-sectional area of inlets and outlets, relative to wetland volume, is preferred.

12. Vegetation Class/Subclass (Primary) (predictor for effectiveness)

Ranking: Wetlands with predominantly forested, scrub-shrub, floating vascular aquatic bed, or persistent emergent vegetative cover are more likely to remove or transform nutrients, whereas those with predominantly moss-lichen cover are less likely to perform this function.

Rationale: Forested and scrub-shrub vegetation may retain nutrients on a long-term basis in woody tissues. Wooded riparian wetlands have been shown to be effective for nitrogen removal, primarily through denitrification (Kitchens et al. 1975/SC:fo; Karr and Schlosser 1978/IL:R; Kuenzler et al. 1980/NC:fo;Schlosser and Karr 1981a/IL:R,b/IL:fo; Todd et al. 1983/GA:fo; Yates and Sheridan 1983/GA:P; Lowrance et al. 1984/GA:fo; Nessel and Bayley 1984/FL; Peterjohn and Correll 1984/MD:fo; Cooper et al. 1986/NC:fo; P-Schnabel 1986; Ehrenfeld/NJ:P 1987).

Woody vegetation may also store nutrients seasonally because uptake is greatest in spring and summer (Kitchens et al. 1975/SC:fo, Boyt et al. 1976/FL:P, Hickok et al. 1977/MN:P, Mitsch et al. 1977/IL:fo, Dierberg and Brezonik 1978/FL:fo, Fritz and Helle 1978/FL:Pfo, Tilton and Kadlec 1979/MI:P, Lowrance et al. 1984/GA:fo). However, uptake is usually less than for emergent vegetation and upland vegetation (Ehrenfeld 1987/NJ:P).

Persistent woody vegetation in deeply flooded riverine wetlands offers frictional resistance to water flow and retains nutrients via burial (see Predictor 12, Section 3.5, for references). However, woody riverine wetlands that are frequently flooded usually support single-stemmed, widely spaced plants (Klimas et al. 1981/*fo, Ehrenfeld 1987/NJ:P), which are less effective for sediment *retention and nutrient removal.*

Persistent emergent vegetation has higher value than nonpersistent emergent vegetation because it may have greater effects on subsurface retention and may accumulate some nutrients on an annual basis in persistent tissues. Although nonpersistent species may have a major effect during the growing season (Whigham et al. 1986/MD:P), persistent vegetation probably binds the sediment better, favoring nutrient retention by burial (Grant and Patrick 1970/PA:R, Reimold 1972, Banoub 1975/EU:Lem, Broome et al. 1975/NC:Eem, Klopatek 1975/WI:Pem, Lee et al. 1975/*, Patrick and DeLaune 1976/LA:Eem, Spangler et al. 1976/WI:R,P, Hickok et al. 1977/MN:P, Stevenson et al. 1977/NJ:E, Dolan et al.1978/FL:Pem, Kibby 1978, Loucks and Watson 1978/WI:L, Prentki et al. 1978/WI:L, Simpson et al. 1978/*, Valiela et al. 1978/MA:Eem, Tilton and Kadlec 1979/MI:P, MacCrimmon 1980/ONT:P, Odum and Smith 1981/*Ptem, Nichols 1983/*).

For nitrogen (ammonia) removal, bulrushes are better than common reed, which is better than cattail (Seidel 1976/*, Gersberg et al. 1986/CA:Pem); however, cattail may be effective for seasonal phosphorus retention (Brown 1985). Species with deep roots (30 cm) appear to be best, because they enhance denitrification by extensively oxidizing the sediments (Gersberg et al. 1986/CA:Pem, Dunbabin et al. 1988). Denitrification rates are greatest to a depth of about 2 feet, with a decline in the rate below 3 inches (Brodrick et al. 1988).

Nonrooted aquatic bed vegetation derives its nutrients from the water column (Camfield 1977/em; Klopatek 1978/*Rem; Whigham et al. 1980/NJ:em, 1986), as does epiphytic vegetation (van der Valk et al. 1979/*). Therefore, nonrooted aquatic plants are important in short-term removal and/or transformation of nutrients, both from the water column and secondarily from the sediment (Chen and Barko 1988/*ab). Submersed aquatic vegetation (e.g., *Potamogeton*) can support nutrient retention by flocculation, as a result of pH increases associated with photosynthetic activity (Wetzel 1983/*). However, submerged and floating-leaved aquatic bed vegetation can reduce physical aeration (as quantified by Thyssen et al. 1983/*Rab); the resultant anoxic conditions in underlying waters and sediments may favor release of phosphorus (Morris and Barker 1977, Dahm et al. 1987). Submersed (e.g., eelgrass) and floating-leaved (e.g., duckweed) aquatic bed vegetation generally is ineffective for denitrification (the principal nitrogen removal mechanism in most wetlands) because their roots do not penetrate and oxidize the underlying sediment (Iizumi et al. 1980).

Rooted aquatic bed vegetation (e.g., waterhyacinth) derives a major portion of its nutrient requirements from the sediments (MacCrimmon 1980/ONT:P, Reddy and Sutton 1984/FL:em). Nutrients are released to the water column when these plants die, and may be leached from living plants (Wolverton et al. 1976/MI:P, Mudroch and Capobianco 1979, Tilton and Kadlec 1979/MI:P, Barko and Smart 1980, Diaz et al. 1982/*E), thus offsetting retention associated with other processes (Barko and Smart 1983/L). Although rooted vascular plants may help underlying sediments retain phosphorus by raising the redox potential, this effect may be minor (Chen and Barko 1988/*ab).

Sphagnum moss generally grows in saturated, acidic conditions, which inhibit decomposition. Under such conditions nutrients, especially phosphorus, can be stored at high concentrations (Hooper and Morris 1982/MI:P). However, because vegetational resistance to water flow rapidly diminishes when vegetation is submersed, moss-lichen cover is ineffective at reducing water velocity. Therefore, such vegetation usually does not cause sedimentation and biological uptake of nutrients.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate. Loading rates, sediment type, and detention time are also key factors.

Measure: Determination of dominant vegetation class or subclass occurring in the wetland.

Directness of Measure: Low to moderate. Direct measure of sedimentation, root morphology and density, redox potential, and other factors is preferred.

17. Vegetation Form Richness (predictor for effectiveness)

Ranking: Wetlands with high vegetation form richness are more likely to perform nutrient removal/transformation.

Rationale: As discussed for Predictor 12 (Vegetation Class/Subclass), different vegetation forms are involved in the nutrient cycle in different ways. For example, epiphytes (growing on submersed and emergent plants) and free-floating plants take nutrients directly from the water, whereas emergent species obtain nutrients primarily from the sediments. Rooted aquatic bed vegetation derives nutrients from both sources (see Predictor 12 for references). Thus, the greater the plant form richness, the more likely that at least one group of plants will be able to utilize, and thus retain at least temporarily, available nutrients (Whigham et al. 1986/MD:P).

Confidence in Ranking: Moderate.

Potential Importance to Function: Low. A homogeneous stand of one highly effective species may be more effective for removal and transformation of nutrients.

Measure: Determination of the wetland's plant form richness.

Directness of Measure: Low.

21. Land Cover of the Watershed (predictor for opportunity)

Ranking: Wetlands having watersheds with predominantly forest and scrub-shrub cover are less likely to receive nutrients from upslope drainage, and therefore have less opportunity for nutrient removal/transformation than those with watersheds dominated by impervious surfaces, agricultural fields, or exposed soils.

Rationale: Forested watersheds usually have low nitrogen and phosphorus loadings (Likens and Bormann 1974/*R, Yorke and Herb 1976/MD, Bormann and Likens 1979/NH:R, Duda 1982/NC, Chang et al. 1983, Lowrance et al. 1984/GA:fo, Peterjohn and Correll 1984/MD:fo). Fertilization associated with urban and agricultural runoff increases the nitrogen and phosphorus loading, as does the presence of livestock (Jones et al. 1976/IA:Eem, Kuenzler and Craig 1986/NC,VA).

Sediment loss from watersheds with exposed soils is great (Wolman and Schick 1967/MD:R, Yorke and Herb 1976/MD), and because nutrients can be associated with sediment particles, nutrient loadings to wetlands can be high as well when extensive watershed or bank erosion is occurring (Elder 1985/FL:fo). Urbanization dramatically increases phosphorus loading, but has less of an effect on nitrogen loading (Dillon and Rigler 1975, Loucks and Watson 1978/WI:L). In North America, phosphorus loading from domestic sources is approximately 0.80 kg/person/year (Dillon and Rigler 1975). The following mean values (kg/ha/year) are based on studies reviewed by Farnworth et al. (1979):

	Exported	
Watershed Land Cover	**Total N**	**Total P**
Forest	3.10	0.10
Forest and pasture	4.70	0.28
Agricultural	10.60	0.18
Urban	6.70	4.80

Confidence in Ranking: Moderate to high.

Potential Importance to Function: Moderate to high. Proximity to source and (in the case of phosphorus) watershed slope, shading, soil type, and geology may be at least as important (Anderson 1974; Fannin et al. 1985/WY; Prairie and Kalff 1988a,b/QUE:R). However, proportionate coverage may be a better indicator than proximity, at least in small watersheds (10 to 20 square miles) (Omernik et al. 1981). Also, atmospheric deposition in some regions may be a more important source of nutrients than direct runoff (Correll and Ford 1982/MD:E).

Measure: Determination of the watershed's dominant land cover type.

Directness of Measure: Moderate to high.

23. Ditches/Canals/Channelization/Levees (predictor for effectiveness)

Ranking: Wetlands with channelized distributaries or other modifications that cause surface water to leave at a faster rate than normally would occur are less likely to remove and/or transform nutrients than those without such modifications.

Rationale: Accelerated transport of waters away from the wetland offers less potential for sediment deposition or nutrient assimilation before flow leaves the wetland. Watersheds with large drainage densities (length of ditches per unit area) have shorter detention times (e.g., 2.2 days versus 4.5 days for undrained areas, Bedient et al. 1976/FL). Watersheds drained by agricultural ditches, and those that have less than 7 percent remaining wetlands, are appreciably less effective for removing nutrients than are those with more extensive wetlands (Bedient et al. 1976/FL, Chescheir et al. 1987/NC:Pem). See also Predictor 41.

Ditching may also accelerate nutrient release by increasing soil erosion (Craig et al. 1980/LA:Eem), causing soil subsidence and oxidation, encouraging land uses (e.g., cropland) that cause large nutrient loadings, and shortcutting the natural flow paths and thus reducing processing times (Kadlec and Tilton 1979/*, Tilton and Kadlec 1979/MI:P).

Confidence in Ranking: Moderate to high.

Potential Importance to Function: Moderate.

Measure: Determination of the presence of ditches, canals, levees, or similar artificial features that cause surface water to leave at a faster rate than if such features were not present.

Directness of Measure: Low to moderate.

24. Soils (predictor for effectiveness)

Ranking: Wetlands with predominantly fine mineral sediments (e.g., alfisols, clays) or those with sediments containing high levels of aluminum or iron are more likely to remove or transform nutrients, especially phosphorus.

Rationale: Nutrients can enter sediments (and thus be removed at least temporarily) either directly or by complexing with dissolved or suspended particulate matter and precipitating (Darnell et al. 1976/*, Windom 1977). Ferric iron, manganese, zinc, copper, and aluminum can complex with and remove phosphorus (Windom 1977). This complexing process depends on the pH, the concentration and types of these cations, and the concentration of phosphates, organic compounds, and sulfates (Farnworth et al. 1979/*). The phosphorus adsorption capacity of wetlands, and hence probably their long-term nutrient assimilative capacity, can best be predicted by the oxalate-extractable aluminum content of their sediments (Richardson 1985/P,R). Mineral soils, because of their usually higher concentrations of aluminum, typically have much higher capacities to retain phosphorus (Richardson 1985/P,R). Organic and clay particles may also remove nitrogen and/or phosphorus from the water column (Klopatek 1978/*Rem). but additional retention occurs only until adsorption sites are saturated.

Confidence in Ranking: Moderate. Phosphorus removal via burial may occur only minimally if incoming suspended sediments are mainly clay, as such finer particles settle slowly (if at all).

Potential Importance to Function: Moderate.

Measure: Determination of whether the wetland has primarily fine mineral soils or if the wetland's sediments have elevated concentrations of aluminum or iron. The 4,000 mg/kg threshold is inferred from Richardson (1985/P,R).

Directness of Measure: Low. Direct measurement of oxalate-extractable (amorphous) aluminum is strongly preferred.

26. Nutrient Sources (predictor for effectiveness and opportunity)

Ranking: Wetlands that receive major drainage from nutrient-rich sources such as sewage outfalls, phosphate mines, feedlots, pastureland, landfills, eroding streambanks, fertilized soils, or soils that have been tilled, burned, or recently cleared have a greater opportunity for nutrient removal/transformation than those without such sources. For effectiveness, a variation of this predictor is used as a classification variable rather than a determinant (i.e., if sheet flow is the primary nutrient source, one set of criteria apply, whereas another set is used if channel flow is the primary nutrient source).

Rationale: By definition, presence of nutrient-rich sources of surface water provides opportunity for nutrient removal and/or transformation.

Confidence in Ranking: Moderate to high (opportunity); not applicable (effectiveness).

Potential Importance to Process: High (opportunity); not applicable (effectiveness). See Predictor 21.

Measure: Documentation of potential nutrient sources such as sewage outfalls, phosphate mines, tile drains, canals, feedlots, landfills, septic fields, fertilized soils, tilled soils, etc. The threshold given for numbers of houses or people within the impact zone is from Nichols (1983/*).

Directness of Measure: Low. Direct measurement of nutrient levels or (better yet) their biotic effects is preferable to assuming presence based on stereotypical sources. If sources are assumed, then coincidence with periods of maximum runoff should be considered, since some sources are associated only with seasonally intermittent activities (Shahane 1982). Models for quantifying the transport of nutrients to water bodies are widely available (see review by Westerdahl et al. 1981/*L).

28. Direct Alteration (predictor for effectiveness)

Ranking: Wetlands that have been tilled, filled, or excavated, or those that have had an outlet added or an inlet blocked, are less likely to remove and/or transform nutrients.

Rationale: Direct alterations that reduce the wetland's ability to receive or hold water (and as a result the sediments and nutrients carried by it), or those that alter wetland hydroperiods, disturb sediments, or destroy vegetation that otherwise enhances nutrient retention, will often negatively impact the ability of a flowing-water wetland to retain and/or transform nutrients.

Confidence in Ranking: Moderate. Nutrient retention in newly created wetlands may initially be greater than in old ones due to higher mineral content of the sediment and possibly greater sedimentation rates.

Potential Importance to Function: High.

Measure: Documentation of direct alterations such as tillage, excavation, filling, addition of an outlet, or blockage of an inlet.

Directness of Measure: Low. The effects of location, size, and type of alteration must also be considered.

33. Most Permanent Hydroperiod (predictor for effectiveness)

Ranking: Wetlands with nontidal areas that are permanently flooded or saturated, or tidal areas that are irregularly exposed or irregularly flooded, are more likely to perform nutrient removal/transformation.

Rationale: Wetlands with constantly (or nearly constantly) saturated substrates tend to retain nutrients, partly because the rate of oxygen diffusion into constantly saturated soils is slow (Bella et al. 1972/OR:E, Reddy and Patrick

1975/FL). These conditions usually favor phosphorus retention (e.g., Wolaver and Spurrier 1988b/SC:Eem) because (1) constantly saturated soils are typically anoxic except for a thin layer at the water-sediment or water-air interface, (2) the rate of decomposition (nutrient release) is lower under anaerobic than under aerobic conditions (Cook and Powers 1958/NY:Pem; Hickok et al. 1977/MN:P; Gallagher 1978/*; Gosselink and Turner 1978/*; Klopatek 1978/*Rem; Kadlec and Kadlec 1979/*; Mathias and Barica 1980; Phillips 1980/*Eab; McKee and Seneca 1982/NC:Eem; Diaz et al. 1982*E; Hsieh and Weber 1984), and (3) sediment retention, and thus phosphorus retention, can be greater in irregularly exposed or subtidal areas (Stevenson et al. 1988).

Under these conditions, ammonium may accumulate and be released to overlying waters, resulting in net loss of nitrogen from the wetland (Klopatek 1978/*Rem). However, some oxidation of the sediments, either by turbulent mixing, drawdown, or plant roots, is essential for optimal nitrogen removal (Bowmer 1987/AU:P).

The close association of anaerobic and aerobic conditions at the surface of saturated sediments, as well as rapid fluctuations between anaerobic and aerobic conditions (e.g., tidal and floodplain wetlands), also favors nitrogen removal (Engler and Patrick 1974/LA; Reddy and Patrick 1975/FL,1976/FL; Tilton and Schwegler 1978/*; Heliotis and DeWitt 1983/MI:P; Bowden 1984/MA:tem). Sustained drawdowns do not always enhance nitrogen release (Brinson et al. 1983/NC:fo, 1984; Verry 1986/*). Nutrient retention is often greatest at the horizontal edge that separates surface waters from saturated soils that are not inundated (Karr and Schlosser 1977/*, DeLaune et al. 1978/LA:Eem, Livingston 1979/*).

Fluctuations and drawdowns usually do not benefit phosphorous retention. During periods of periodic flooding (drying and rewetting), phosphorus adsorption by wetlands is lessened (Nichols 1983/*) and enhanced decomposition and oxidation lead to phosphorus release, particularly in nonclay soils (Cook and Powers 1958/NY:Pem; Correll 1978/*E; Crow and MacDonald 1978/*; Klopatek 1978; van der Valk et al. 1979/*; Beauchamp and Kerekes 1980/NB:P; Day et al. 1980/LA:fo, 1981/*fo; Brinson et al. 1981a/*fo; Yarbro 1983/NC:fo; Dahm et al. 1987; Schoenberg and Oliver 1988/GA:Pab). Seasonal inundation of sediments frequently (but not inevitably, see Kadlec 1986/*) mobilizes sediment nutrients and makes them available for export.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate to high. Vegetation type, which is largely determined by hydroperiod, also may have a major role in nutrient cycling, particularly in eutrophic wetlands. Redox potential, and thus nutrient mobilization potential, may increase with marsh surface elevation (Bertness and Ellison 1987/RI:Eem).

Measure: Determination of the wetland's most permanent hydroperiod.

Directness of Measure: Low to moderate.

36. Vegetated Width (predictor for effectiveness)

Ranking: Wetlands with wide stands of vegetation (primarily emergent or woody) are more likely to remove/transform nutrients.

Rationale: Extensive stands of vegetation offer frictional resistance to water flow, thus enhancing nutrient removal by sedimentation and burial (see Predictor 12). Nutrient retention is usually greatest in shallow depths (especially 1.2 to 3.0 m) where wetland vegetation typically prevails. Wetlands with length-to-width ratios greater than 3.0 (parallel to flow) are more likely to remove nutrients (Knight 1987). Forested riparian areas commonly intercept and retain subsurface, nonpoint nitrogen within distances of 40 to 50 feet (Cooper et al. 1986). Average distances required in natural wetlands for 50-percent removal of domestic wastewater nutrients range from 200 to 2,000 feet, with an average of about 600 feet (Kadlec 1987/*). Distances of 3,700 to 7,600 feet were associated with residence times of 15 to 75 hours in one wetland having a width of 4,900 feet and slope of 0.015 percent (Chescheir et al. 1987/NC:Pem).

Confidence in Ranking: High.

Potential Importance to Function: Moderate. The vegetation type, soil type, gradient, and diffusion pattern are also important and interact with wetland width. In watersheds with sandy soils, wetland vegetation must be located very close to the nutrient source (or have long roots) in order to play a major role in nitrogen removal (Ehrenfeld 1987/NJ:P).

Measure: Determination of the average width of emergent, scrub-shrub, or forest vegetation in the wetland.

Directness of Measure: Low. The proportionate width (i.e., vegetation width relative to open-channel width) may be a better indicator.

41. Velocity (spatially dominant) (predictor for effectiveness)

Ranking: Wetlands with low flow velocities are more likely to remove and/or transform nutrients than are those with high flow velocities.

Rationale: The slower the flow velocity, the more likely nutrients will be retained by sedimentation and burial (for references, see Predictor 12). Moreover, longer residence times associated with slower currents favor nutrient-removing processes such as denitrification (Mulholland 1981/NC:fo). Retention times of at least 5 days are preferred for pretreated and nonpoint wastes, although retention of untreated wastes for 20 to 40 days is commonly suggested (e.g., Bedient et al. 1976/FL).

Confidence in Ranking: Moderate. Nutrient uptake and storage by some plants may be greater in flowing water than in standing water (Prescott

1968/*ab, Brown et al. 1979/*fo), but this seems generally untrue (Brown and Peterson 1983/IL:fo).

Potential Importance to Function: High.

Measure: Determination of the water velocity throughout most of the wetland during annual peak flow.

Directness of Measure: Moderate. Detention time is a better measure.

56. Dissolved Solids or Alkalinity (predictor for effectiveness)

Ranking: Wetlands having waters with low alkalinity levels (less than 20 mg/l calcium carbonate ($CaCO_3$)) are less likely to remove or transform nutrients.

Rationale: Phosphates can be precipitated with calcium, which is associated with alkaline conditions (Lee et al. 1975/*, Fetter et al. 1978/WI:Pem, Nichols 1983/*). This ranking does not apply to nitrogen.

Confidence in Ranking: Moderate. Alkaline waters may also indicate near-saturation conditions for phosphorus.

Potential Importance to Function: Moderate.

Measure: Determination of alkalinity ($CaCO_3$) levels in the wetland's waters.

Directness of Measure: Moderate.

3.7 Production Export

1. Climate

Ranking: Wetlands located in intense storm regions or those with erosive rainfall are more likely to export production.

Rationale: Heavy rainfall, wind, and tides associated with intense storms can move large quantities of organic matter from wetlands into downstream water bodies. Storms can produce tidal surges and large waves in coastal wetlands, expand flooding, and thus increase the export of production (Nixon and Oviatt 1973/RI:Eem; Pickral and Odum 1977/VA:E; Livingston and Loucks 1979/*; Odum et al. 1979/*Eem; Odum 1980/*Eem; Hackney and Bishop 1981/MS:Eem; Casey and Farr 1982/UK:R; Elder and Mattraw 1982FL:fo; Verhoff et al. 1982;

Borey et al. 1983/TX:Eem; Simenstad 1983/*E; Tate and Meyer 1983/GA:R; Livingston 1984/*FL:E; Mattraw and Elder 1984/FL:fo; McPherson and Sontag 1984/FL:E; Thayer et al. 1984/*Eab; Wolaver et al. 1984/SC:E, 1988; Nixon 1988/*L,E,M).

Confidence in Ranking: Low. Storms may also cause prolonged reduction in production due to scouring of plants, plant propagules, and animals.

Potential Importance to Function: Moderate.

Measure: Determination of whether the wetland is located.in an intense storm region, or if it has a rainfall erosivity factor greater than 300. The thresholds used for this predictor are arbitrary.

Directness of Measure: Low.

2. Acreage

Ranking: Larger wetlands are more likely to export production.

Rationale: Total production is higher in large wetlands (although not necessarily on a per unit area basis). Thus, more production is available for export. Productivity of lacustrine wetlands can be estimated from data on lake size, mean depth, and nitrogen concentration, using the empirical equations of Smith and Wallsten (1986).

Confidence in Ranking: Low. Some empirical analyses indicate that larger lakes have less wetland production per unit area, at least in the case of submersed aquatic plants (Duarte et al. 1986).

Potential Importance to Function: Moderate.

Measure: Determination of the surface area of the wetland and any wetlands within 1 mile that are connected by surface water. The 5-acre threshold is arbitrary.

Directness of Measure: Low.

4. Wetland Location, Size of Watershed

Ranking: Wetlands with large watersheds are more likely to export production.

Rationale: Wetlands with proportionately large watersheds (at least up to about stream order 5) tend to be more productive in some regions (Moulton 1970/MA, Naiman and Sedell 1979, Welcomme 1979/*). Also, the larger a wetland's watershed, the greater the runoff (Fetter 1980/*), the more likely the wetland is to be flushed by flood waters, and thus the greater the likelihood of substantial production being exported.

Confidence in Ranking: Low. Small watersheds, because they are more numerous, may be just as important on a cumulative basis. The dynamic water levels that often characterize isolated wetlands in headwater areas may result in increased nutrient mobilization and perhaps export (Dunne and Leopold 1978, Odum et al. 1978/*FL). The usual desynchronization of export from headwater watersheds may result in more diversified downstream food chains. In lower stream reaches, considerable production may be buried in infrequently flooded, off-channel wetlands typical of the usually flatter downstream terrain.

Potential Importance to Function: Moderate.

Measure: Determination of the wetland's watershed area. The thresholds for this predictor are arbitrary.

Directness of Measure: Low.

5. Wetland/Watershed Area Ratio

Ranking: Wetlands that comprise a large portion of their watershed are more likely to export significant quantities of their production downstream.

Rationale: The larger a wetland, the greater its role (relative to other potential sources of production in a given landscape) for exporting production.

Confidence in Ranking: Low. Proportionately large wetlands may mediate runoff surges that otherwise could help export production from wetlands further downstream. They may also experience proportionately greater internal cycling (as opposed to export) of their production.

Potential Importance to Function: Low.

Measure: Determination of the ratio of wetland to watershed area. The 20-percent threshold is arbitrary.

Directness of Measure: Low.

7. Gradient

Ranking: Wetlands with steep gradients are more likely to export production than are those with gradual gradients.

Rationale: The steeper the gradient, the greater the flow velocity and the greater the potential for export. At least in the case of forested wetlands, primary productivity may also be greater in flowing than stagnant waters (Heinselman

1970/MN:P; Hynes 1970/*R; Conner and Day 1976/LA:fo; Gosselink and Turner 1978/*; Odum 1978/*FL,1979, 1981; Brown et al. 1979/*fo; Fredrickson 1979/MO:fo; Balling et al. 1980/CA:Eem; Day et al. 1980/LA:fo; Mendelssohn and Seneca 1980/NC:Eem; Phillips 1980/*Eab; Brown 1981/FL:Pfo; Horner and Welch 1981/WA:Rab). Thus, the potential for production export may be greater. Primary productivity, for example, can be 40 percent greater in flowing versus still waters (Brown et al. 1979/*fo).

Confidence in Ranking: Moderate to high. High flow velocities can physically damage plants, thus reducing productivity. In these cases, reduced stream gradient may enhance productivity (Minshall et al. 1983/PA,OR,IL:R). See also Predictor 41 - Velocity (spatially dominant).

Potential Importance to Function: Moderate. Lower gradients are likelier to support more extensive stands of wetland vegetation. Seasonal peak biomass of lacustrine submersed aquatic plants can be predicted from gradient, using equations of Duarte and Kalff (1986) and Duarte et al. (1986).

Measure: Determination of channel gradients from topographic maps.

Soundness of Measure: Moderate.

8. Inlets, Outlets

Ranking: Wetlands with outlets are more likely to export their production than those without outlets.

Rationale: Production normally cannot be efficiently transported out of wetlands unless there is at least an intermittent connection to other areas (Odum and Heald 1975/*Efo). Presence of an inlet as well as an outlet suggests better flushing, and thus, at least in the case of some forested wetlands, higher production (Brown and Lugo 1982/fo). In cool regions, flow-through wetlands generally have more dynamic water levels than isolated wetlands; nutrient mobilization and export are thus likely to be greater. Flushing may also prevent buildup of toxic conditions (e.g., excessive peat accumulation in cool climates, excessive salinity in potholes, excessive sulfide buildup in salt marshes (e.g., Portnoy et al. 1987/M:Eem), which could limit production and consequently its export.

Confidence in Ranking: Moderate to high. Some production in isolated wetlands can be exported by living animals (e.g., emerging insects, migrant birds). Primary productivity differences between natural and semi-impounded salt marshes can be only minor, although the form and timing of exported production may differ (McKellar et al. 1987/SC:E,P). Prolonged inundation can limit production in forested wetlands.

Potential Importance to Function: High.

Measure: Determination of inlets and outlets for surface water flow.

Directness of Measure: High.

10. Wetland System

Wetland system is used in the interpretation keys as a classification variable rather than a predictor. It is used to separate wetland types prior to further analyses.

11. Fringe Wetland or Island

Ranking: Fringe wetlands are more likely to export their production than are nonfringe wetlands.

Rationale: Because fringe wetlands occur along a channel or adjacent to a standing body of water, such wetlands readily convey material to downstream areas. When fringe wetlands are inundated during flooding events or by tidal action, production can be exported to adjacent bodies of water by receding waters or ebb tides, respectively (e.g., Hackney and Bishop 1981/MS:Eem).

Confidence in Ranking: High.

Potential Importance to Function: Moderate. Fringe wetlands may have less production available for export, as a result of stress from winds, currents, and ice scouring which removes propagules before plants can become established.

Measure: Determination of whether the wetland is a fringe wetland.

Directness of Measure: Moderate.

12. Vegetation Class/Subclass (Primary)

Ranking: Wetlands dominated by moss-lichen vegetation are less likely to export production. Of the remaining vegetation classes, areas dominated by forest and scrub-shrub vegetation are less likely to have large amounts of production available for export than are those dominated by emergent or aquatic bed vegetation.

Rationale: In nontidal systems, emergent and nonrooted aquatic bed species are generally more productive than species of other vegetation classes (Pomeroy and Wiegert 1981, Nixon 1982/*Eem, Teal 1986/*). Aquatic bed vegetation can benefit production export by transferring nutrients from the sediment to the water column. It usually decomposes more rapidly than emergent vegetation (Findlay and Tenore 1982/Eem) and may be more nutritious (see

Section 3.8, Predictor 12, for discussion and references pertaining to the relative utilization of different types of wetland plants).

Wetlands with woody vegetation usually have lower primary productivity than nonwooded wetlands (Brinson et al. 1981a/*fo), and their detritus may decompose more slowly. Moss-lichen wetlands (except those which are riverine, see Naiman 1983/QUE:R) are usually the least productive of vegetated wetlands and less important to nutrient cycling because acidity and associated conditions inhibit decomposition (Heinselman 1970/MN:P, Small 1972/P) and plant growth (Richardson et al. 1978/MI:Pem).

Open water, even if defined as wetlands, is likely to have lower net annual primary productivity than vegetated areas, and as a result, lower potential for production export. However, less visible benthic microalgae and phytoplankton which typify these areas can be very productive (Nixon 1980/*).

In tidal systems, aquatic bed species can be at least as productive as emergent species (Bach et al. 1986/NC:Eab) and are generally more productive than their freshwater counterparts (Stevenson 1988/*ab). Tidal scrub-shrub species (e.g., mangroves) can be more important exporters of detritus than their freshwater counterparts, the riverine swamps (Lugo et al. 1988/*fo). Tables that catalog available productivity estimates of wetland plants are given in Adamus and Stockwell (1983/*).

Confidence in Ranking: *Moderate.* Montane bogs (moss-lichen wetlands) with outlets can be locally important exporters of detritus (Erman and Chouteau 1979/CA:R). Forested floodplain wetlands in some watersheds may be effective in exporting their production (e.g., 13 g C/m^2/year; Mattraw and Elder 1984/FL:fo, Lambou 1985/LA:fo).

Potential Importance to Function: High. In some riverine systems, vegetation type is more important than stream size or position in determining the amount, timing, and quality of available production (Sidle 1986). However, the total amount of detritus may be more important (for supporting consumers) than its type (Culp and Davies 1985/BC:R).

Measure: Determination of dominant vegetation class and subclass.

Directness of Measure: Low. Direct measurements of community production and its export are preferred.

13. Vegetation Class/Subclass (Secondary)

Ranking: Wetlands in which emergent vegetation and aquatic beds occupy much of the total wetland area are more likely to have considerable production available for export.

Rationale: See Predictor 12 above.

Confidence in Ranking: Moderate.

Potential Importance to Function: High.

Measure: Determination of vegetation classes and subclasses that occupy at least 1 acre or 10 percent of the wetland. The 1-acre and 10-percent thresholds are arbitrary.

Directness of Measure: Low.

15. Vegetation/Water Interspersion

The influence of vegetation-water interspersion as a predictor for wetland functions depends upon two factors: (1) vegetation interspersion and (2) whether water flows through the wetland as sheet or channel flow. As a result, vegetation-water interspersion is defined with two predictors.

15.1 Vegetation Interspersion

Ranking: Wetlands with a high degree of vegetation-water interspersion are more likely to export their production than those with very sparse or extremely dense vegetation.

Rationale: Low production is typically evidenced by very low vegetation density. Sparse vegetation may also be incapable of retaining much drifting organic matter, which is vital as a shelter and food source for most invertebrates (Anderson and Sedell 1979). Very dense vegetation may impede water circulation, reducing the dispersal and export of food sources. Also, shading from dense stands of emergent vegetation can limit the productivity of potentially important benthic algae (Zedler et al. 1980/CA:Eem).

Confidence in Ranking: Moderate. Small pools within an emergent wetland may be depressional and thus trap organic matter rather than allowing sustained export (Borey et al. 1983/TX:Eem). Sparse vegetation may merely reflect excessive shading by algae (Steward and Ornes 1975/FL, Phillips et al. 1978, Davis and Brinson 1980/*ab, Phillips 1980/*Eab, Niemeier and Hubert 1986/IA:L), rather than reduced wetland production.

Potential Importance to Function: Moderate.

Measure: Determination of relative interspersion of vegetation and water.

Directness of Measure: Low.

15.2 Sheet Versus Channel Flow

Ranking: Wetlands in which flow occurs mostly as sheet flow are more likely to export their production than are those with predominantly channel flow.

Rationale: The greater the degree of contact between vegetation and moving water, the greater the potential for production export (Nelson and Kadlec 1984/*Pem). Nitrogen fixation, which can result in increased nitrogen available for export, is generally highest at the vegetation-open water interface (Phillips 1984/*Eab).

Confidence in Ranking: Moderate.

Potential Importance to Function: High.

Measure: Determination of the type of flow occurring in the wetland.

Directness of Measure: Low to moderate.

19. Fetch/Exposure

Ranking: Wetlands that are moderately sheltered are more likely to export production than are those that are well sheltered or extremely exposed.

Rationale: Within limits, the greater the fetch, the greater the wave energy, vertical mixing, and potential export of organic materials and associated nutrients (Gallagher 1978/*, Gosselink and Turner 1978/*, Knutson et al. 1982/VA:Eem, Nelson and Kadlec 1984/*Pem). Up to 25 percent of eelgrass detritus in open water is exported, but in sheltered areas, over 90 percent remains within the bed or is exported only a short distance to surrounding marshes and beaches (Thayer et al. 1978/*Eab).

Moderate wave energy also increases productivity of some plants (Tiner 1981, Topinka et al. 1981/ME:E,M), and thus the amount of food exported and potentially available (see Section 3.5 for quantification). Phytoplankton growth and chlorophyll-a concentrations are higher in moderately vegetated wetlands than in open, exposed situations (Rabe and Gibson 1984/Lab). Excessive fetch, by supporting the continual mixing and replenishment of water column nutrients, could favor algal growth at the expense of vascular plant growth.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Determination of fetch and the area sheltered by vegetation or topographic relief.

Directness of Measure: Low.

22. Flow, Gradient, Deposition

Ranking: Wetlands that have flows sufficient to moderately or seasonally scour the wetland are more likely to export their production than are those that are either not scoured or are severely scoured.

Rationale: The potential for production export increases with increasing flow velocities (Heinle and Flemer 1976, Mulholland 1981/NC:fo). Scouring physically removes and exports accumulated nutrients. Occasional scouring may also thin stands of wetland plants, thus enhancing circulation, reproductive vigor, and productivity. However, severe and frequent scouring can denude wetlands for long periods, resulting in decreased production, and consequently decreased production export.

Confidence in Ranking: Low.

Potential Importance to Function: High.

Measure: Determination of whether scouring is present or if the potential for scouring exists (flow present).

Directness of Measure: Low.

28. Direct Alteration

Ranking: Unaltered wetlands are more likely to have greater production and more useful export regimes than are those altered by tilling, filling, excavation, adding outlets, or blocking inlets.

Rationale: Tilling, filling, or excavating wetlands removes and buries wetland vegetation, and thus precludes or decreases production export. Adding an outlet where none previously existed and blocking an inlet can diminish the frequency and duration of wetland flooding and can cause toxic levels of sulfide (King et al. 1982/GA:Eem). Production and its export may consequently be less (Odum et al. 1979/*Eem). The 3-year period is arbitrary.

Confidence in Ranking: Moderate. The effect depends also on the type and extent of activity, how the alteration was accomplished, wetland type, and other factors.

Potential Importance to Function: Moderate.

Measure: Determination of whether the wetland has been tilled, filled, or excavated in the past 3 years or if an outlet has been added or an inlet has been blocked.

Directness of Measure: Low.

31. Water/Vegetation Proportions

Ranking: Estuarine, marine, palustrine, or lacustrine wetlands with at least 10 percent of their total area covered by visible, standing, surface water are more likely to export production. Riverine wetlands in which the area of aquatic bed vegetation is larger than the unvegetated submerged areas are more likely to export production.

Rationale: Since production is exported from wetlands by water, wetland vegetation must be flooded to effectively export above-ground production (Nelson and Kadlec 1984/*Pem).

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Determination of what percentage of the wetland is covered by standing water. The 10-percent threshold is arbitrary. If the wetland is riverine, determination of whether the area of submerged vegetation is larger than open-water areas.

Directness of Measure: Moderate.

34. Water Level Control

Ranking: Wetlands without artificial water control structures are more likely to export production than are those with such structures.

Rationale: Water level control structures can reduce discharge and the flooding frequency of downstream wetlands, and thus decrease production export (Kondratieff and Simmons 1984, Voshell and Parker 1985/VA:R, Perry and Sheldon 1986/MT:L,R). Water control structures that reduce water flow through wetlands can decrease productivity or even kill vegetation (Quennerstadt 1958, Lantz et al. 1967/LA:ab, Conner and Day 1976/LA:fo, Sklar and Conner 1979/LA:fo, Wright and Szluha 1980, Odum 1981/*, Klimas 1987/TX:Lfo).

Confidence in Ranking: Low. The effect depends on the particular type of control structure, its proximity, volume of flow intercepted, and other factors.

Potential Importance to Function: Moderate.

Measure: Documentation of the existence of an artificial control structure that influences the wetland's hydrology.

Directness of Measure: Moderate.

35. Flooding Extent and Duration

Ranking: Wetlands where the extent and duration of flooding are intermediate are more likely to export production.

Rationale: Seasonal flooding enhances the decomposition and export of detritus (Burns 1978/FL:fo, Frederickson 1979/MO:fo, Mattraw and Elder 1984/FL:fo, Lambou 1985/LA:fo) and increases the access of consumer organisms to this potential food source, thus further enhancing its dispersal. Also, the productivity of forested and emergent wetlands is often greater where they are seasonally flooded rather than permanently flooded or flooded for long periods (Broadfoot and Williston 1973; Chapman 1976/*E; Burns 1978/FL:fo; Fredrickson 1979/MO:fo, Odum 1979/*fo, 1981; Day et al. 1980/LA:fo; Hook 1984). Increased decomposition of leaf litter can be associated with longer durations of inundation (Cuffney 1988, Gurtz and Tate 1988). Although the relatively dry portions of wetlands are often most productive (e.g., Elder and Cairns 1982/FL:fo, Cuffney 1988), their production is less susceptible to being exported (Cuffney 1988).

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate to high.

Measure: Determination of the extent and duration of flooding in relation to threshold levels. Threshold levels are from US Army Corps of Engineers (1980) and are arbitrary.

Directness of Measure: Low to moderate.

36. Vegetated Width

Ranking: Wetlands in which the average width of the area dominated by emergent, scrub-shrub or forested vegetation is greater than 20 feet are more likely to export production than those with vegetated widths less than 20 feet.

Rationale: Wider wetlands have more production available for potential export.

Confidence in Ranking: Moderate. Production per unit area is not necessarily greater in wide wetlands (see Predictor 2).

Potential Importance to Function: Moderate.

Measure: Determination of the width of the area dominated by emergent, scrub-shrub, or forested vegetation in the wetland. The 20-foot threshold for width is arbitrary.

Directness of Measure: Low. Proportionate width (rather than absolute width) may be a better indicator.

41. Velocity (spatially dominant)

Ranking: Wetlands in which flow is moderate are more likely to have useful regimes for exporting production than those in which flow is either slow or very rapid.

Rationale: Within limits, the potential for production export increases with increasing flow velocities (Silberhorn et al. 1974/VA:Eem, Heinle and Flemer 1976, Gosselink and Turner 1978/*, Odum 1980/*Eem, Mulholland 1981/NC:fo). Primary productivity and decomposition rates are usually higher in wetlands with flowing waters (Heinselman 1970/MN:P; Hynes 1970/*R; Chapman 1976/*E; Conner and Day 1976/LA:fo; Odum 1978/*fo,1979/*fo; Gosselink and Turner 1978/*; Brown et al. 1979/*fo; Fredrickson 1979/MO:fo; Balling et al. 1980/CA:Eem; Day et al. 1980/LA:fo; Mendelssohn and Seneca 1980/NC:Eem; Phillips 1980/*Eab; Brown 1981/FL:Pfo; Phillips 1984/*Eab), thus increasing the potential for production export. For example, Brown et al. (1979/*fo) found that net primary productivity was 40 percent greater in wetlands with flowing waters as compared to those with stagnant waters. Extreme flow velocities stress plants, both physically and by reducing light through suspension of sediments (Hynes 1970/*R, Chapman 1976/*E,1977/*E; Davis and Brinson 1980/*ab, Phillips 1980/*Eab, Topinka et al. 1981/ME:E,M). Eelgrass, an aquatic bed species, can withstand a maximum velocity of about 4.9 fps (Fonseca et al. 1983/RI,NC:ab) and has a maximum standing crop at about 1.5 fps (Zieman 1982/*FL:ab).

Confidence in Ranking: High.

Potential Importance to Function: Moderate.

Measure: Determination of the water velocity throughout most of the wetland during annual peak flow.

Directness of Measure: Low to moderate.

45. Substrate Type (spatially dominant)

Ranking: Wetlands containing a substrate type other than sand are more likely to export production.

Rationale: Primary productivity is usually low on sand (and sometimes cobble-gravel) substrates because of low nutrient availability and instability (Harms 1973/FL:fo; Chapman 1976/*E,1977/*E).

Confidence in Ranking: Low to moderate. Tidal sand substrates can have exceptional primary productivity rates, particularly if irregularly exposed (Shaffer 1988/LA:Eab), as well as high macroinvertebrate densities (Franz and Harris 1988/NY:E).

Potential Importance to Function: Moderate to high.

Measure: Determination of the dominant surface substrate in the wetland.

Directness of Measure: Low.

47. pH

Ranking: Wetlands where the pH is circumneutral (6.0 to 8.5) are more likely to support substantial production than are those with more acidic or alkaline pH values.

Rationale: Productivity of wetland vascular plants is greatest at circumneutral pH, and as a result, more material is available for export (Heinselman 1970/MN:P, Small 1972/P). The rate of decomposition is low under acidic conditions (Heinselman 1970/MN:P, Chamie and Richardson 1978/*p, Murray and Hodson 1984/GA:P), and lower pH generally results in impoverishment of fauna, especially fish, with consequent reduction in the ability to disperse or cycle any production. Higher pH (within the circumneutral range) generally results in better buffering and higher productivity (Cook and Powers 1958/NY:Pem).

Confidence in Ranking: High.

Potential Importance to Function: Moderate.

Measure: Determination of the pH of the water in the wetland.

Soundness of Measure: Low to moderate.

51. Plant Productivity

Ranking: Wetlands with high primary productivity are more likely to export production.

Rationale: The greater the productivity, the greater the amount of organic matter potentially available for export.

Confidence in Ranking: High.

Potential Importance to Function: Moderate to high. Flushing action and its timing are at least as important.

Measure: Determination of the net annual above-ground.productivity.

Directness of Measure: High.

55. Suspended Solids

Ranking: Wetlands having lower suspended solids concentrations are more likely to support sufficient production for eventual export.

Rationale: Turbidity reduces light penetration and consequently primary productivity, at least among algae and submersed aquatic bed species (Strawn 1961/FL:Mab, Odum 1963/TX:Eab, Hart and Fuller 1972/MD:E, Thayer et al. 1975/*Eab, McNabb 1976/*ab, Peterson and Peterson 1979/*NC:E, Davis and Brinson 1980/*ab, Phillips 1980/*Eab, Diaz et al. 1982/*E, Duarte et al. 1986).

A 25 NTU (about 100 mg/l) increase in turbidity in a shallow riverine wetland can reduce production of algae and submersed aquatics by 50 percent (Lloyd et al. 1987/AK:R), and a mere 5 NTU increase (about 20 mg/l) has been shown to reduce the productive area of a lake by about 80 percent (Lloyd et al. 1987/*).

If deposited, suspended solids may smother some wetland plants by reducing oxygen diffusion into the substrate (Featherly 1940, Yeager 1949/IL:fo, King and Ball 1967/MI:R), although this is not inevitable. Wetlands dominated by robust emergent and woody species can probably tolerate sediment accretion rates of less than 5 cm per year (van der Valk et al. 1983/AK:em). Suspended solids can also precipitate and bury phytoplankton, thus inhibiting export of their production (Lackey et al. 1959/FL). Suspended solid loads in riverine wetlands may scour shallow-rooted wetland plants, resulting in reduced productivity.

Confidence in Ranking: Low to moderate. Incoming suspended solids usually carry adsorbed nutrients and thus may benefit primary productivity if light penetration is maintained (Copeland and Dickens 1969/*E, Kaplan et al. 1974, Karr and Schlosser 1977/*, Gosselink and Turner 1978/*, Boto and Patrick 1979/*). Detritus can also decompose faster when a fine layer of silt is present (Brinson 1977/NC:fo, Gurtz and Tate 1988). Moderate sediment subsidies may also be essential in erosive environments to physically augment the fine substrate necessary for most wetland plants (Chapman 1976/*E).

Potential Importance to Function: Moderate.

Measure: Determination of suspended solids level or Secchi disc reading of runoff or surface water entering the wetland. The maximum depth at which submersed plants can grow is generally 2.7 to 5 times the Secchi depth (Cole 1975/*). Thresholds used in Volume II are from data summarized by Wesche and Rechard (1980/*R) and Lloyd et al. (1987/AK:R).

Directness of Measure: Moderate.

56. Dissolved Solids or Alkalinity

Ranking: Inland wetlands with moderate alkalinity levels are more likely to support greater primary productivity and thus have more production available for export.

Rationale: Within reasonable limits, increased alkalinity benefits aquatic plant growth and primary productivity in freshwater systems due to increased nutrient availability and buffering capacity (Jenkins 1982/L). As a result, fish standing crop, and thus the potential for production being exported, usually increases with increasing total dissolved solids or alkalinity levels (Ryder 1965, Jenkins and Morais 1971/L, Jenkins 1982/L). However, in very alkaline waters (perhaps 150 to 250 ppm), phosphorus may become less available and ammonia may be converted to highly toxic ammonium.

Confidence in Ranking: Moderate. High levels of dissolved solids are often accompanied by watershed contamination with other pollutants, which reduce production.

Potential Importance to Function: Moderate to high.

Measure: Determination of alkalinity ($CaCO_3$) levels and the morphedaphic index in relation to threshold levels. The 20-mg/l threshold for alkalinity is similar to that of Fried (1974/*NY), Larson (1976/*MA), and many state water quality codes; the morphedaphic index (Ryder 1965) thresholds are from Jenkins and Morais (1971/L).

Directness of Measure: Moderate.

57. Eutrophic Condition

Ranking: Wetlands with moderate or high nutrient levels and loading rates are more likely to sustain higher production for eventual export.

Rationale: Insufficient concentrations or ratios of inorganic nutrients (especially of phosphorus in freshwater systems and nitrogen in saltwater systems) can frequently limit production of algae and vascular aquatic plants, as well as limiting secondary production (Welch et al. 1988), which otherwise makes primary production more available for export (Watson et al. 1984/AU:Eab).

Production in contiguous forested wetlands may be increased by increasing the phosphorus loading rates to at least 50 g/m^2/year (Brown and Lugo 1982/fo, Lugo et al. 1988/*fo). In lakes, nitrogen concentrations, in conjunction with lake size and mean depth, are a statistically significant predictor of emergent cover (Smith and Wallsten 1986). Nitrogen-loaded runoff can improve the palatability of some wetland plants to consumers by reducing plant phenolic content (Buchsbaum et al. 1981).

Invertebrate richness (Perry and Sheldon 1986/MT:L,R) and diatom richness (Marcus 1980) are higher below than above eutrophic lakes. The effect of more productive lakes extends farther downstream (more than 100 meters) than that of less productive ones (Perry and Sheldon 1986/MT:L,R). Lakes (and by inference, wetlands) can be effective production exporters (Cushing 1964/SAS:L,R, Maciolek and Tunzi 1968/NV:R,L, Gibson and Gilbraith 1975/QUE:R,L). Particulate organic matter concentrations of 300 to 600 mg/m^3 in the outfalls of lakes tended to produce the greatest invertebrate species richness in 13 Montana streams (Perry and Sheldon 1986/MT:L,R).

The levels and proportions of inorganic nutrients also influence the plant species composition of the wetland and thus the quality and availability of various types of food sources for production export. Freshwater plant communities, unless highly enriched, tend to be more limited by phosphorus than nitrogen (Loucks and Watson 1978/WI:L, Schindler 1978/MAN:L, Bowden 1986/MA:tem, Richardson and Marshall 1986/MI:Pem), whereas estuarine and marine wetlands tend at least seasonally to be more nitrogen-limited than phosphorus-limited (Broome et al. 1975/NC:Eem; Valiela et al. 1975/MA:Eem,1976/MA:Eem; Patrick et al. 1976/LA:E; Linthurst and Seneca 1980/NC:Eem). Riverine wetlands (if permanently flooded) are the least likely to be nutrient-limited (Farnworth et al. 1979/*).

High nutrient levels often favor phytoplankton over macrophytes (especially submersed aquatic bed species), which sometimes can be shaded out by phytoplankton blooms (Steward and Ornes 1975/FL, Phillips et al. 1978, Davis and Brinson 1980/*ab, Phillips 1980/*Eab, Niemeier and Hubert 1986/IA:L), although exceptions (e.g., Lee and Olsen 1985/RI:E) may occur. High nitrogen concentrations relative to phosphorus favor green algae and flagellate species, whereas high phosphorus-to-nitrogen ratios favor blue-green algae. Green algae and flagellates are usually, but not always, more easily utilized in food chains than are blue-green algae (Loucks and Watson 1978/WI:L). Elevated nutrient levels can also mitigate the adverse effects of some contaminants on ecosystem function (e.g., Fairchild et al. 1984).

Confidence in Ranking: Moderate. Excessive nutrient additions can inhibit the minor contribution of nitrogen fixation to production export (Carpenter et al. 1978/MA:Eem). Also, eutrophic conditions typically co-occur with factors which restrict production (e.g., anoxia, contaminants).

Potential Importance to Function: Moderate to high. Excess production can inhibit flushing and export. Factors such as depth, redox potential (Linthurst 1980/NC:Eem, Howes et al. 1981/MA:Eem, Morris 1984/SC:Eem, Morris and Dacey 1984/MA,SC:Eem), gradient, and wave exposure may override the effects of nutrient additions on plant biomass (Dennison and Alberte 1985/ab, Duarte and Kalff 1988/VT:Lab).

Measure: Determination of water nutrient levels or their indicators in relation to threshold levels. Thresholds are from Vollenweider (1976), Binns and Eiserman (1979/WY:R), and Taylor et al. (1980/*).

Directness of Measure: High.

3.8 Aquatic Diversity/Abundance

1. Climate

Ranking: Estuarine wetlands located in an intense storm region, or those with smaller tidal ranges and less erosive rainfall, are more likely to have a great on-site diversity and/or abundance of fish and invertebrates. In addition, lacustrine and palustrine wetlands that remain unfrozen for more than 1 month during most winters are likely to have a relatively great on-site diversity and/or abundance of fish and invertebrates.

Rationale: These factors are correlative but probably not causal. Intense storm regions and rainfall erosivity are generally correlated with high nutrient inputs (Odum et al. 1979/*fo), high productivity, and high diversity of aquatic organisms (Odum et al. 1979/*Eem). Lacustrine and palustrine wetlands that are continuously frozen during most winters can have low fish and invertebrate diversity because many species are limited by low dissolved oxygen concentrations (Tonn and Magnuson 1982/WI:L,P). However, a high tidal range in these situations would likely override any freshwater or nutrient inputs associated with storm runoff. Prolonged ice cover, particularly if covered by snow, results in depletion of dissolved oxygen and death for many fish and invertebrates (Tonn and Magnuson 1982/WI:L,P).

Confidence in Ranking: Moderate (ice factor) to low (storm, tidal, rain erosion factors).

Potential Importance to Function: Moderate.

Measure: Determination of whether estuarine wetlands are located in intense storm regions, in areas where the rainfall erosivity factor is greater than 300, and where the tidal range is less than 3 feet. For lacustrine and palustrine wetlands, a determination of how long the wetland is frozen during most winters. The thresholds are arbitrary.

Directness of Measure: Low.

2. Acreage

Ranking: Wetlands larger than 40 acres (measurement includes unconstricted contiguous waters) are more likely to exhibit great fish and invertebrate diversity and/or abundance than small wetlands.

Rationale: For lakes in general, fish species diversity increases with increasing surface area and length of shoreline (Barbour and Brown 1974, Moyle and Cech 1982*, Tonn and Magnuson 1982/WI:L,P). Mollusc (Lassen 1975, Aho 1978/EU:L), midge (Driver 1977/MAN:Pem), and crustacean (Fryer 1985) richness also increases with increasing lake area.

Confidence in Ranking: Moderate. Fish densities in small permanent floodplain ponds showed no significant increase with pond size (Cobb et al. 1984).

Potential Importance to Function: Moderate.

Measure: Determination of the size of the wetland and any accessible wetlands within 1 mile. Acreage thresholds of 40 acres for lacustrine and 200 acres for palustrine wetlands are arbitrary.

Directness of Measure: Low. Size may merely be a surrogate for habitat complexity, hydroperiod permanency, or chemical stability, all of which may have similar beneficial effects. Distance from similar habitat is also an important predictor (Driver 1977/MAN:Pem).

4. Location and Size

Ranking: Wetlands that are tidally influenced, located near large water bodies, or within large watersheds are more likely to provide a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Distinct shifts in diversity and community composition of both fish and their invertebrate prey occur as watershed area increases (i.e., moving downstream toward lower elevations, Harrell et al. 1967/OK:R, Lotrich 1973/KY:R, Vannote et al. 1980/*R). In riverine systems in most regions, food is more available and flows are more dependable in higher order streams and low-elevation wetlands, which tend to be found in larger watersheds. Invertebrate species richness (Minshall et al. 1985/ID:R), macrophyte diversity (Sheath et al. 1986/RI:Rab), and fish productivity (Lotrich 1973/KY:R) are generally higher in streams of order 3 or higher. Fish yields locally (Moulton 1970/MA; Tonn and Magnuson 1982/WI:L,P; Rahel 1984/WI:L) and worldwide (Welcomme 1979/*) have been positively correlated with watershed area. Tidal marshes, unlike riverine systems, **may have higher** fish and invertebrate productivity in the headwater (second and third order) creeks than in higher order tidal creeks (Weinstein 1979/NC:Eem, Rozas and Odum 1987b/VA:tem).

Confidence in Ranking: Moderate. Invertebrate richness failed to show any correlation with stream order or width in the lake-outlet study of Perry and Sheldon (1986/MT:L,R).

Potential Importance to Function: Low. Greater human disturbance, land cover changes, and increased prevalence of exotic species in downstream areas frequently have an overriding effect and cause diversity to decline (Goldstein 1981). Stream size and location may be more important in small coastal island streams (Bronmark et al. 1984/EU:R).

Measure: Determination of the size of the wetland's watershed. Associated riverine widths and flow characteristics can be predicted from watershed area using equations of Thomas and Benson (1970), Osterkamp and Hedman (1982/NE:R), and others.

Directness of Measure: Low. Richness generally appears to increase the most between stream orders 3 and 4, and the 100-square mile threshold for watershed size is a crude approximation of the acreage typically associated with this transition (Dunne and Leopold 1978).

5. Wetland/Watershed Area Ratio

Ranking: Wetlands located in watersheds with many other wetlands are more likely to have greater on-site diversity and/or abundance of fish and invertebrates.

Rationale: The diversity and abundance of fish and invertebrates may be greater where there are opportunities for input of nutrients and immigration of aquatic organisms from upslope areas (Hatcher 1973/TN:L. Tonn and Magnuson 1982/WI:L,P, Rahel 1984/WI:L). Connected wetlands form protective corridors for fish to migrate and disperse among habitats (Brinson et al. 1981a/*fo).

Confidence in Ranking: Moderate. The presence of upstream wetlands may have a depressive effect on some macroinvertebrate communities (Smock et al. 1985/SC:R) and phytoplankton (Guildford et al. 1987/MAN:L), perhaps due to lower pH, less available iron and nutrients, and dissolved oxygen depletion associated with intense microbial activity and leaching of humic acids in wetlands.

Potential Importance to Function: Low.

Measure: Determination of whether upslope wetlands comprise more than 5 percent of the total watershed of the wetland. The 5-percent threshold is arbitrary.

Directness of Measure: Low.

7. Gradient

Ranking: Riverine wetlands that have annual floodplains wider than their channels or those with low channel gradients are more likely to support a great on-site diversity of fish and invertebrates.

Rationale: Wetlands with annual floodplains wider than their channels and those with low channel gradients generally have low water velocities. Wetlands with low water velocities usually have greater fish and invertebrate diversity than wetlands with fast-flowing waters (Moyle and Cech 1982/*, Scarnecchia and Bergersen 1987/CO:R).

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Determination of whether the annual floodplain is wider than the wetland channel, or comparison of the channel gradient to threshold levels given in Volume II.

Directness of Measure: Low.

8. Inlets, Outlets

Ranking: Wetlands are ranked as follows from highest to lowest with respect to their probabilities for supporting a notably great on-site diversity and/or abundance of fish and invertebrates: the wetland has (1) both an inlet and outlet, (2) either an inlet or an outlet, and (3) no inlet or outlet.

Rationale: Connections to adjacent waters permit movements by fish and invertebrates, permit access to other colonizing organisms (Aho 1978/EU:L, Luey and Adelman 1980/MN:R, Gilmore et al. 1981/FL:Eem, Harrington and Harrington 1982/FL:Eem, Barnby et al. 1985, Neill and Turner 1987/LA:Eem, Rey and Crossman 1987/FL:E,P), and allow possible input of allochthonous material. Fish may use tributaries for spawning or nurseries. In ice-hazard regions, fish may use tributaries as winter refuge from anoxia (Johnson and Moyle 1969/MN:Lem, Tonn and Magnuson 1982/WI:L,P). In a disturbed stream, invertebrate populations 200 meters closer to a source population recovered twice as fast as those farther downstream (Gore 1982/WY:R). In marine and estuarine systems, fully contiguous wetlands tend to have greater freshwater inflow, which may enhance the diversity, and sometimes the abundance, of aquatic life (Cross and Williams 1981/*, Barnes 1988). See also Predictor 48.

Confidence in Ranking: Low. In some inland regions (e.g., Prairie Pothole), isolated wetlands have a much higher diversity and abundance of aquatic life than connected wetlands. Some impounded tidal marshes have greater fish abundance, though lower diversity (Rey and Crossman 1987/FL:E,P).

Potential Importance to Function: High.

Measure: Documentation of inlets and outlets for surface water flow.

Directness of Measure: High.

10. Wetland System

Wetland system is used in the interpretation keys as a classification variable rather than a predictor. It is used only to separate wetland types prior to further analyses.

11. Fringe Wetland or Island

Ranking: Lacustrine and palustrine wetlands that form at least part of a fringe wetland or an island are more likely to have a great on-site diversity and/or abundance of fish and invertebrates than nonfringe wetlands or those not associated with islands.

Rationale: Fringe wetlands are at the interface between terrestrial and aquatic communities where both density and diversity of species are high (Odum 1979/*fo). Many species concentrate at the wetland vegetation's border (fringe) with open water (Holt et al. 1983, Macdonald et al. 1987/BC:Eem). Fringe and island wetlands have a high degree of connectivity with contiguous channels or standing bodies of water and provide physical habitat complexity. Fish and invertebrate species richness/abundance are increased in fringe and island wetlands by their easy accessibility to fauna from surrounding waters.

Confidence in Ranking: Low. The increased exposure to wind and currents that characterizes fringe wetlands may result in lower densities of invertebrates (Jonasson and Lindegaard 1979).

Potential Importance to Function: Low.

Measure: Determination of whether the wetland composes all or part of a fringe wetland or an island.

Directness of Measure: Low.

12. Vegetation Class/Subclass (Primary)

Ranking: Wetlands dominated by aquatic bed vegetation are more likely to have a great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Most types of vegetated wetlands contain larger and more diverse populations of fishes and invertebrates than do unvegetated wetlands

(Percival and Whitehead 1929/R, Pennak and Van Gerpen 1947/CO:R, Minkley 1963/KY:R, Kimble and Wesche 1965, Wetzel 1975/*, Hall and Werner 1977/MI:L, Teels et al. 1978/MS, Ware and Gasaway 1978/FL:L, Thayer et al. 1979/*Eab, Menzie 1980/NY:Rab, Mittelbach 1981, Diaz et al. 1982/*E, Zajac and Whitlatch 1982/CT:E, Reice and Stiven 1983, Stoner 1983, Weinstein and Brooks 1983/NC:Eem, Durocher et al. 1984/TX:L, Floyd et al. 1984, Gilinsky 1984, Heck and Thoman 1984/VA,MD:Eab, Morin 1984/NC:L, Zimmerman and Minello 1984/TX:Eem, Chubb and Liston 1986/MI:Lem, Poe et al. 1986/MI:L, Rozas and Odum 1987a/VA:tem, Cyr and Downing 1988).

Densities of invertebrates are often greater in aquatic beds than in stands of emergent vegetation (Voights 1976/IA:P, Scheffer et al. 1984/EU:R, Schramm et al. 1987), probably because greater surface area is available in aquatic beds due to the dissected form of the leaves (Dvorak and Best 1982/Eu:Lab). The peak biomass of submersed vegetation can be predicted for lacustrine wetlands using the equations of Duarte and Kalff (1986) and Duarte et al. (1986). Algae-dominated wetlands may have exceptionally high diversity and productivity of aquatic invertebrates (Dudley et al. 1986/CA:Rab), though the algae may shade out aquatic bed macrophytes. Wetlands dominated by green algae or diatoms, for example, can provide a food source that is highly palatable to consumers (Boyd 1971/*Lab, Dudley et al. 1986/CA:Rab) and that in turn supports higher invertebrate production (Dudley et al. 1986/CA:Rab, Huryn and Wallace 1987/NC:R).

Freshwater wooded wetlands have seldom been compared to aquatic bed or emergent wetlands, with regard to relative abundance and diversity of aquatic invertebrates and fish. Density and species richness at the outer surface water edge in freshwater wooded systems appear to be generally great, both for invertebrates (Hubert and Krull 1973, Ziser 1978/LA:fo, Benke et al. 1979/GA:RRUB, Beckett et al. 1983/MS:R, Pollard et al. 1983/LA:fo, Cobb et al. 1984, Batema et al. 1985, Sklar 1985/LA, Thorp et al. 1985/GA:fo) and for fish (Guillory 1979, Welcomme 1979/*, Pollard et al. 1983/LA:fo, Ross and Baker 1983/MS:R, Stewart 1983). Among wooded wetlands, those of the broad-leaved deciduous subclass, especially alder and willow species, produce particularly great amounts of detritus especially desired by consumers (Chapman 1966, Sedell et al. 1975/*fo, Anderson and Cummins 1979, Smock and Harlowe 1983/VA:P, Sidle 1986).

In estuarine and marine systems, densities and species richness of fish and invertebrates can be greater in scrub-shrub (mangrove) wetlands than in aquatic plant beds (Thayer et al. 1987/FL:Efo), and greater in aquatic beds than in adjacent unvegetated bottoms (Menzie 1980/NY:Rab, Stoner 1983, Heck and Thoman 1984/VA,MD:Eab, Rozas and Odum 1987a/VA:tem).

Confidence in Ranking: High.

Potential Importance to Function: High. The extent and density of vegetation or detritus are probably more important than its type (Culp and Davies 1985/BC:R).

Measure: Determination of the vegetation class/subclass of the wetland.

Directness of Measure: Moderate.

13. Vegetation Class/Subclass (Secondary)

Ranking: Wetlands with at least 10 percent or 1 acre of aquatic bed vegetation are more likely to have a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: See Predictor 12 - Vegetation Class/Subclass (Primary).

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of how much of the wetland is occupied by aquatic plant beds. The thresholds are arbitrary.

Directness of Measure: Moderate.

Information Sources: See Predictor 12 - Vegetation Class/Subclass (Primary).

15. Vegetation-Water Interspersion

The influence of vegetation-water interspersion as a predictor for aquatic diversity/abundance depends upon two factors: (1) vegetation interspersion and (2) the way in which water enters the wetland. As a result, the vegetation-water interspersion is defined with two predictors.

15.1 Vegetation Interspersion

Ranking: Wetlands that contain vegetation interspersed with open water (i.e., channels, pools, or other open-water areas) are more likely to support a notably great on-site diversity and/or abundance of fish and invertebrates. Those with very dense vegetation and no channels or open-water areas are less likely to support this function.

Rationale: Some vegetation is beneficial to fish dependent on aquatic invertebrates because densities of invertebrates are high in such habitat (see Predictor 12 above). However, wetlands almost completely covered by dense vegetation (no open water) experience reduced fish movement (Cope et al. 1970; Strange et al. 1975; Vince et al. 1976/MA:E; Bailey 1978/*; Colle and Shireman 1980; Crowder and Cooper 1982; Savino and Stein 1982; Pollard et al. 1983/LA:fo; Zimmerman and Minello 1984/TX:Eem; Lodge et al. 1985/IN:L; McIvor et al./VA:tem, in press). They also have fewer large fish, lower invertebrate density (Swanson and Meyer 1977/ND), and perhaps reduced fish production. Vegetation-water interspersion is also a key factor in influencing the abundance of brown shrimp and some fish in salt marshes (Zimmerman et al. 1984/TX:Eem).

Thus, intermediate categories of interspersed vegetation (e.g., densities less than 200 g dry weight/m^2, or about 50 g dry weight/m^3) support greater fish growth (Macan 1949/*, Vince et al. 1976/MA:E, Crowder and Cooper 1982, Pardue 1983/*E,R, Wiley et al. 1984/IL:Lab, Engel 1988/WI:Lab). A submersed plant biomass of 52 g dry weight/m^3 appears to be optimal for supporting largemouth bass in small ponds (Wiley et al. 1984/IL:Lab). Plant species richness in riverine wetlands declines in stands where biomass exceeds 200 g/m^2 (Day et al. 1988).

Sinuosity of the open water-wetland plant edge, which is often associated with interspersion, has been positively correlated with fish standing crop (Buckley et al. 1976/IA:R, Zimmer and Bachman 1978/IA:R), but the relationship is less clear for adults than for juveniles (Menzel and Fierstine 1976/IA:R). Meanders (sinuosity) help dampen the scouring effect of floods (Karr and Schlosser 1977/*). Shoreline irregularity has also been positively correlated with fish standing crops (Jenkins 1967/L, Jenkins and Morais 1971/L).

Confidence in Ranking: Moderate.

Potential Importance to Function: *Moderate.*

Measure: Determination of the degree of vegetation-water interspersion in the wetland.

Directness of Measure: Low.

15.2 Sheet Versus Channel Flow

Ranking: Wetlands where (under average flow conditions) water enters in a channel and then spreads out over a wide area are more likely have a great on-site diversity and/or abundance of fish and invertebrates.

Rationale: An even distribution of water throughout a wetland provides an optimal combination of shelter from predators, complex substrates for attachment or feeding, and ample exchange of dissolved oxygen and nutrients. See also Predictor 15.1 above.

Confidence in Ranking: Moderate.

Potential Importance to Function: Low.

Measure: Documentation of whether water enters the wetland in a channel and then spreads out over a wide area within the wetland.

Directness of Measure: Low.

16. Vegetation Class Interspersion

Ranking: Wetlands with intermediate or high vegetation class interspersion are more likely have a great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Wetlands that contain an interspersed mosaic of different vegetation classes provide a greater variety of food, shelter, and other habitat requirements. Thus, wetlands with moderate/high vegetation richness and interspersion can support a greater density and species richness of aquatic animals than those with low interspersion (Weinstein and Brooks 1983/NC:Eem, Rozas and Odum 1987c/VA:tem). In palustrine wetlands, invertebrate richness is greatest where aquatic bed and emergent classes are interspersed (Voights 1976/IA:P).

Confidence in Ranking: Moderate.

Potential Importance to Function: Low.

Measure: Determination of the degree of vegetation class interspersion in the wetland.

Directness of Measure: Low.

17. Vegetation Form Richness

Ranking: Wetlands that contain numerous vegetation forms in relatively even proportions are more likely to support a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Fish diversity is greater in stream habitats with greater diversity of depths, substrate types, and currents (Gorman and Karr 1978/IN:R). Increased wetland plant form diversity and biomass have been directly correlated with fish diversity in lacustrine and palustrine systems (Tonn and Magnuson 1982/WI:L,P, Rahel 1984/WI:L). A diversity of plant forms is also likely to support a diversity of macroinvertebrates (Chapman 1966, Dvorak and Best 1982/Eu:Lab, Lodge 1985/IN:L) and provide greater food chain support due partly to differing phenologies among plants (i.e., sustainable nutrient supplies, Sheridan and Livingston 1979/FL:E).

Confidence in Ranking: Moderate.

Potential Importance to Function: High.

Measure: Determination of the number of vegetation classes or subclasses present in the wetland.

Directness of Measure: Low.

18. Shape of Upland/Wetland Edge

Ranking: Wetlands with irregular or sinuous wetland-upland edges are more likely to support a notably greater on-site diversity and/or abundance of fish and invertebrates than are those with smooth, regular edges.

Rationale: Particularly in fringe wetlands of narrow width, an irregular upland ecotone can augment habitat structure and provide shelter, thus enhancing diversity of the open water-wetland plant edge.

Confidence in Ranking: Low.

Potential Importance to Function: Low.

Measure: Determination of whether the wetland-upland border is irregular or smooth.

Directness of Measure: Moderate to low.

20. Vegetation Canopy

Ranking: Riverine wetlands with sufficient vegetation or topographic relief on adjacent banks to provide moderate shade to much of the wetland at midday are more likely to support a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Shade (indicated by the fraction of streambank forested within 1.5 miles upstream) can determine the weekly maximum water temperature in riverine systems, and thus the presence of trout (Barton et al. 1985/ONT:R) and salmon (Theurer et al. 1985/WA:fo). One stream section with 80 percent more cover had almost 80 percent more trout (Elser 1968). Canopy cover is especially important in systems where maximum summer temperatures are close to 68° F, water circulation is poor, and submerged cover is lacking (Burns 1972/FL:fo, Murphy et al. 1981/OR:fo, Scrivener and Andersen 1984/BC:R).

Although nearly complete shading may be physiologically necessary to most salmonid fish in some regions, extensive shading can also reduce fish species diversity (Murphy et al. 1981/OR:fo) and aquatic primary productivity (Naiman and Sedell 1980). Greater algal productivity is associated with moderate canopy removal (e.g., some logging) (Lyford and Gregory 1975/OR:R, Lowe et al. 1986/NC:Rab). Areas of intermediate shade and mixed shoreline or wetland vegetation types, especially deciduous cover (see Predictor 12), can thus provide an optimal mix of cooling and primary production (Minshall 1968/ID:R). Aquatic invertebrate and vertebrate densities often increase accordingly (Newbold et al. 1980/CA:fo, Murphy and Hall 1981/OR:fo, Murphy et al. 1981/OR:fo, Hawkins et al. 1982/OR:fo).

Confidence in Ranking: Moderate. Summer cooling can occur despite a lack of shade if ground water inputs are great (Bilby 1984).

Potential Importance to Function: High.

Measure: Determination of the degree to which the wetland is shaded and whether shaded and unshaded areas of the wetland are adequately interspersed.

Directness of Measure: Low. Water temperature and light measurements are more direct and thus are preferred. Also, models are available for quantifying canopy-temperature and light-photosynthesis interactions in wetlands (e.g., Cosby et al. 1984/*).

21. Land Cover of the Watershed

Ranking: Wetlands in watersheds dominated by impervious surfaces are less likely to support a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Vegetation cover changes and creation of impervious surfaces in upslope watershed areas can cause unnatural, accelerated runoff, aberrant wetland hydroperiods, and higher levels of sediment and toxicants in wetlands (Aubertin and Patric 1974). Deterioration in food quality for fishes (Sloane-Richey et al. 1981/WA:R), fish species richness, and declines in the health of some species' populations generally occur in areas of greatest watershed development and human population growth (e.g., Leonard and Orth 1986, Hughes and Gammon 1987/OR:R, Nash 1988, Steedman 1988/ONT:R). Declines in macroinvertebrate habitat quality (Hammer 1972) and functional richness also are associated with urban runoff (Pratt and Coler 1979, Pitt and Bozeman 1982, Garie and McIntosh 1986/NJ:R, Pedersen and Perkins 1986). See also Predictor 27.

Confidence in Ranking: Moderate. Although urbanization may alter the community structure of aquatic life, abundance does not decrease in all cases (e.g., Jones and Clark 1987). Intermediate levels of canopy removal and enrichment are typically associated with low-intensity land cover alteration, and these may enhance aquatic diversity or abundance (Odum 1985/*; also see Predictors 20 and 57).

Potential Importance to Function: High.

Measure: Determination of the majority of the wetlands' watershed land cover.

Directness of Measure: Low.

23. Ditches/Canals/Channelization/Levees

Ranking: Wetlands without functioning ditches, canals, levees, or similar artificial features that cause water to leave faster than would occur under natural conditions are more likely to have great on-site diversity and/or abundance of fish and invertebrates than those with such structures.

Rationale: Ditches within wetlands generally decrease the hydraulic retention time and alter the timing of inundation, both seasonally and in response to specific precipitation events. One result is that wetlands become isolated from other water bodies for longer time periods. Wetlands so altered are accessible to aquatic organisms less frequently and for shorter periods of time than natural wetlands. Levees also can block access to wetlands and prevent immigration of species (Gilmore et al. 1981/FL:Eem, Neill and Turner 1987/LA:Eem).

Channelization of rivers and streams removes both riffle areas and pools which provide habitat for many species (Bayless and Smith 1967/NC:R, Tarplee et al. 1971/NC:fo, Buckley et al. 1976/IA:R, Griswold 1978, Huish and Pardue 1978/NC:fo, Simpson et al. 1982/*, Portt et al. 1986). When marsh tidal creeks are channelized, shallow depositional creek banks that are used by juvenile invertebrates and fishes are eliminated (Rozas and Odum 1987c/VA:tem, McIvor and Odum 1988/VA:tem), and the number and biomass of these taxa are reduced (Mock 1967/TX:E, Trent et al. 1976/TX:E).

By making aquatic habitats more homogeneous, reducing access, and altering wetland chemistry (e.g., causing anoxia, acidification, and sulfur oxidation in salt marshes), ditches and channelization can reduce aquatic productivity and diversity (Mock 1967/TX:E, Balling et al. 1980/CA:Eem, Barnby and Resh 1980/CA:Eem, Gilmore et al. 1981/FL:Eem, Balling and Resh 1982/CA:Eem, Harrington and Harrington 1982/FL:Eem, Pollard et al. 1983/LA:fo, Barnby et al. 1985, Neill and Turner 1987/LA:Eem, Portnoy et al. 1987/M:Eem).

Confidence in Ranking: Moderate. The effect also depends on the type of alteration, proximity, extent, construction methods, wetland type, and other factors. Some studies have detected no major effects of channelization on aquatic communities, particularly in situations where sediment types were not altered (e.g., King and Carlander 1976/IA:R, Possardt et al. 1976/VT:R, Whitaker et al. 1979). An increase in plant richness for 6+ years after drainage was reported by Thibodeau and Nickerson (1985/MA:Pfo) and may be the result of mobilizing nutrients from the sediments. Intermediate levels of channel disturbance may increase diversity of the fish fauna (Leidy and Fiedler 1985/CA:R).

Potential Importance to Function: Moderate. Drainage may have a greater effect on wetland vegetation composition than increased seasonal inundation

(Thibodeau and Nickerson 1985/MA:Pfo), and substrate (sediment) type may be a more direct predictor of aquatic diversity than presence of channelization or ditching (Griswold 1978).

Measure: Documentation of ditches, canals, channelization, levees, or other artificial features that cause water to leave at a faster rate than would occur under natural conditions.

Directness of Measure: Low.

25. Sediment Sources

Ranking: Wetlands without sources of inorganic sediment or those that do not frequently experience activity (e.g., boating) that causes sediment resuspension are more
likely to exhibit a great on-site diversity and/or abundance of fish and invertebrates.

Rationale: See Predictor 55.

Confidence in Ranking: Moderate.

Potential Importance to Function: Low.

Measure: Determination of potential source(s) of inorganic sediment and toxicants such as storm-water outfalls, irrigation return waters, surface mines, exposed soils, erosion-prone soils, gullies, sand or gravel pits, or severely eroding stream or road banks.

Directness of Measure: Low. Direct measurement of suspended or deposited sediment and the exposure of biota to its effects is preferable to assuming presence based on stereotypical sources. If sources are assumed, then coincidence with periods of maximum runoff should be considered, as some sources are associated only with seasonally intermittent activities. Models for quantifying the transport of sediment into and through water bodies are widely available (e.g., Universal Soil Loss Equation, ANSWERS model, some Hydrologic Engineering Center models).

27. Contaminant Sources

Ranking: Wetlands without waterborne contaminants or sources that potentially contribute such contaminants are more likely to have a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Waterborne contaminants reduce the density and/or species richness of aquatic communities (Krebs and Burns 1977, Krebs and Valiela 1978/MA: Eem, Moore et al. 1979/MN:Pem), and food chains become shorter with the elimination of top predators (Odum 1985/*). See also Predictor 21.

Confidence in Ranking: High. The effect depends also on the type of contaminant, proximity, season of application, wetland type, consumer behavior, and other factors. Diversity can be increased by toxic inputs if diversity is low to begin with (Odum 1985/*), or can remain unchanged if compensating mechanisms are present (Schindler 1987/MAN:L).

Potential Importance to Function: High. Atmospheric sources of metals and synthetic organics (via deposition or precipitation) can be significant as well (Lazrus et al. 1970, Rappaport et al. 1985/U.S.).

Measure: Determination of potential sources of waterborne toxic substances such as mines, landfills, leaking subsurface tanks, salt/brine seepage, pesticide-treated areas, contaminated aquifers, severe oil runoff, irrigation return water, industrial and sewage outfall, or heavily traveled roads.

Directness of Measure: Low. Direct measurement of contaminant levels or (better yet) exposure of aquatic life and accumulation in tissues is preferable to assuming presence based on stereotypical sources. If sources are assumed, then coincidence with periods of maximum runoff and fish or invertebrate presence should be considered, since some sources are associated only with seasonally intermittent activities. Models for quantifying the transport of contaminants are widely available (e.g., for metals from highways and dredged material, for pesticides from agricultural fields), and some which predict the fate of contaminants and exposure of biota are also becoming available (e.g., Burns 1985/*).

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Documentation of waterborne contaminants or possible sources that could contribute such contaminants.

Directness of Measure: Low.

28. Direct Alteration

Ranking: Unaltered wetlands are more likely to exhibit a notably great on-site diversity and/or abundance of fish and invertebrates than those that have been altered by tilling, filling, excavation, addition of inlets, or blockage of outlets.

Rationale: Tilling, filling, or excavating wetlands removes and buries aquatic vegetation, and thus reduces diversity/abundance of many types of fish and invertebrates. Adding an outlet where none previously existed or blocking an inlet alters the hydroperiod, decreases the available aquatic habitat, and thus can adversely affect aquatic organisms.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate. The effect depends on the specific type of alteration, proximity, time elapsed, construction practices used, and other factors.

Measure: Documentation of wetland tilling, filling, or excavation in the past 3 years, the addition of outlets, or blocking of inlets.

Directness of Measure: Moderate.

31. Water/Vegetation Proportions

Ranking: Wetlands that have moderate amounts of their total area covered by unvegetated surface water are more likely to exhibit great on-site diversity of fish and invertebrates.

Rationale: See Predictor 15.1.

Confidence in Ranking: Moderate.

Potential Importance to Function: High.

Measure: Determination of the portion of the wetland that is flooded less than 6.6 feet deep and what part of the total inundated area is dominated by vegetation. The categories of vegetation cover used in Volume II are arbitrary.

Directness of Measure: Moderate.

32. Hydroperiod (spatially dominant)

Ranking: The rankings for this predictor depend on the wetland system. For estuarine and marine wetlands, those that are not irregularly flooded are more likely to have a great on-site diversity of fish and invertebrates. Riverine wetlands that are seasonally or permanently flooded are more likely to have a great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Wetlands with at least some areas that are permanently or frequently flooded, or wetlands connected to permanent water, provide a greater amount of aquatic habitat for longer periods of time than disconnected wetlands with shorter hydroperiods, and thus often have higher fish and invertebrate diversity. Permanently inundated areas may provide refugia for aquatic life when other areas dry up. In contrast, aquatic habitat is available infrequently and unpredictably in irregularly flooded estuarine or marine wetlands, and physiochemical conditions are frequently extreme.

Although infrequently flooded riverine wetlands occasionally have high fish and invertebrate diversity (Brinson et al. 1981a/*fo, Pollard et al. 1983/LA:fo), their most productive parts are the parts flooded for the longest duration (but not permanently) (Cobb et al. 1984). Increasing invertebrate diversity can even be used to indicate increasing hydroperiod permanency in prairie potholes (Driver 1977/MAN:Pem). Low water levels during periods of spawning can have detrimental effects on fish populations (Lantz et al. 1965). Water level was the single most important factor in explaining the densities of brown shrimp in a Texas salt marsh (Zimmerman and Minello 1984/TX:Eem). In another salt marsh, pools more than 15 cm above mean high high water (MHHW) had lower richness than those less than 10 cm above MHHW (Barnby et al. 1985). See related Predictors 8, 23, 34, and 35.

Confidence in Ranking: Moderate to low. Nutrient availability, and consequently aquatic production, may be less in permanently flooded wetlands than in seasonally flooded ones (see Predictor 35).

Potential Importance to Function: High. However, in estuarine systems, salinity can be a more important determinant of species richness than the closely interrelated hydrologic regime (Barnes 1988).

Measure: Determination of the flooding regime of the largest percentage of the wetland.

Directness of Measure: High to moderate. Duration of connection to other water bodies, distance to refugia habitats, and type of substrate also influence recolonization and thus diversity.

33. Most Permanent Hydroperiod

Ranking: Wetlands most likely to exhibit great on-site diversity and/or abundance of fish and invertebrates with respect to this predictor include: estuarine wetlands that are not irregularly flooded; permanently flooded, intermittently exposed, and artificially flooded lacustrine/palustrine wetlands; and permanently flooded and intermittently exposed riverine wetlands.

Rationale: Wetlands with at least some areas that are permanently or frequently flooded provide a greater amount of aquatic habitat for longer periods of time than those with shorter hydroperiods, and therefore have higher fish and invertebrate diversity/abundance. Water level was the most important factor explaining brown shrimp densities in a Texas salt marsh (Zimmerman and Minello 1984/TX:Eem). In lacustrine/palustrine wetlands, low water levels during periods of spawning can have detrimental effects on fish populations (Lantz et al. 1965).

Potential Importance to Function: High.

Measure: Determination of the flooding regime of that portion of the wetland that is inundated or saturated most of the year.

Directness of Measure: Moderate.

34. Water Level Control

Ranking: Wetlands with drastic artificial water-level fluctuations are less likely to support a notably great on-site diversity of fish and invertebrates.

Rationale: Water-level control structures (especially those with deep-release outlets) can alter hydroperiods within and below the impounded area, reduce the outflow of detritus (Kondratieff and Simmons 1984, Voshell and Parker 1985/VA:R) with resultant decline of downstream aquatic insect diversity and biomass (Herlong and Mallin 1985, Perry and Sheldon 1986/MT:L,R), and thus depress the growth of insectivorous fishes downstream (Trotzky and Gregory 1974/ME:R, Rimmer 1985, Paragamian and Wiley 1987/IA:R). Aquatic invertebrate richness within impoundments can also be diminished (Sklar and Conner 1979/LA:fo). Large water-level fluctuations expose spawning areas, denude shoreline vegetative cover, and reduce aquatic invertebrate richness and abundance (Cushman 1985, Gislason 1985, Irving 1985, Holland 1987/MN:R, Erman and Ligon 1988/CA:R).

In reservoirs with shore slopes of less than 0.16, wetland plant communities were richest and most extensive where the water-level fluctuation in the year prior to sampling was less than 12 feet (Smith et al. 1987/UK:L). Benthic communities appear unaffected by maximum daily water-level changes of less than 1 foot (Smith et al. 1981/UK:L), but fluctuations of greater than 3 feet can affect diversity (Fisher and Lavoy 1972/MA:L). Reduction in an annual spring drawdown from between 33 and 40 feet to between 20 and 23 feet tripled the benthic invertebrate populations of a Missouri River reservoir (Benson and Hudson 1975/SD:L). Equations for quantifying fish response to water-level conditions are provided by Ploskey et al. (1984/US:L).

Confidence in Ranking: Moderate. Depends on proximity to dam, effectiveness of dam, location of outlet, and other factors. Intermediate levels of hydrologic disturbance (e.g., beaver dams) may increase aquatic diversity and abundance (Mackay and Waters 1986/MN:R, McDowell and Naiman 1986/QUE:R).

Potential Importance to Function: Moderate to high.

Measure: Documentation of artificial structures that influence wetland hydrology.

Directness of Measure: Low.

35. Flooding Extent and Duration

Ranking: Wetlands that experience seasonal flooding of large extent and duration are more likely to support a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Spring flooding provides important spawning, feeding, and nursery areas for many species, and generally reduces the number and magnitude of many limiting factors (e.g., low dissolved oxygen concentration) (Starret 1951/IA:R, Johnson 1963/*, Marzolf 1978/*fo, Pollard et al. 1983/LA:fo, Ross and Baker 1983/MS:R). The magnitude of the effect on fish biomass can be estimated for some water bodies using equations of Ploskey et al. (1984/US:L). Recolonization of newly inundated areas by macroinvertebrates can occur within weeks (Fisher et al. 1982a/NC:E), but may require 3 months (Williams and Hynes 1977/ONT:R, Kaster and Jacobi 1978/IL:R), and densities in areas of recently inundated vegetation can be very high (e.g., Murkin and Kadlec 1986/Man:Pem). The duration of inundation is a key determinant of invertebrate community structure in ephemeral parts of headwater streams (Delucchi 1988). Southeastern floodplain ponds with the most fish received more than 10 weeks of flooding per year (Cobb et al. 1984). See also Predictor 32.

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of the extent and duration of flooding in the wetland. Threshold levels are from US Army Corps of Engineers (1980) and are arbitrary.

Directness of Measure: Low.

40. Bottom Water Temperature

Ranking: Wetlands with cool-to-warm water temperatures (greater than 50° F) are more likely to support a notably great on-site diversity and/or abundance of fish and invertebrates in lacustrine/palustrine systems. Those with extreme water temperatures are less likely to be as diverse or productive. For riverine systems, seasonally flooded areas with summer temperatures greater than 69° F, and areas with stable water levels and summer temperatures less than 69° F, are more likely to support this function.

Rationale: Submersed macrophytes show increased growth from 10° C to about 28° C (McNabb 1976/*ab, Barko et al. 1984/ab). Extremely warm water temperatures are physiologically stressful both directly and to the degree to which they lower dissolved oxygen concentrations (Mills 1971/*, Wesche and Rechard 1980/*R). In contrast very cold temperatures, although less likely to be directly lethal, can limit the reproduction and growth of some species. Warmer mean temperatures and wider temperature fluctuations (within local limits) increase the abundance and diversity of riverine invertebrates (Vannote et al. 1980/*R).

Confidence in Ranking: High.

Potential Importance to Function: Moderate to high. Weekly maximum water temperature was a more statistically significant predictor of trout occurrence in southern Ontario streams than were gradient, width, depth, concentration of fine particulates, or discharge variables (Barton et al. 1985/ONT:R).

Measure: Determination of the average daily minimum summer water temperature at the deepest part of the wetland. Thresholds are approximate.

Directness of Measure: High (if measured regularly).

41. Velocity (spatially dominant)

Ranking: Riverine wetlands with low water velocities during peak annual flow are more likely to exhibit a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Fish and invertebrate diversity is highest at low or intermediate water velocities (Moyle and Cech 1982/*, Beckett et al. 1983/MS:R). Detritus and nutrients important to aquatic life accumulate in such low-velocity areas (McDowell and Naiman 1986/QUE:R, Huryn and Wallace 1987/NC:R). The abundance of such detritus is probably more important than its quality (Culp and Davies 1985/BC:R). Assuming water temperatures are not abnormal, most fish can maintain their position in currents with velocities (feet/second) up to about 5 to 10 times their body length. For coldwater riverine species, sustained velocities of greater than 8.2 ft/sec are nearly always impassable, while velocities in the 1.5 to 2.5 ft/sec range are generally preferred. Juvenile trout occupy habitat near the edges of streams where the velocity is zero (Scarnecchia and Bergersen 1987/CO:R).

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Determination of the velocity throughout most of the wetland during peak, annual flow.

Directness of Measure: Low.

45. Substrate Type (spatially dominant)

Ranking: Estuarine and marine wetlands with bedrock or rubble substrates, and palustrine, lacustrine, and riverine wetlands with sand substrates, are less likely than other wetland types to support a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Organic sediments such as peat and muck, which are usually more prevalent in vegetated aquatic areas than in nearby unvegetated bottoms, generally have greater densities of fish and aquatic invertebrates (Osenga and Coull 1983, Cyr and Downing 1988) (see also Predictor 12 for related references). Recolonization following exposure of the substrate during drawdown also occurs faster in organic sediments (Kaster and Jacobi 1978/IL:R). Sand generally supports low densities of macroinvertebrates (Whitlatch 1980, Beckett et al. 1983/MS:R, Anderson and Day 1986/IL:R), in part because of its physical instability. Recolonization in disturbed sandy sediments is slower than in organic sediment (Driver 1977/MAN:Pem) or on moss-covered rock substrates (Gurtz and Wallace 1984/NC:R).

Substrate (sediment) preferences of fish are summarized quantitatively and species-specifically in Bovee (1978/*R) and Wesche and Rechard (1980/*R). Precise data for spawning substrate preferences of some salmonids are given by Platts et al. (1979), Shirazi and Seim (1979/*R), Peterson (1981/*E), Hale et al. (1985/*), and Raleigh and Nelson (1985/*).

Confidence in Ranking: Moderate. Abundance and biomass of aquatic invertebrates is occasionally greater in mud or organic sediments than in either cobble-gravel (Marzolf 1978/*fo, McDowell and Naiman 1986/QUE:R, Huryn and Wallace 1987/NC:R) or aquatic beds (Anderson and Day 1986/IL:R), but taxa richness in mud is generally less than in aquatic beds (Driver 1977/MAN:Pem). In estuarine systems, macrobenthic density and taxa richness can be greater in sand than in muddy sand, which has a higher organic content and greater heavy metal content (Franz and Harris 1988/NY:E). The relationship is nonlinear, with a total organic content in the range 0.7 to 1.0 percent being adequate in urban estuarine sediments for supporting a diversity of macrobenthic invertebrates, and greater or lesser amounts being stressful (Franz and Harris 1988/NY:E).

Potential Importance to Function: Moderate. Although substrate type is important for invertebrates, for riverine fish it is much less important than current velocity and water depth, except for spawning (Hynes 1970/*R). In estuarine systems, salinity can be a more important determinant of species richness than can substrate (sediment) type (Barnes 1988).

Measure: Determination of the dominant substrate type in the wetland.

Directness of Measure: Low.

46. Physical Habitat Interspersion

Ranking: Wetlands that contain a mosaic of substrate types, velocities, and depths are more likely to have greater on-site diversity and/or abundance of fish and invertebrates.

Rationale: See Predictor 49.

Confidence in Ranking: High.

Potential Importance to Function: Moderate.

Measure: Determination of the number and distribution of substrate types, velocity categories, and depth classes.

Directness of Measure: Moderate.

47. pH

Ranking: Wetlands in which the pH is circumneutral are more likely to support a greater on-site diversity and/or abundance of fish and invertebrates.

Rationale: Values between pH 5.6 and 8.6 are best for aquatic production (Cook and Powers 1958/NY:Pem, Darnell et al. 1976*, Fryer 1980) and richness (Friday 1987/UK:P,L). Invertebrate richness can be positively correlated with pH 3 (Driver 1977/MAN:Pem). Acidification of wetland habitats can be detrimental to fish populations because toxic metals, particularly inorganic aluminum, are mobilized by low pH (Haines 1981/*, Baker and Schofield 1982, Magnuson et al. 1984/*, Wiener 1987).

Confidence in Ranking: Moderate to high. Acidic conditions may eliminate predation by fish and result in richer communities of algae, aquatic invertebrates, and other consumers (e.g., birds), since fish are generally more acid-sensitive than other fauna (Bendell and McNicol 1987, Wiener 1987).

Potential Importance to Function: Moderate.

Measure: Determination of the pH of water in the wetland.

Directness of Measure: Moderate.

48. Salinity and Conductivity

Ranking: Lacustrine/palustrine wetlands that have salinities less than 5 ppt and estuarine/marine wetlands with salinities less than 40 ppt are more likely to have a great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Lacustrine, palustrine, and riverine wetlands are dominated by species that generally cannot survive in salinities greater than 5 ppt (Remane 1971/*E, Silberhorn et al. 1974/VA:Eem). Estuarine and marine wetlands with hypersaline waters generally have lower densities and numbers of species than similar wetlands with salinities less than 40 ppt (Copeland and Nixon 1974/*E). Productivity of vascular plants is also generally lower in hypersaline wetlands than in other wetland types.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate to high. Salinity can be a major predictor of aquatic diversity in land-locked, brackish, coastal lagoons (Barnes 1988).

Measure: Determination of the salinity or conductivity of the wetland.

Directness of Measure: High.

49. Aquatic Habitat Features

Ranking: Riverine wetlands containing relatively equal proportions of pools (or backwater sloughs) and riffles are more likely to support a notably great on-site diversity and/or abundance of fish and invertebrates. Straight, featureless channels and embayments are less likely to have diverse aquatic faunas.

Rationale: Because many aquatic organisms show preferences for specific substrate types, water velocities, and depths, wetlands with a high diversity of these characteristics have higher fish and invertebrate species richness and abundance (Day and Pearcy 1968/OR:M, Gorman and Karr 1978/IN:R, Moyle and Cech 1982/*, MacDonald et al. 1987/BC:Eem, Scarnecchia and Bergersen 1987/CO:R, Schlosser 1987/IL:R).

Habitat complexity, and thus aquatic diversity/abundance, in riverine wetlands is particularly likely to be associated with pools, which provide nursery areas for young fish (Bustard and Narver 1975/BC:R, Levy and Northcote 1981/BC:Eem; Peterson 1982a/AK:M,b; Heifetz et al. 1986/AK:R). Deeper pools are also used by large adult fishes as refuges during low-flow periods. Pools sometimes support more aquatic invertebrates than do riffles (McDowell and Naiman 1986/QUE:R), although their availability to fishes may be greater in riffles (Hynes 1970/*R, Power and Matthews 1983/OK:R, Schlosser 1987/IL:R).

In riverine wetlands, habitat complexity and increased secondary production are also associated with increased presence of undercut banks and large woody debris (Marzolf 1978/*fo, Smock et al. 1985/SC:R, Harmon et al. 1986/*fo).

Confidence in Ranking: High.

Potential Importance to Function: Moderate.

Measure: Determination of whether a riverine wetland has an adequate mixture of riffles and pools, backwaters, or similar slow-water areas. The pool-riffle ratio thresholds are based on best habitat for coldwater fish given by Nunnally and Keller (1979/*R) and are probably not universally applicable.

Directness of Measure: Moderate.

52. Freshwater Invertebrate Density

Ranking: Wetlands that have high aquatic invertebrate densities are more likely to support a notably great on-site diversity and/or abundance of freshwater fish species.

Rationale: Freshwater fish production is generally, but not always, correlated with aquatic invertebrate production (Hynes 1970/*R, Binns and Eiserman 1979/WY:R).

Confidence in Ranking: Moderate. Terrestrial insect inputs may be at least as great as aquatic invertebrate production in lower order riverine wetlands. Factors other than food are frequently more prevalent as limitations.

Potential Importance to Function: *High.*

Measure: Determination of the wetland's benthic and epibenthic macroinvertebrate densities during the growing season.

Directness of Measure: High. Thresholds are based on a limited review of thc literature and are mainly applicable to benthic communities of riverine systems. Maximum macroinvertebrate biomass on wetland plants themselves (epiphytic fauna) may exceed 9,714 kg per kilogram of plant (wet weight) (Schramm et al. 1987).

53. Tidal Flat Invertebrate Density/Biomass

Ranking: Estuarine/marine/riverine wetlands that have high tidal flat invertebrate densities are more likely to support a great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Numerous fish and invertebrate species consume tidal flat invertebrates. High productivity of such fauna may depend on high densities of tidal flat invertebrates (Tyler 1971, Peterson and Peterson 1979/*NC:E, Whitlatch 1982/*E).

Confidence in Ranking: Moderate.

Potential Importance to Function: High.

Measure: Determination of the density and biomass of macroinvertebrates in the tidal flats of the wetland.

Directness of Measure: High. Thresholds are based on the review by Diaz et al. (1982).

55. Suspended Solids

Ranking: Wetlands that receive runoff or surface waters with low levels of suspended solids (especially inorganic) (usually less than 80 mg/l and never exceeding 200 mg/l) are more likely to support a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: High levels of suspended solids can have detrimental impacts on aquatic life, both direct (e.g., clogged gills) and indirect (e.g., effects on spawning area suitability, fish movements, and the food chain, which result from reduction in plant productivity due to blocking of solar radiation) (Farnworth et al. 1979/*). Although direct tolerance to turbidity is species specific, estuarine and marine fishes as a whole tend to be more sediment-tolerant than coldwater species, and adult stages more tolerant than egg or larval stages.

Data on tolerances of coldwater species are partly catalogued by Lloyd (1987/*). Most coldwater species grow best at suspended solids concentrations less than 80 mg/l, but concentrations less than 25 mg/l (about 5 NTU) have been shown to be required for optimal aquatic invertebrate production (Lloyd et al. 1987/AK:R). Suspended solids concentrations of as little as 35 mg/l and 10 NTU have resulted in reduced fish feeding (Bachman 1984, Berg and Northcote 1985). Concentrations of 120 mg/l (or 25 NTU) reduce growth of some coldwater fishes (Sigler et al. 1984).

For warmwater fish species associated with turbid waters or bottom habitats, TSS concentrations of 1,000 mg/l may be tolerated by some species (Peddicord and McFarland 1978/*). However, concentrations exceeding 500 mg/l may inhibit hatching success of some warmwater species, and relatively impoverished fish and invertebrate communities were associated with turbidities exceeding about 50 NTU in Southeast floodplain ponds (Cobb et al. 1984). In a depositional environment, invertebrate communities may be moderately impacted by increases of less than 40 mg/l above background levels, and severely decimated (60-percent reduction) by increases of over 120 mg/l above background (Gammon 1970/IN:R). Turbidity can further exacerbate fish mortality by raising the water temperature and thus depleting dissolved oxygen (Reed et al. 1983/NC).

Confidence in Ranking: Moderate to high. Nutrients associated with sediment runoff can sometimes be beneficial, and moderate concentrations of suspended sediment may provide cover for some fish (McCrimmon 1954;

Gammon 1970/IN:R; Cyrus and Blaber 1987a,b; Minello et al. 1987/KY:R). Adverse effects on aquatic invertebrates are not inevitable (e.g., Rabeni and Minshall 1977).

Potential Importance to Function: Moderate. In some estuaries (e.g., coastal Louisiana), progressive loss of wetland habitat is resulting from **insufficient** sediment.

Measure: Determination of the suspended solids level or Secchi disc reading of runoff or surface water entering the wetland. Thresholds are from data summarized by Lloyd (1987/*) and Wesche and Rechard (1980/*R), as well as water quality codes of several states.

Directness of Measure: Moderate. Suspended solids, while correlated with turbidity, are not identical. Secchi disc readings are somewhat subjective and may be inaccurate in highly stained waters. Data from measurements taken over many seasons and hydrologic events are preferred.

56. Dissolved Solids or Alkalinity

Ranking: Wetlands with low (less than 20 mg/l $CaCO_3$) alkalinity are less likely to support this function. Also, wetlands with either low (less than 7) or high (greater than 35) morphedaphic indices (total dissolved solids/mean depth) are more likely to support a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Moderately elevated alkalinity levels usually are associated with increased fish standing crop (Ryder 1965, Jenkins and Morais 1971/L, Scarnecchia and Bergersen 1987/CO:R).

Confidence in Ranking: High. This relationship is less certain in coldwater riverine environments (Hynes 1970/*R), at alkalinity levels greater than 40 mg/l (Moyle 1956/), and for wetlands with heavy angling pressure or large inputs of domestic waste.

Potential Importance to Function: Moderate. Sportfish harvest in Midwestern lakes and reservoirs was more strongly correlated with chlorophyll-a concentrations over the range 0.008 to 0.060 mg/l than with alkalinity or the morphedaphic index (Jones and Hoyer 1982).

Measure: Determination of the alkalinity of wetland water and the morphedaphic index of the wetland. Threshold levels for the morphedaphic index are from the empirical study of Jenkins and Morais (1971/L) and represent maximum yields of reservoir fish. The 20-mg/l alkalinity threshold is similar to that given by Fried (1974/*NY), Larson (1976/*MA), and many state water quality codes. This threshold relates mainly to freshwater primary productivity.

Directness of Measure: Moderate.

57. Eutrophic Conditions

Ranking: Oligotrophic wetlands are less likely to support a notably great on-site diversity and/or abundance of fish and invertebrates.

Rationale: Fish biomass has been significantly correlated with nitrate concentrations up to approximately 2.0 mg/l (Binns and Eiserman 1979/WY:R). Yield may also be correlated with phosphorus (Jones and Hoyer 1982). Wetland invertebrate production responds positively to moderate nutrient enrichment, at least in oligotrophic or low-alkalinity systems (Cyr and Downing 1988, Welch et al. 1988). Particulate organic matter concentrations of 300 to 600 mg/m^3 in the outfalls of lakes tended to produce the greatest invertebrate species richness in 13 Montana streams (Perry and Sheldon 1986/MT:L,R).

Aquatic macrophyte richness also can be greater (at least in initially acidic wetlands) if wetlands are exposed to nonpoint-source nitrogen enrichment (Ehrenfeld 1983/NJ:P). In the same situation, more plant species were endemic to enriched sites than to pristine sites (Morgan and Philip 1986). However, if existing alkalinity levels are at least moderate, enrichment has relatively little effect on aquatic bed plants (Madsen and Adams 1988/WI:R). Moderately eutrophic conditions can also enhance protozoan colonization of wetlands (Henbry and Cairns 1984), amphibian richness (Beebee 1987/UK:Pem), bird richness (Harris and Vickers 1984/FL:P), and can reduce the effects of chemical toxicity (Fairchild et al. 1984).

Confidence in Ranking: Moderate. Greatest plant richness in some riverine wetlands occurred at sites with less phosphorus enrichment, low plant densities, little plant litter accumulation, and intermittently exposed hydroperiod (Day et al. 1988). Eutrophication can also shorten food chains and reduce top predators in lakes (Odum 1985/*).

Potential Importance to Function: Moderate.

Measure: Determination of water nutrient levels or their indicators in relation to threshold levels. Thresholds are from Vollenweider (1976), Binns and Eiserman (1979/WY:R), and Taylor et al. (1980).

Directness of Measure: Low.

61. Dissolved Oxygen

Ranking: Wetlands where dissolved oxygen concentrations are greater than 4 mg/l and 60 percent saturation are more likely to support a notably great on-site diversity and/or abundance of fish and invertebrates than those with lower DO concentrations.

Rationale: Adequate DO is physiologically essential to all fishes and invertebrates (Hynes 1970/*R, Moyle and Cech 1982/*) and often naturally limits the richness of invertebrate (Ziser 1978/LA:fo) and fish (Tonn and Magnuson 1982/WI:L,P) communities in wetlands.

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of DO concentration of water in wetland.

Directness of Measure: Moderate.

3.9 Wildlife Diversity/Abundance

1. Climate (predictor for breeding, migration, and wintering)

Ranking: Wetlands in areas where evaporation exceeds precipitation are most likely to support a notably great on-site diversity and/or abundance of wetland-dependent birds. Wetlands not prone to icing are more likely to support a notably great on-site diversity and/or abundance of wetland-dependent birds.

Rationale: Wetlands located in precipitation deficit regions are likely to provide habitats that are extremely limited in availability and distribution. Wetland water levels in such regions are dynamic; this condition increases the mobilization and cycling of nutrients, which in turn results in high biotic densities and diversity. Climate is also used as a classification variable in the Habitat Suitability evaluations of Volume II, inasmuch as wetlands prone to winter icing receive less use (e.g., Morton et al. 1987/VA:E,P).

Confidence in Ranking: Low.

Potential Importance to Function: Moderate.

Measure: Determination of whether the wetland is located in a precipitation-deficit region or whether on-site evaporation exceeds precipitation.

Directness of Measure: Low.

2. Acreage (predictor for breeding, migration, and wintering)

Ranking: Larger wetlands (or those directly connected to large water bodies or tracts of suitable undeveloped habitat) are more likely to support a notable on-site diversity and/or abundance of wetland-dependent birds than are small wetlands.

Rationale: Larger wetlands usually provide a greater variety of physical habitat and food resources than do small wetlands. The need for area differs greatly by species. Increasing the size of small areas usually has a greater impact on species richness than increasing the size of large areas (MacArthur and Wilson 1967, Harris 1984/*fo). Species richness in large areas may be influenced more by internal habitat heterogeneity and isolation from similar habitat (Askins et al. 1987). Small and/or narrow wetlands, because they are more exposed to the effects of adjoining habitat, are more vulnerable to effects of habitat degradation caused by sedimentation, trampling, drought, and hydroperiod alteration. Their fauna is also more susceptible to loss associated with predation (Bostrom and Nilsson 1983/EU:P, Andren et al. 1985/EU:fo, Wilcove 1985) and human disturbance. However, small wetlands may have great regional significance in areas lacking many wetlands.

For nonforested wetlands, breeding waterbird density generally increases with area size, at least in prairie potholes (Patterson 1976) and peat bogs (Bostrom and Nilsson 1983/EU:P, Lahti and Ranta 1985/fo). Larger wetlands tend to have proportionally richer nesting avifauna (Brown and Dinsmore 1988/IA:Pem).

Emergent wetlands less than 3 acres in size may not provide all the habitat needs for some breeding dabbling ducks (Dzubin 1969). Some waterbird species need large areas for molting and roosting (Hoy 1987/TX:L). To support the breeding of a diverse (24 species) regional avifauna, a cumulative wetland acreage of 200 to 900 acres is probably needed (Brown and Dinsmore 1986/IA:Pem).

For forested wetlands, species richness and density of songbirds, particularly of certain characteristic "forest interior" species, increase with the size of the forested tract (Galli et al. 1976/NJ:fo, Robbins 1979/MD:fo, Ambuel and Temple 1983/WI:fo, Blake and Karr 1984/IL:fo, Harris 1984/*fo, Lynch and Whigham 1984/MD:fo, Askins et al. 1987), even when habitat complexity and richness do not seem to increase with area (Moller 1987/EU:fo, Blake and Karr 1987/IL:fo).

Confidence in Ranking: Low to moderate. Large fringe wetlands are more likely to be exposed to severe winds and waves, which stress wildlife and reduce abundance of some foods (see Predictor 18). Large fringe wetlands often attract recreationists, and the resultant disturbance may reduce wildlife diversity or use (see Predictor 30). An equivalent total acreage of smaller wetlands may provide a greater cumulative benefit to wetland wildlife, because variety in habitat structure and productivity is generally greater among a set of many small habitats than in a single large one (Dzubin 1969, Wilson and Willis 1975, Ruwaldt et al. 1979/SD:Pem, Weller 1979/*, Simberloff and Abele 1982/FL, Kantrud and Stewart 1984/ND:Pem, Brown and Dinsmore 1986/IA:Pem, Lahti and Ranta 1986/fo). Relatively high densities of nesting waterfowl occasionally occur in fringe wetlands as small as

0.6 to 1.2 acres, and brood densities may be highest in some regions in wetlands of 1.3 to 3.7 acres (Hudson 1983/MT:Pem).

Potential Importance to Function: Moderate. Other factors such as habitat richness and type (Boecklen 1986, Martin 1986/AZ:fo, May 1986/*, Freemark and Merriam 1986/Ont:fo), the degree of contrast with the richness and productivity of adjoining uplands (Angelstam 1986/EU:fo), and regional wetland density, i.e. isolation (Brown and Dinsmore 1986/IA:Pem) may be stronger predictors of faunal diversity, at least in regions with abundant wetlands.

Measure: Determination of wetland acreage. The 5-acre minimum threshold is arbitrary.

Directness of Measure: Moderate.

3. Complex, Cluster, Oasis (predictor for breeding, migration, and wintering)

Ranking: A wetland that is the only wetland within a wide area (an oasis), or is part of a dense regional cluster or complex (or is singularly very large), is more likely to exhibit notable on-site diversity and/or abundance of wetland-dependent birds.

Rationale: Oasis wetlands draw in wildlife from large areas, and thus assume disproportionate importance (Weller and Fredrickson 1974/IA:Pem). Riparian systems in the West and forested wetlands in the Southeast and Midwest, for example, often provide the only structurally complex habitat in regions dominated by open land or land cleared for agriculture; they may have much higher diversities and/or densities of wildlife than do upland systems (Hooper 1967/VA:fo, Austin 1970, Bottorff 1974/CO:fo, Gaines 1974/CA:fo, Blem and Blem 1975/IL:fo, Lewke 1975/WA:fo, Stevens et al. 1977/*, Dickson 1978/*, Hair et al. 1978, Knopf 1986/CO). Riparian areas also serve as important travel corridors for many wildlife species. Oases may also serve as "relict" areas for rare species, or as vital stop-over habitats for migrating birds.

Complex/cluster wetlands are important because they provide diverse habitats (Dzubin 1969, Ruwaldt et al. 1979/SD:Pem, Weller 1979/*, Kantrud and Stewart 1984/ND:Pem, Brown and Dinsmore 1986/IA:Pem). For example, in the Prairie Pothole Region, areas with 5 basins (wetlands) per square kilometer were used by gadwalls, whereas those with only 2.8 wetlands/ km^2 were not (Leitch and Kaminski 1985/SAS:Pem). The importance of singularly large wetlands, which may qualify as clusters in Volume II, is discussed above under Predictor 2.

Confidence in Ranking: High. Preliminary analyses of coastal nesting bird colonies, however, failed to show a statistically significant relationship between colony size and surrounding wetland area (Erwin et al. 1987/US:E).

Potential Importance to Function: Moderate.

Measure: Determination of the proximity of the wetland to other wetlands, and whether the wetland can be classified as an oasis or part of a cluster. The cluster and oasis size thresholds, which are equal to 20 percent of the estimated mean statewide density of wetlands (acres/square mile), are arbitrary.

Directness of Measure: Low. Criteria for cluster and oasis are best derived through analysis of the home range and energetics of the particular species present.

4. Location and Size

4.1 Proximity to Tidal Waters, the Great Lakes, or a Major River (predictor for migration and wintering)

Ranking: Wetlands located close to tidal waters, the Great Lakes, or a major river are more likely to support a notable on-site diversity of wetland-dependent birds than are those not located near such a body of water.

Rationale: Migratory waterfowl travel major river courses (Craighead and Craighead 1949/WY:Rem, Golet and Larson 1974/*MA, Bellrose 1976/*, Raveling 1977/MAN), and tidally influenced wetlands are of major importance to wintering waterfowl (Weller 1975/*, Diefenbach et al. 1988). The Great Lakes are particularly important as staging areas and migration corridors for waterfowl (Bellrose 1976/*). Thus, wetlands in close proximity to these areas may be important during migration and wintering.

Confidence in Ranking: Moderate.

Importance to Function: Moderate.

Measure: Determination of the proximity of the Great Lakes, tidal waters, or major rivers. The 5-mile threshold is arbitrary.

Directness of Measure: Low.

4.2 Watershed Size (predictor for breeding)

Ranking: Wetlands with larger watersheds are more likely to support notably great on-site diversity and/or abundance of wetland-dependent birds compared to those with smaller watersheds.

Rationale: Wetlands with larger watersheds are more likely to persist than those with smaller watersheds. Also, larger watersheds provide a greater source area for nutrients to the wetland, perhaps making the wetland more productive.

Confidence in Ranking: Low. The dynamic water levels that characterize headwater wetlands (i.e., isolated wetlands in small watersheds) usually support greater nutrient cycling, invertebrate food production and, thus, wildlife than do the more stable water levels of isolated wetlands in large watersheds.

Potential Importance to Function: Moderate.

Measure: Determination of the wetland's watershed size. The 1-square mile threshold is arbitrary.

Directness of Measure: Low.

7. Gradient (predictor for breeding)

Ranking: Wetlands with lower gradients are more likely to exhibit a notable on-site diversity and/or abundance of wetland-dependent birds than are wetlands with steep gradients.

Rationale: Water velocity usually increases with increasing gradient. Extreme water velocities may keep sediments in suspension, thus inhibiting wetland establishment and persistence and reducing the availability of food organisms. Extreme velocities are also avoided by most wetland wildlife.

Confidence in Ranking: Low.

Potential Importance to Function: Low to moderate.

Measure: Determination of gradients in relation to threshold levels necessary to create depositional velocity conditions.

Directness of Measure: Low.

8. Inlets, Outlets (predictor for breeding, migration, and wintering)

Ranking: Wetlands with permanent outlets are more likely to support a notably great on-site diversity and/or abundance of wetland-dependent birds than are wetlands without permanent outlets.

Rationale: Wetlands without outlets tend to concentrate toxicants when toxicants are present. Accumulation of peat may limit aquatic food chain productivity and diversity, at least in cooler climates. Preferences of particular species for connected versus isolated wetlands are documented by Craighead and Craighead (1949/WY:Rem), Coulter and Miller (1968/ME,VT), Kitchen and Hunt (1969/WI), Reed and Moisan (1971/QUE:Eem), Stewart and Kantrud (1973/ND:Pem), Dawe and Davis (1975/BC), Raveling (1977/MAN), and Hepp and Hair (1979).

Confidence in Ranking: Low. In some regions (e.g., Prairie Pothole) isolated wetlands are exceptionally productive and critical to wildlife.

Potential Importance to Function: Low to moderate.

Measure: Determination of presence or absence of permanent outlets.

Directness of Measure: Low.

10. Wetland System (predictor for breeding)

Ranking: Lacustrine, palustrine, or riverine wetlands are more likely to support a notably great on-site diversity and/or abundance of wetland-dependent birds during the breeding season than are estuarine systems. Marine wetlands are least likely to support a notable diversity and/or abundance of breeding wetland-dependent birds.

Rationale: Lacustrine, palustrine, and riverine systems provide the habitat richness necessary to support a diversity of wetland-dependent bird species. Monotypic tidal salt marshes (estuarine intertidal emergent wetlands) typically support fewer species as nesters than do most freshwater wetlands. Marine wetlands also have fewer nesting bird species, partly because physical cover is lacking. Estuarine wooded (e.g., mangrove) wetlands can support substantial numbers of breeding colonial waterbirds including herons, egrets, ibises, and roseate spoonbills (Soots and Landin 1978/*). Unconsolidated shores in estuarine systems also can support substantial colonies of breeding waterbirds such as pelicans, black skimmers, and several tern species (Soots and Landin 1978/*). Thus, even though the diversity of breeding birds in estuarine systems may not be great, some types of estuarine wetlands support substantial numbers of a few colonial nesting species. For migratory waterbirds, estuarine systems are extremely important.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Determination of wetland system (and subsystem if tidal riverine).

Directness of Measure: Low. Salinity (see Predictor 48) is a more direct predictor.

12. Vegetation Class/Subclass (Primary) (predictor for migration and wintering)

Ranking: Wetlands dominated by forested or scrub-shrub vegetation are more likely to support a notable on-site diversity and/or abundance of migrating and wintering wetland-dependent birds.

Rationale: Forested and scrub-shrub vegetation generally provide more habitat structure through vertical layering and increased patchiness resulting from horizontal overlap of layers (Roth 1976). As a result, these areas can support a wider species diversity than vegetation forms that are less complex. See Predictor 17.

The habitat suitability evaluation for individual waterfowl groups (Table 2) is supported, in the case of this predictor, by the following:

Nesting/Summering:
Group 1: Krapu 1974/ND:Pem; Stewart and Kantrud 1974/ND:Pem; Bellrose 1976,1979/*; Kantrud and Stewart 1977/ND:Pem; Peterson and Low 1977/UT:Pem; Swanson 1977; Dwyer et al. 1979/ND; Flake 1979/*, Ruwaldt et al. 1979/SD:Pem, Mack and Flake 1980/SD:Pem.
Group 2: Hanson et al. 1949; Mendall 1949/ME; Parnell and Quay 1962/NC:em; Coulter and Miller 1968/ME,VT; Prince 1968/NB; Cowardin 1969/NY:fo; Erskine 1971/NS; Reed and Moisan 1971/QUE:Eem; Renouf 1972/NB; Gilmer et al. 1975/MN:em; Reed 1975/QUE:tem; Bellrose 1976,1979/*; Fefer 1977/ME:P; Landers et al. 1977/SC:FO; Flood et al. 1979/*; Hepp and Hair 1979; Reinecke and Owen 1980/ME:em.
Group 3: Kitchen and Hunt 1969/WI, Erskine 1971/NS, Renouf 1972/NB, Bellrose 1976/*.
Group 4: Hanson et al. 1949, Mendall 1958/VT,NH,ME, Prince 1968/NB, Renouf 1972/NB, Bellrose 1976/*, Fefer 1977/ME:P.
Groups 5 and 6: Bergman 1973/Man:Lab, Thompson 1973, Stewart and Kantrud 1974/ND:Pem, Bellrose 1976/*, Kantrud and Stewart 1977/ND:Pem, Sugden 1978.
Group 7: Craighead and Craighead 1949/WY:Rem, Dawe and Davis 1975/BC, Bellrose 1976/*, Raveling 1977/MAN.
Migration/Wintering
Group 1: Allan 1956/Eem, Cronan and Halla 1968/RI, Cowardin 1969/NY:fo, Palmisano 1973/*Eem, Chabreck et al. 1975/LA:Tem, Bellrose 1976/*, Chabreck 1979/*b, Joyner 1980/ONT:Pem, Gordon et al. 1987/SC:E,P.
Group 2: Mendall 1949/ME; Hartman 1960/ME:tem; Palmisano 1973/*Eem; Bellrose 1976/*; Landers et al. 1976, 1977/SC:Eem; Chabreck 1979/*b; Odum et al. 1979/*Eem; Allen 1980/TX; Gordon et al. 1987/SC:E,P.

Confidence in Ranking: Moderate to high.

Potential Importance to Function: High.

Measure: Determination of vegetation class and subclass of the wetland.

Directness of Measure: Moderate.

14. Islands (predictor for breeding)

Ranking: Wetlands on or associated with islands are more likely to support a notably great on-site diversity and/or abundance of breeding wetland-dependent bird species.

Rationale: Islands are important to nesting waterbirds because of the following characteristics: relative freedom from predation; greater capacity for territorial occupancy because of the increased shoreline available; and proximity of water, food, loafing, and nesting cover (Hammond and Mann 1956:345/MA:P). The importance of islands to particular species has been documented by Schreiber (1977,1979), Chaney et al. (1978), Buckley and Buckley (1980/*), Briggs et al. (1981/CA:M,E), Duebbert et al. (1983/ND,SD,MT:P), Hines and Mitchell (1983/SAS:Pem), and others.

Confidence in Ranking: High.

Potential Importance to Function: Moderate. Effect depends also on the size of the island and distance from mainland.

Table 2
Harvested Waterfowl Species Groups for Which Breeding, Migration, and Wintering Habitat Suitability Can Be Evaluated by WET

Group	Species Description
1	Prairie dabblers
2	Black duck
3	Wood duck
4	Common/red-breasted mergansers
5	Hooded merganser
6	Canvasback, redhead, ruddy duck
7	Ring-necked duck
8	Scaup (greater and lesser)
9	Common goldeneye
10	Bufflehead
11	Fulvous and black-bellied whistling ducks
12	Canada, white-fronted, snow and Ross' geese
13	Tundra swan
14	Brant

Measure: Determination of whether the wetland is located on or associated with an island. Size and distance thresholds are arbitrary.

Directness of Measure: Low.

15. Vegetation/Water Interspersion (predictor for breeding, migration, wintering)

Ranking: Wetlands with good vegetation-water interspersion are more likely to support a notably great on-site diversity and/or abundance of wetland-dependent birds.

Rationale: High vegetation-water interspersion is important because of the increased variety of vegetation types and cover conditions that results from such interspersion. Contact zones between open water and vegetation provide ready protection from wind, waves, and predators, and may provide natural territorial boundaries (Golet and Larson 1974/*MA) and/or isolating cover for breeding waterfowl (Murkin et al. 1982/MAN). Water-vegetation transition zones also provide habitat elements for both open-water species and species inhabiting adjacent vegetation (Weller and Spatcher 1965/IAM:Pem, Willard 1977). In addition, these transition zones are inhabited by species that are adapted specifically to the edge environment (e.g., yellow-headed blackbirds, gallinules, American coots, least bitterns, ruddy ducks, and redheads). See also Predictor 31.

Confidence in Ranking: Moderate to high. Ringleman and Longcore (1982/ME:P) and Rumble and Flake (1983) found no correlation between duck use and edge indices for the range of values they considered.

Potential Importance to Function: High.

Measure: Determination of relative vegetation-water interspersion.

Directness of Measure: Moderate.

16. Vegetation Class Interspersion (predictor for breeding, migration, and wintering)

Ranking: Wetlands with well interspersed vegetation classes are more likely to support a notable on-site diversity and/or abundance of wetland-dependent birds.

Rationale: Most species require several different cover types in one area to meet their requirements for food, shelter, nesting, loafing, and protection from predators (e.g., Dzubin 1969, Dwyer et al. 1979/ND, Ruwaldt et al. 1979/SD:Pem). The less energy expended in moving from one cover type to another in order to meet life requisites, the more suitable the area (Leopold 1933/*). As discussed for Predictor 15 (Vegetation-Water Interspersion), increased interspersion also results in an increased amount of edge habitat, which is important to species diversity (Harris et al. 1983/WI:P).

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of the horizontal pattern of vegetation class interspersion.

Directness of Measure: High.

17. Plant Form Richness (predictor for breeding, migration, and wintering)

Ranking: Wetlands with numerous well interspersed vegetation forms are more likely to support a notable on-site diversity and/or abundance of wetland-dependent birds.

Rationale: The presence of several vegetative zones or vertical strata encourages breeding bird use of semipermanent and seasonal wetlands (Kantrud and Stewart 1984/ND:Pem). Marshes with complex plant zonation provide several layers of vegetation, which in turn provide an increased number of available niches for breeding birds (Coulter and Miller 1968/ME,VT, Krapu 1974/ND:Pem, Landers et al. 1977/SC:FO, Swanson and Meyer 1977/ND, Weller 1978/*, Dwyer et al. 1979/ND, Flake 1979/*, Flood et al. 1979/*, Hepp and Hair 1979, Ruwaldt et al. 1979/SD:Pem). Increased patchiness helps explain why scrub-shrub areas have more species than some forested areas, even though scrub-shrub habitat has fewer vertical layers than most forested habitat (Roth 1976).

In forested areas, bird use is strongly tied to the diversity of vegetation life forms (Swift et al. 1984) and tree species (Tramer and Suhrweir 1975). Bird species diversity increases as the number and density of foliage layers increase (MacArthur and MacArthur 1961/fo, MacArthur et al. 1964/fo, Karr and Roth 1971/IL:R, Willson 1974, Roth 1976). Adding a tree layer to an area greatly increases breeding bird richness (Willson 1974). See also Predictor 12 (Vegetation class/subclass) and Predictor 16 (Vegetation Class Interspersion).

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of the number of vegetation classes and subclasses found in the wetland. All thresholds are arbitrary.

Directness of Measure: Moderate to high.

18. Shape of Upland/Wetland Edge (predictor for breeding, migration, and wintering)

Ranking: Wetlands in which the wetland-upland edge is irregular are more likely to support a notably great on-site diversity and/or abundance of wetland-dependent birds.

Rationale: Wetlands having a sinuous or irregular shape are likelier to have a greater interspersion of cover types and more edge. For ponds of equal area, higher brood densities have been observed on more irregularly shaped ponds (Hudson 1983/MT:Pem). A size of 5 acres provides the best ratio of shoreline length to surface area (Millar 1971/P). Sinuous wetlands also provide greater resistance to water flow, resulting in lower water velocities, which are generally preferred by waterfowl (especially dabbling ducks and geese). See also Predictors 15 and 16.

Confidence in Ranking: Moderate.

Potential Importance to Function: Low.

Measure: Determination of whether the wetland-upland edge is irregular.

Directness of Measure: Low.

19. Fetch/Exposure (predictor for breeding and wintering)

Ranking: Sheltered wetlands are more likely to support a notable on-site diversity and/or abundance of breeding and wintering wetland-dependent birds.

Rationale: Sheltered areas, lacking wave scour, may host a greater abundance of food and cover, making these areas attractive to waterfowl, wading birds, and aerially foraging species. Also, particularly in winter, sheltered areas may reduce metabolic stresses associated with inclement weather. For example, during the day, open water on a reservoir was used proportionately less by all duck species except migrating redheads; however, open water was used by ducks for roosting (Hoy 1987/TX:L).

Confidence in Ranking: Low to moderate. In areas experiencing winter freezing, exposed areas may remain open longer, making them attractive to waterfowl. Also, large open areas can be more important than vegetated wetlands for waterfowl roosting and molting (Hoy 1987/TX:L).

Potential Importance to Function: Moderate.

Measure: Determination of the fetch and whether at least 1 acre of the wetland is sheltered.

Directness of Measure: Moderate.

20. Vegetative Canopy (predictor for breeding)

This predictor pertains specifically to nontidal wetlands with a channel (i.e., streambank vegetation canopy).

Ranking: Wetlands adjacent to an upland that shades a significant portion of the wetland at midday are more likely to support a notable on-site diversity and/or abundance of wetland-dependent birds.

Rationale: Adjacent forested uplands that provide shade at midday may moderate the microclimate of wetlands (Johnson and Beck 1988/*). Vegetated contiguous uplands also reduce nonpoint pollution associated with watershed disturbances (Mussallem and Lynch 1980/*, Gosselink and Lee 1987/*) by preserving channel stability, retarding runoff, and trapping sediments and nutrients (Brinson et al. 1981a/*fo; see also Section 3.5, Predictor 36). As a result, loss of species is minimized in wetlands adjacent to large undeveloped uplands. For example, Stauffer and Best (1980/IA:fo) found that a corridor of at least 600 feet was needed to maintain some breeding bird species in Iowa. A similar width was required by many breeding birds in Virginia (Tassone 1981/VA:fo). Also see Predictor 21.

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate.

Measure: Determination of the presence of sufficient vegetative canopy (or topographic relief) in and around the wetland to shade at least 80 percent of the wetland's shallow-water area. The 80-percent threshold is arbitrary.

Directness of Measure: Moderate.

21. Land Cover in the Watershed (predictor for breeding, migration, and wintering)

Ranking: Wetlands having watersheds not dominated by impervious surfaces are more likely to support a notable on-site diversity and/or abundance of wetland-dependent birds. During migration and wintering, wetlands having watersheds dominated by cultivated agricultural areas are more likely to provide notable on-site diversity of wetland-dependent birds, especially waterfowl.

Rationale: No essential cover is provided by impervious surfaces that accompany urban development. High levels of human visitation in urban settings may also discourage wildlife use (Simpson and Kelsall 1979, Ream 1980/*, Ryder et al. 1980, Vaske et al. 1982, Dickman 1987/UK:fo; see also Predictor 30), as may contaminants (see Predictor 27).

Cultivated areas, when associated with wetlands, provide feeding areas for migratory and wintering wildlife, especially waterfowl (Carothers et al. 1974/fo, Whitmore 1975/UT:fo, Conine et al. 1978, Chabreck 1979/*b, Heitmeyer and Frederickson 1981/LA,MO,AR:fo, Guthery et al. 1984/TX:P). Also, by eliminating natural cover, agricultural areas tend to focus bird use on remaining wetlands.

Relative preferences of waterfowl for different terrestrial cover types has been documented for particular species as follows (1 = preferred, where it surrounds wetland; "Group" refers to the waterfowl groups listed in Table 2):

Group	Forested or Scrub	Grassland	Cropland	Developed
1	2	2	1-2	2-3
2	1-2	2	2	2-3
3-6,8	1	1	1	1-2
7,9	2-3	2	1	2-3
Group 1: Glover 1956/IA:Pem, Gates 1965/WI:Pem, Evans and Wolfe 1967/NE:Pem, Martz 1967/ND, Jarvis and Harris 1971/OR:Pem, Oetting and Cassel 1971, Bellrose 1976/*, Duebbert and Lokemoen 1976/SD:em, Higgins 1977/ND:Pem, Vorhees and Cassel 1980.				
Group 2: Coulter and Miller 1968/ME,VT, Prince 1968/NB, Renouf 1972/NB, Reed 1975/QUE:tem, Bellrose 1976/*, Fefer 1977/ME:P, Landers et al. 1977/SC:FO, Flood et al. 1979/*.				
Group 3: Hanson et al. 1949, Kitchen and Hunt 1969/WI, Bellrose 1976/*.				
Group 4: Hanson et al. 1949, Mendall 1958/VT,NH,ME, Prince 1968/NB, Bellrose 1976/*.				
Groups 5 and 6: Bellrose 1976/*, Sugden 1978.				
Group 7: Craighead and Craighead 1949/WY:Rem, Dawe and Davis 1975/BC, Bellrose 1976/*, Raveling 1977/MAN.				
Group 9: Bellrose 1976/*.				

Confidence in Ranking: Moderate to high. Urban wetlands occasionally are used more by wildlife than are wetlands surrounded by forest or other natural cover, due to the focusing of traditional wildlife use into a few relict areas, or to the placement of cities at the mouths of rivers and other ecologically rich sites traditionally used by wildlife (Erwin et al. 1987/US:E). Enriched urban waters may also attract some species (e.g., Campbell 1984/UK:E).

Agricultural wetlands are occasionally used less than natural cover due to disturbance, increased predation, and contamination with pesticides and other substances (Leitch and Kaminski 1985/SAS:Pem, Grue et al. 1986/*, Baines 1988/UK:Pem, Klett et al. 1988). When use is encouraged to the point of causing overcrowding, long-term adverse effects (i.e., disease outbreaks, vegetation denudation) may result. Agriculture and other nutrient-enriching activities may trigger the following sequence of problems in wetlands (Crowder and Bristow 1988/ONT:L).

For the waterfowl, the effect of inshore eutrophication is thus an initial increase in food plants, a gradual replacement of favorite species by less desirable plants, and finally a total loss of submersed and floating-leaved plants coincident with an extension of cattail marsh. The extended marsh in turn declines, having been exposed to wave erosion through loss of the deeper zones of vegetation.

Also, food chains become shorter due to the loss of top predators in eutrophic lakes (Odum 1985/*).

Potential Importance to Function: Moderate.

Measure: Determination of the major land cover type in the wetland's watershed.

Directness of Measure: Low.

23. Ditches/Canals/Channelization/Levees (predictor for breeding, migration, and wintering)

Ranking: Wetlands without artificial structures that increase the flow of surface water from the wetland are more likely to support a notably great on-site diversity and/or abundance of wetland-dependent birds.

Rationale: Obviously, draining a wetland precludes use by wetland-dependent species. Unditched marshes generally have longer hydroperiods, more consistent water depths, richer food resources (see Predictor 23 in Section 3.8), and more wildlife (Clarke et al. 1984/MA:Eem, Portnoy et al. 1987/M:Eem, Wilson et al. 1987b/MA:Eem). Although ditched salt marshes sometimes have greater wildlife diversity, unditched salt marshes often have more uncommon species (Burger et al. 1982/NJ:Eem). Contiguous forested wetlands are often cleared following stream channelization. Even when such wetlands are protected from land clearing, after channelization they often change from dynamic wet systems to drier habitats where flooding occurs infrequently and waterbird use diminishes (Fredrickson 1979/MO:fo).

Confidence in Ranking: Moderate.

Potential Importance to Function: Moderate. The effect depends also on the type of alteration, proximity, extent, construction methods, wetland type, and other factors. Conversion of irregularly flooded (supratidal) wetlands to regularly flooded hydroperiods by means of ditching, as done for "open marsh water management" for mosquito control, can maintain or increase wildlife use (Meredith and Saveikis 1987/DE:tem).

Measure: Determination of evidence for alteration of the wetland's hydrologic regime by ditches, canals, levees, or similar artificial features that cause surface water to leave at a faster rate than it would if these structures were not present.

Directness of Measure: Moderate.

27. Contaminant Sources (predictor for breeding)

Ranking: Wetlands free of potential sources of toxic material are more likely to support a notable on-site diversity and/or abundance of breeding wetland-dependent birds.

Rationale: Input of toxic materials may reduce wildlife populations directly by causing mortality and decreased productivity, or indirectly by impacting the habitat (Crowder and Bristow 1988/ONT:L). Reductions in waterfowl use have been correlated with degraded water quality (e.g., Reichholf 1976/EU).

Confidence in Ranking: High. The effect depends also on the type of contaminant, proximity, season of application, wetland type, consumer behavior, and other factors.

Potential Importance to Function: High. Atmospheric sources of metals and synthetic organics (via deposition or precipitation) may occasionally be significant as well (Lazrus et al. 1979, Rappaport et al. 1985/U.S.).

Measure: Determination of sources of waterborne toxic substances such as mines, landfills, leaking subsurface tanks, salt/brine seepage, pesticide-treated areas, contaminated aquifers, oil contamination, irrigation return water, industrial and sewage outfalls, or heavily traveled highways.

Directness of Measure: Low. Direct measurement of contaminant levels or (better yet) wildlife exposure and accumulation in tissues is preferable to assuming their presence based on potential sources. If sources are assumed, coincidence of periods of maximum runoff and wildlife presence should be considered, as some sources are associated only with seasonally intermittent activities. Many areas that appear to be contaminated continue to attract wildlife (see Predictor 21).

28. Direct Alteration (predictor for breeding, migration, and wintering)

Ranking: Wetlands that have not been altered directly by tilling, filling, excavation, addition of outlets where none existed before, or blockage of inlets are more likely to support a notable on-site diversity and/or abundance of wetland-dependent birds.

Rationale: Direct alteration of wetlands disrupts a variety of ecosystem functions that affect wildlife populations. For example, tillage and excessive or early mowing destroy cover for nesting birds (Duebbert and Frank 1984/ND, SD:Pem, Kantrud and Stewart 1984/ND:Pem).

Confidence in Ranking: Moderate. The effect depends also on the type of alteration, proximity, timing, extent, construction methods, age, wetland type, and other factors.

Potential Importance to Function: High.

Measure: Determination of presence of direct alterations to the wetland occurring within the last 3 years. The 3-year threshold is arbitrary.

Directness of Measure: Low.

30. Disturbance (predictor for breeding, migration, and wintering)

Ranking: Wetlands without major, frequent disturbances are more likely to support a notably great on-site diversity and/or abundance of wetland-dependent birds.

Rationale: Human disturbance, such as hunting (Conroy et al. 1987/NJ:E,P, Gordon et al. 1987/SC:E,P) and people traveling on foot (Burger 1981/NY:Eem), can discourage use of an area by wildlife (Pomerantz et al. 1988/*), especially during the breeding season or during harsh weather. This is particularly true for long-distance migrants that feed in large flocks at the ground or water level (Batten 1977), colonial-nesting species (e.g., Williams and Martin 1970, Markham and Brechtel 1979, Schreiber 1979, Tremblay and Ellison 1979), and large raptors (e.g., Stalmaster and Newman 1978/WA:fo). Disturbance reduced the use of wetlands by waterfowl and nongame waterbirds as well (Kaiser and Fritzell 1984/MO:R, Hoy 1987/TX:L).

Disturbance can also increase the food requirements of waterbirds. Korschgen et al. (1985/MO:P,L) found that only five boating disturbances per day increased the energy requirements of canvasbacks by 20 percent, requiring consumption of an additional 23 grams of food. Wintering bald eagles may take flight when approached to a distance of 800 to 1,000 feet (Stalmaster and Newman 1978, Knight and Knight 1984/WA:R). Motor boats can disturb waterfowl up to 3,300 feet away (Hoy 1987/TX:L), which reduces the birds' energy reserves.

Disturbance also can alter habitat by trampling and fire, particularly in less frequently flooded wetlands and along wetland-upland transition zones (e.g., Cole and Marion 1988/*fo).

Confidence in Ranking: Moderate. Some species adjust somewhat to human visitation. Moderate trampling of vegetation, as occurs with intermediate levels of human visitation, may improve interspersion and diversity of plant forms in some situations, thus improving habitat (Racey and Euler 1982/ONT:fo). Wildlife density may consequently increase in some mildly disturbed areas (Robertson and Flood 1980/ONT:L).

Potential Importance to Function: Low to moderate.

Measure: Determination of access and use of wetland areas by humans. Depth and distance thresholds are arbitrary.

Directness of Measure: Low.

31. Water/Vegetation Proportions (predictor for breeding, migration, and wintering)

Ranking: Wetlands with relatively even proportions of vegetation and water are more likely to support a notable on-site diversity and/or abundance of wetland-dependent birds, primarily waterfowl.

Rationale: Edge length, but not surface water area, was positively correlated with puddle duck production in the Prairie Pothole region (Mack and Flake 1980/SD:Pem). Maximum species richness and abundance occur when a well interspersed cover-to-water ratio of 50:50 exists (Weller and Spatcher 1965/IA:Pem). The greatest densities of nesting dabbling duck pairs tend to be associated with areas having a 50:50 open water-to-vegetation ratio (Kaminski and Prince 1981, Murkin et al. 1982/MAN). Marshes with 50 to 70 percent open water well interspersed with emergent vegetation produce the greatest bird diversities and numbers (Weller and Fredrickson 1974/IA:Pem, Weller 1979/IA:Pem). The proportion of shallow intertidal pools enclosed within a salt marsh can be the best single predictor of avian diversity in the marsh (Wilson et al. 1987b/MA:Eem). See also Rationale for Predictors 15 (Vegetation/Water Interspersion) and 16 (Vegetation Class Interspersion).

Confidence in Ranking: Moderate to high.

Potential Importance to Function: High.

Measure: Determination of vegetation and water proportions in the wetland. All thresholds are arbitrary.

Directness of Measure: Moderate.

32. Hydroperiod (spatially dominant) (predictor for migration, wintering)

Ranking: Wetlands in which at least a portion is permanently flooded, intertidal, or intermittently exposed are more likely to support a notably great on-site diversity and/or abundance of wetland-dependent birds. These need not be the **spatially** dominant hydroperiods in the wetland. Wetlands whose spatially dominant hydroperiod is "artificially flooded nontidal" (as defined by Cowardin et al. 1979/*) are more likely to support a notable on-site diversity and/or abundance of migrating and wintering wetland-dependent birds.

Rationale: Permanently or intertidally flooded and intermittently exposed areas are likely to provide a variety of habitat types ranging from open water to vegetation adapted to moist soil conditions. As a result, conditions are available for a wide variety of wildlife species during most years, and may provide refugia for wetland-dependent species during periods of drought. For example, among forested New England wetlands, saturated wetlands (i.e., relatively stable water levels) have greater bird species richness and density than seasonal (i.e., less stable water levels) wetlands (Swift et al. 1984). In the prairie region, the greatest diversity is supported by permanent wetlands (Faanes 1982/ND:em). In estuarine wetlands, increasing the tidal influence in isolated, supratidal pools increases or maintains waterbird use (Meredith and Saveikis 1987/DE:tem), partly as a result of enhancing growth of key food plants (Mahaffy 1987/DE:ab).

However, in the prairie region the greatest **density** of breeding birds is in semipermanent wetlands (Faanes 1982/ND:em, Kantrud and Stewart 1984/ND:Pem).

Inland wetlands that have arisen or been maintained as a result of flooding from dams, pumps, or siphons are classified as "artificially flooded" and often serve as resting points for migrating wildlife (e.g., Hoy 1987/TX:L), especially in coastal areas that have lost much of their original acreage of freshwater wetland. Most state and Federal waterfowl refuges fall into this category. Also, these areas are often used as wintering sites as long as they remain ice-free and adequate food is available.

Artificial wetlands whose water levels have been regulated specifically for vegetation management aimed at optimizing wildlife use (Weller 1978/*) are likely to be productive. This is particularly true if they are larger than 1 acre, not long and narrow, and are drawn down and flushed of excessive sediment, organic matter, and salts every few years (Hardin 1987/DE:tem, Whitman and Cole 1987/DE:E,P). Thus, many artificially flooded areas (e.g., greentree reservoirs, managed marshes) are likely to provide the habitat features necessary to support a high abundance and diversity of wildlife species on a consistent basis (Gordon et al. 1987/SC:E,P, Prevost 1987/SC:E,P).

The habitat suitability evaluation for individual waterfowl groups (Table 2) in Volume II is supported, in the case of this predictor, by the adjoining tabulation:

Group 1: Stewart.and Kantrud 1973/ND:Pem, Kantrud and Stewart 1977/ND:Pem, Dwyer et al. 1979/ND, Ruwaldt et al. 1979/SD:Pem.
Groups 2, 4, and 7: Reinecke and Owen 1980/ME:em.

Confidence in Ranking: Low to moderate. The benefits depend on the specific management scheme being employed, the wetland type, location, age, sediment type, and other factors.

Potential Importance to Function: Moderate.

Measure: Determination of the wetland's dominant hydroperiod.

Directness of Measure: Low.

33. Most Permanent Hydroperiod (predictor for breeding)

Ranking: Wetlands having areas of at least 1 acre or 10 percent of their area that is permanently flooded or intermittently exposed as their most permanent hydroperiods are more likely to support a notably great on-site diversity and/or abundance of wetland-dependent birds.

Rationale: Permanently flooded and intermittently exposed areas are likely to provide a variety of habitat types ranging from open water to vegetation adapted to moist soils. As a result, habitats are available for a wide variety of bird species during most years, and may provide refugia for wetland-dependent birds during periods of drought. For example, Faanes (1982/ND:P) noted that the greatest on-site diversity in the prairie region was supported by permanent wetlands, whereas the greatest density of birds was in semipermanent wetlands. Kantrud and Stewart (1984/ND:Pem) also found the greatest density of breeding birds in semipermanent wetlands. Swift et al. (1984/MA:Pfo) found that, among forested New England wetlands, saturated wetlands (i.e., relatively stable water levels) had greater bird species richness and density than seasonal wetlands (i.e., less stable water levels). However, Weller (1978/*) noted that because many emergent plants germinate only in shallow water or on mud flats, periodic drawdowns (i.e., area is semipermanently flooded) are necessary to maintain the 1:1 cover-to-water ratio which provides the greatest species richness in freshwater marshes.

Confidence in Ranking: Moderate. Because many emergent plants germinate only in shallow water or on mud flats, periodic drawdowns (i.e., area is intermittently exposed) are necessary in some situations to maintain optimal cover-to-water ratios for wildlife. Thus, in these situations, permanently flooded wetlands may be less attractive to waterbirds (Weller 1978/*). Permanent flooding may also reduce on-site bird diversity in forested wetlands (Klimas et al. 1981/*fo).

Potential Importance to Function: Moderate.

Measure: Determination of the most permanent hydroperiod in the wetland.

Directness of Measure: Low to moderate.

34. Water Level Control (predictor for migration and wintering)

Water control structures designed specifically for wildlife management are excluded from this predictor.

Ranking: Wetlands dependent upon upstream or downstream control structures (other than those designed specifically for fish and wildlife management) are less likely to support a notably great on-site diversity and/or abundance of migrating and wintering wetland-dependent birds.

Rationale: Wetlands dependent upon upstream or downstream artificial structures (e.g., dams and dikes) for their existence are likely to undergo large, sudden water-level fluctuations. Large water-level fluctuations are likely to have detrimental impacts on habitats used by migrating and wintering wildlife.

Confidence in Ranking: Moderate. The effect depends also on the type of dam, proximity, extent, wetland type, and other factors.

Potential Importance to Function: High.

Measure: Determination of existence of artificial control structures upstream or downstream which may impact the wetland. All thresholds are arbitrary.

Directness of Measure: Low.

36. Vegetated Width (predictor for breeding)

Ranking: Wetlands with greater vegetated widths are more likely to support a notably great on-site diversity and/or abundance of breeding wetland-dependent birds.

Rationale: See Predictor 20 (Vegetative Canopy).

Confidence in Ranking: Moderate.

Measure: Determination of the average width of erect vegetation in the wetland. The 20- and 500-foot thresholds are arbitrary.

Directness of Measure: Moderate.

38. Type Combinations (predictor for breeding, migration, and wintering)

Ranking: Wetlands that are near wetlands of a different classification are more likely to support notable on-site diversity and/or abundance of wetland-dependent birds.

Rationale: Combinations of different wetland types in proximity benefit wildlife by providing diverse foods, or because preferred foods are found under different conditions than are preferred nesting or roosting sites. Type combinations are especially important for large and/or wide-ranging water-birds (e.g., waterfowl, shorebirds), and during periods when seasonal changes in food and cover availability occur (Powell 1987/FL:Eem). Thus, regional wetland diversity, not simply within-wetland diversity, is essential for maintaining avian diversity/abundance (Weller and Fredrickson 1974/IA:Pem, Patterson 1976, Chabreck 1979/*b, Flake 1979/*, Weller 1979/*, Burger et al. 1982/NJ:Eem, Josselyn 1982*CA, Harris and Vickers 1984/FL:P, Nelson and Kadlec 1984/ *Pem, Conroy et al. 1987/NJ:E,P, Gordon et al. 1987/SC:E,P, Gray et al. 1987, Morton et al. 1987/VA:E,P).

A regional diversity of hydroperiod types is particularly important (Dzubin 1969, Dwyer et al. 1979/ND, Ruwaldt et al. 1979/SD:Pem, Kantrud and Stewart 1984/ND:Pem, Powell 1987/FL:Eem), especially in precipitation-deficit regions and in tidal systems.

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of the wetland type and its proximity to wetlands of another type. Thresholds used for size and degree of isolation are arbitrary.

Directness of Measure: Moderate to high.

39. Special Habitat Features (predictor for breeding, migration, and wintering)

Ranking: Wetlands containing special habitat features such as standing snags with cavities larger than 2 inches; trees with diameters greater than 10 inches; plants bearing fleshy fruits, mast, or cones; tilled land with waste grain; evergreen tree stands with over 80 percent canopy closure; native prairie; or exposed bars are more likely to support a notable on-site diversity and/or abundance of wetland-dependent birds than those without such features.

Rationale: Many wetland species require these special habitat features to meet life requisites for reproduction, food, or cover. For example, many cavity-nesting birds require the presence of standing snags with appropriate size cavities for nest sites (Wetzel 1958, Hair et al. 1978). Heron rookeries are often situated in large trees (Bjorkland 1975, Werschkul et al. 1976, Portnoy 1978, White et al. 1982). Exposed bars are required by many shore and wading birds for nesting, roosting and/or feeding sites (Dodd 1978/*, Powell 1987/FL:Eem). Fleshy fruits, mast, cones, and waste grain provide food for a variety of wetland species (Martin et al. 1951/*, Collins 1961, Beck 1977). The wintertime abundance and distribution of this food source may limit North American songbirds at least as strongly as summer nesting habitat (Harris 1984/*fo).

Confidence in Ranking: Moderate. Such features, although beneficial to wildlife, are not always so locally scarce as to be limiting.

Potential Importance to Function: High.

Measure: Documentation of presence of the special habitat features. The thresholds are only weakly based on empirical data and, in some cases (e.g., tree diameter), should be regionally specific. Wetland size categories are arbitrary.

Directness of Measure: Moderate.

41. Velocity (spatially dominant) (predictor for breeding)

Ranking: Wetlands with slow flow velocities are more likely to support a notable on-site diversity and/or abundance of breeding wetland-dependent birds.

Rationale: See Rationale for Predictor 7 (Gradient).

Confidence in Ranking: Low.

Potential Importance to Function: Low to moderate.

Measure: Determination of the velocity throughout most of the wetland during annual peak flow. Thresholds are arbitrary.

Directness of Measure: Low.

45. Substrate Type (spatially dominant) (predictor for breeding, migration, and wintering)

Ranking: Wetlands having substrates other than bedrock, rubble, or cobble-gravel are more likely to support a notable on-site diversity and/or abundance of wetland-dependent birds.

Rationale: Mineral, muck, peat, and sandy soils are more likely to support sufficient vegetation to provide food and cover for a diversity of avian species than are bedrock, rubble, or cobble-gravel substrates.

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of the dominant surface sediment in the upper 3 inches of the wetland.

Directness of Measure: Low to moderate.

47. pH (predictor for migration and wintering)

Ranking: Wetlands with generally circumneutral to alkaline (pH > 6.0) waters are more likely to support a notable on-site diversity and/or abundance of migrating and wintering wetland-dependent birds.

Rationale: Alkaline pH values generally reflect better buffering and higher productivity, whereas highly acidic conditions result in impoverished fauna (Cook and Powers 1958/NY:Pem, Blancher and McNicol 1988). Species diversity and productivity are greatest in the pH range of 5.6 to 8.5. See also the discussion of this predictor in Section 3.8 (Aquatic Diversity/Abundance).

Confidence in Ranking: Moderate.

Potential Importance to Function: Low to moderate.

Measure: Determination of the pH of the wetland's waters.

Directness of Measure: Low to moderate.

48. Salinity and Conductivity (predictor for breeding)

Ranking: Wetlands with salinities less than 30 ppt are more likely to support a notably great diversity and/or abundance of breeding wetland-dependent birds.

Rationale: Few ducks use marine wetlands during the brood rearing period (Weller 1975:100/*). Alkali wetlands typically have low densities of breeding waterfowl (Kantrud and Stewart 1977/ND:Pem). Similarly, Patterson (1976) found negative relationships between chloride concentrations of pond water and use of those ponds by breeding duck pairs or broods in Ontario. The number of plant species found in prairie potholes decreases markedly with increasing salinity (Stewart and Kantrud 1972/ND:P). This reduction in plant species diversity no doubt influences selection and use of such areas by birds.

The habitat suitability evaluation for individual waterfowl groups (Table 2) in Volume II is supported, in the case of this predictor, by the following:

Nesting/summering suitability:
Groups 1, 5, and 6: Parnell and Quay 1962/NC:em, Christiansen and Low 1970/UT:Eem, Bellrose 1976/*, Swanson et al. 1979/ND:Pem.
Group 2: Hanson et al. 1949, Coulter and Miller 1968/ME,VT, Reed and Moisan 1971/QUE:Eem, Bellrose 1976/*, Chabreck 1979/*, Reinecke and Owen 1980/ME:em.
Group 3: Bellrose 1976/*.
Group 7: Craighead and Craighead 1949/WY:Rem, Dawe and Davis 1975/BC, Raveling 1977/MAN.

Migration/wintering suitability:
Group 2: Mendall 1949/ME, Hartman 1960/ME:tem, Palmisano 1973/*Eem, Landers et al. 1976/SC:Eem, Chabreck 1979/*b, Odum et al. 1979/*Eem.
Group 3: Erskine 1971/NS.

Confidence in Ranking: High.

Potential Importance to Function: High.

Measure: Determination of the salinity levels of wetland waters. Thresholds are arbitrary.

Directness of Measure: Low to moderate.

50. Plants: Waterfowl Value (predictor for migration and wintering)

Ranking: Wetlands containing food plants preferred by waterfowl are more likely to support a notably great on-site diversity and/or abundance of migrating and wintering wetland-dependent birds. Preferred plants are shown in Table 5 of Volume II.

Rationale: Wetlands providing food and cover for waterfowl are likely to be attractive for other species.

Confidence in Ranking: High (for waterfowl) to moderate (other species).

Potential Importance to Function: Moderate to high. Presence of rich invertebrate communities (not necessarily associated with preferred plants) may be at least as important (Euliss and Grodhaus 1987/CA:Pem, Swanson 1988/*) and is described by predictors in Section 3.8.

Measure: Documentation of preferred waterfowl food plants covering at least 10 percent or 1 acre of the wetland. Thresholds are arbitrary.

Directness of Measure: Moderate. Site-specific (or at least region-specific) data on food preferences are preferred.

4.0 Social Significance of Wetland Functions

Wetland functions, as physical or biological processes or conditions, are not always of direct relevance or usefulness to human interests. Often, the effects of a function (e.g., retention of sediment) can be viewed as either favorable or unfavorable, depending on the perspectives of the local public or the scientific community (Table 3). Moreover, land development has social significance, too. Thus, the choice is not merely whether a wetland should be allowed to continue providing a service, but whether a proposed development offers a greater long-term, maintainance-free service than a wetland.

Table 3
Conflicting Social Perspectives and Uses of Wetland Functions

Function	Beneficial	Negative
Ground water recharge	Increased water supplies; blockage or dilution of contamination	Surface flow reduction
Floodflow alteration	Flood control	Less flushing (e.g., of silt from salmon spawning gravel)
Sediment stabilization	Shoreline protection	Less flushing (e.g., more frequent dredging required)
Sediment/toxicant retention	Improved downslope environment	Degraded wetland environment
Nutrient removal/ transformation	Tertiary waste treatment by nature (especially important for nonpoint sources	Eutrophication of wetland
Production	Food chain support	Increased export biological oxygen demand; production of hydrogen sulfide; source of weeds; increased drifting of snow near roads
Aquatic diversity/ abundance	Food chain support	Nuisance insects
Wildlife diversity/ abundance	Recreational hunting and observation	Crop depradation; vehicle collision hazard
Recreation	Source of recreational opportunities	Disturbance of wildlife by visitors
Uniqueness/ heritage	Source of aesthetic pleasure	

Attitudes toward wetlands differ among social groups and geographic locations and over time. Dealing with changing attitudes over time presents a particularly troublesome problem in social valuation of wetlands, especially if early attitudes are favorable for wetland destruction. In this situation, wetland destruction would make later preservation impossible, should the socioeconomic climate change to favor preservation.

No attempt is made by WET to place a dollar value on a unit of wetland, partly because many values are intangible and irreplaceable, and partly because technical aspects are insufficiently understood to allow the degree of quantification necessary for economic analysis. Therefore, WET evaluates wetland social significance from a broader perspective by considering the following parameters:

- Official recognition.
- Demand for wetland-based functions.
- Supply of wetland-based functions.
- Availability of substitutes.

Each of these is discussed in the following sections. Also see Sections 2.9 and 2.10 for discussions of social significance as it applies to Recreation and Uniqueness/Heritage.

4.1 Official Recognition

One measure of wetland significance is to identify whether other agencies or institutions have designated a wetland as being somehow special. For example, Goodwin and Niering (1975/*) identified 358 "nationally significant" wetlands, based on a survey of scientists' opinions. The US Environmental Protection Agency has undertaken "Advanced Identification" initiatives aimed at ranking mostly inland wetlands and a "National Estuaries" effort aimed at coastal wetlands. The Corps of Engineers has ranked wetlands using a process termed "Special Area Management Planning." The National Oceanographic and Atmospheric Administration has targeted specific "Estuarine Sanctuaries" for protection and research. The US Fish and Wildlife Service, working with state wildlife and outdoor recreation agencies, is systematically identifying priority wetlands for purchase or easement, and documenting these decisions in "Service Wetland Plans" and "State Outdoor Comprehensive Recreation Plans (SCORPs)" for each state.

The states of New York, Maine, New Jersey, Florida, Maryland, and others are also in the process of ranking parts of their wetland resource. On a private level, The Nature Conservancy, Ducks Unlimited, National Audubon Society, and others have targeted specific wetlands for acquisition, and such information may be used as one factor in judging significance.

Interpreting such official recognition can sometimes be difficult for the following reasons:

(1) Rankings and subsequent designations may have been focused on only a few of many possible wetland functions (e.g., typically on habitat value, to the possible detriment of hydrologic services).

(2) Nearly all such designations have been assigned without comprehensive, technically based field evaluations of all candidate wetlands. In the worst cases, they reflect the biases and values of specialists or narrow segments of the public.

(3) Designations may have been assigned to a larger land unit (e.g., National Park) of which the wetland comprises only a small part; the exact role an included wetland had in the overall designation of the area is nearly impossible to discern.

4.2 Demand for Wetland-Based Functions

Demand for wetland-based functions is often difficult to estimate, and many of the economic techniques used to evaluate user demand have some significant problems when applied to wetland benefits in general. WET uses the *relative location of a wetland as an indicator of the* probable demand for various functions.

Wetlands must be located so that potential benefits from functions performed may be realized by society, in order for society to place significance on the existence of such functions. For example, a wetland downstream of a major community may be valuable for recreation, but its location is such that it will probably have minimal, or possibly negative, value to that community for floodflow alteration. Existing scientific and economic data are usually insufficient to determine how fast a wetland's services diminish as one moves away from the point where services are normally delivered. A few functions, such as endangered species habitat, are valued over a very wide region, often by "users" who never intend to visit the wetland. These users are content with the knowledge that something irreplaceable continues to exist. Other functions, such as sediment stabilization, are usually of great significance only to residents in the immediate vicinity of the wetland.

4.3 Supply of Wetland-Based Functions

Specific quantitative measures of wetland functions (e.g., presence of endangered/threatened species, number of ducks produced, capacity of existing dams for flood control, model-determined estimates of the cumulative effects of channel roughness) provide ideal but often unobtainable data for determining the supply of wetland benefits. Lacking such data, an alternative is to perform functional evaluations (e.g., WET assessments) for each individual wetland in a watershed or region. Such analyses would provide information on the number of wetlands that would be likely to supply specific benefits in the region. However, performing individual analyses for all wetlands in a region would require a significant commitment of time and resources. From experience, we estimate this involves approximately 2 to 4 person-months of effort per 100 square miles of land surface, with survey efficiency being greatest where wetlands have already been located, densities of wetlands on the landscape are great, and road/water access is good.

If the resources to prepare this kind of exhaustive analysis for all wetlands in a region are not available, an alternative (but much less effective approach) is to assume all wetlands to be of equal value per unit area and to use wetland acreage as an indicator of the supply of wetland benefits in the region. Wetland maps from which acreage data can be compiled are available for nearly two thirds of the nation from the US Fish and Wildlife Service's National Wetlands Inventory.

4.4 Availability of Substitutes

Another consideration in assessing the social significance of wetlands is the existence of possible substitutes for wetland functions, such as those shown in Table 4. Although wetlands are less efficient and reliable for some functions in comparison to certain functional substitutes (e.g., wastewater wetlands versus engineered treatment facilities), these substitutes usually do not provide exactly the same functions as the wetland under consideration. Often these substitutes are more costly, less enduring, and plagued with more adverse secondary impacts than would be the case if natural wetlands were used to provide the same function. However, these substitutes should be considered when determining the supply and demand for functions performed by natural wetlands. Also, some wetland functions, such as Uniqueness/Heritage, have no realistic substitutes.

4.5 Wetland Context and Cumulative Impacts Evaluation

Spatial, temporal, and sociopolitical context must be established before attempting to estimate wetland social significance. Spatial context pertains to the significance of wetlands in relation to the area considered. For example, a particular wetland type or function may be scarce locally, although regionally and nationally it may be abundant. The temporal context refers to the temporal trend in availability of the wetland type under consideration. For example, a particular wetland type may be decreasing, although it is still relatively plentiful. Conversely, another wetland type might be increasing but may presently be

Table 4
Examples of Potential Substitutes for Wetland Services (Listing Herein Does Not Imply Endorsement of Any Measure)

Natural Function of Wetland	Examples of Services	Possible Substitutes
Ground water recharge	Replenish aquifer Provide more water for consumptive use	Water conservation regulations Phreatophyte management Mechanical injection Impoundment construction Protection of upland recharge areas
Floodflow altertion	Reduce flood losses Reduce scour of aquatic communities	Impoundment construction Watershed land cover management Flood insurance Stream habitat improvement measures
Sediment stabilization	Reduce flood damages and shoreline erosion	Construction of jetties and breakwaters Shoreline development regulations Flood insurance
Sediment/toxicant retention	Improve water quality	Waste treatment plant Sediment basins/impoundments Terrestrial buffer strips Land treatment measures
Nutrient removal/ transformation	Improve water quality Reduce nuisance algal blooms	Waste treatment plant Nonpoint discharge control Terrestrial buffer strips Chemical control of algae or nutrient cycling Land treatment measures
Aquatic and wildlife diversity/ abundance	Recreation (aesthetics, commercial sport fishing, etc.)	Habitat acquisition protection (e.g., acquire water rights Alternative harvest regulations Predator management Creation and use of propagation facilities (e.g., hatcheries) Habitat management Research on productivity factors Control of pollutants
Recreation	Recreation	Purchase of or easement on upland Intensified management of existing areas

uncommon. The sociopolitical context addresses the perception of wetlands or their functions by various sociopolitical groups. For example, a particular wetland type or function may be equally scarce in two neighboring localities, but may be used or appreciated much more by the citizens of one community, due to distinct sociological and economic characteristics.

Neither science nor economics alone can determine the appropriate evaluation context. However, clues for helping to determine the appropriate spatial and sociopolitical contexts may be obtained by examining institutional policies and legislation. For instance, adoption of voluntary floodplain zoning by local governments in a project area may be taken as an indication that the potential flood control significance of a particular wetland is likely to be viewed in a favorable sociopolitical light. On the other hand, if a particular wetland is thought to be ecologically special, but there are no local ordinances related specifically to protection of ecological resources and there is little other evidence of public sentiment for or against wetlands in that particular community, then the "uniqueness" is probably best decided by examining relative scarcity statewide or nationwide, rather than locally.

The temporal context is particularly vexing because, once altered, it is difficult, if not impossible, to return a wetland to its original functional state. The temporal context should consider direct impacts anticipated for wetlands in the area, as well as cumulative impacts of activities that affect the wetland. Conceptual approaches for evaluating cumulative impacts are presented by Clark and Zinn (1978/*), Coates et al. (1981/*), Gosselink and Lee (1987/*), and others.

The importance of considering the cumulative effects (in time and space) of all activities on wetlands cannot be overstated. Cumulative impacts on the wetland resource result from many individually small, seemingly insignificant developments which collectively may result in substantial functional losses, often at the landscape level. The economic growth induced by developments may have a further ripple effect on wetland quality and landscape function. A typical pattern of wetland loss in many regions is that small wetlands are drained or filled first, leaving only the larger wetlands, which then become filled by runoff that was once held by the many smaller wetlands (Evans and Black 1956/SD:Pem).

The following questions can be evaluated to determine the probable threat and seriousness of cumulative impacts:

(1) What is the anticipated growth in this locality or region, both in spite of the proposed development and because of it (induced growth)?

(2) What is the potential for destruction of this and other wetlands (particularly those of its type) in the region? Could a large percentage of these be cost-effectively logged, farmed, impounded, developed for seasonal recreation, etc., or do they have major physical and economic constraints (i.e., long distance from markets and transportation routes, adverse tax climate, unsuitable soils) that might deter ecologically detrimental uses. To what degree are economically feasible activities in the region dependent on wetlands or water locations?

(3) Is there an extensive network of local or state laws that protect wetlands? How aggressively are incentive-based wetland/riparian conservation programs of the US Department of Agriculture/Soil Conservation Service, the US Fish and Wildlife Service, and others being implemented? Does the existing patchwork of regulations governing development of nonwetland parts of the landscape unintentionally shift the development pressure onto wetlands?

(4) Are there likely to be strong functional links among the region's wetlands, and between the wetlands and the locations where their services are provided? For example, does the region's wetland-dependent fauna include a large component of wide-ranging species with requirements for a diversity of wetland types? Are wetlands tightly linked hydrologically, such that changes in a few may trigger more widespread effects?

5.0 Literature Cited

Ackroyd, E. A., W. C. Walton, and D. L. Hills. 1967. Groundwater contribution to streamflow and its relation to basin characteristics in Minnesota. Minnesota Geological Survey, Report of Investigations 6. 36 p.

Adams, M. S., and R. T. Prentki. 1982. Biology, metabolism, and functions of littoral submersed weedbeds of Lake Wingra, Wisconsin. J. Ecol. 62:457-467.

Adams, S. M., and J. W. Angelovic. 1970. Assimilation of detritus and its associated bacteria by three species of estuarine animals. Ches. Sci. 11:249-254.

Adamus, P. R. 1983. A method for wetland functional assessment; Vol II, FHWA assessment method. US Dept. Trans., Fed. Highway Admin. Rep. No. FHWA-IP-82-24. 134 p.

Adamus, P. R., and R. T. Stockwell. 1983. A method for wetland functional assessment; Vol I, Critical review and evaluation concepts. Rep. FHWA-IP-82-23. Federal Highway Admin., US Dept. of Transportation. 176 p.

Adamus, P. R., E. J. Clairain, Jr., R. D. Smith, and R. E. Young. 1987. Wetland evaluation technique (WET); Vol II, Methodology (Operational Draft Report). Environmental Laboratory, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Aho, J. M. 1978. Freshwater snail populations and the equilibrium theory of island biogeography; II, Relative importance chemical and spatial variables. A. Zool. Fenn. 15:155-164.

Aldridge, B. N., and J. M. Garrett. 1973. Roughness coefficients for stream channels in Arizona. US Geol. Surv. Open-File Report. 49 p.

Allan, P. F. 1956. A system for evaluating coastal marshes as duck winter range. J. Wildl. Manage. 20:247-252.

Allanson, B. R. 1973. The fine structure of the periphyton of *Chara* sp. and *Potamogeton natans* from Wytham Pond, Oxford, and its significance to the macrophyte-periphyton metabolic model of R. G. Wetzel and H. L. Allen. Freshw. Biol. 3:535-542.

Allen, C. E. 1980. Feeding habits of ducks in a greentree reservoir in eastern Texas. J. Wildl. Manage. 44:232-236.

Allen, H. H. 1979. Role of wetland plants in erosion control of riparian shorelines. Pages 403-414 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Tech. Pub. 79-2. Am. Water Resour. Assoc., Minneapolis, MN.

Ambuel, B., and S. A. Temple. 1983. Area-dependent changes in the bird communities and vegetation of southern Wisconsin forests. Ecology 64:1057-1068.

Ammon, D. C., W. C. Huber, and J. P. Heaney. 1981. Wetland use for water management in Florida. ASCE J. Water Resour. Plan. Manage. Div. 315 p.

Andersen, V. J. M. 1974. Nitrogen and phosphorus budgets and the role of sediments in six shallow Danish lakes. Arch. Hydrobiol. 74:528-550.

Anderson, F. O., and B. T. Hargrave. 1984. Effects of *Spartina* detritus enrichment on aerobic-anaerobic benthic metabolism in an intertidal sediment. Mar. Ecol. Prog. Ser. 16:161-171.

Anderson, M. G., and J. P. Low. 1978. Use of sago pondweed by waterfowl on the Delta Marsh, Manitoba. J. Wildl. Manage. 40:233-242.

Anderson, N. H., and J. R. Sedell. 1979. Detritus processing by macroinvertebrates in stream ecosystems. Ann. Rev. Entomol. 24:351-377.

Anderson, N. H., and K. W. Cummins. 1979. Influences of diet on the life histories of aquatic insects. J. Fish. Res. Board Can. 36:335-342.

Anderson, R. V., and D. M. Day. 1986. Predictive quality of macroinvertebrate-habitat associations in lower navigation pools of the Mississippi River. Hydrobiol. 136:101-112.

Andren, J., P. Angelstam, E. Lindstrom, and P. Widen. 1985. Differences in predation pressure in relation to habitat fragmentation. Oikos 45:273-277.

Andrews, C. B., and M. P. Anderson. 1978. Impact of a power plant on the ground water system of a wetland. Ground Water 16:105-111.

Angelstam, P. 1986. Predation on ground-nesting birds' nests in relation to predator densities and habitat edge. Oikos 47:365-373.

Arcement, G. J., Jr., and V. R. Schneider. 1984. Guide for selecting Manning's roughness coefficients for natural channels and flood plains. FHWA-TS-84-204. Fed. Highway Admin., US Dept. of Transporation. 68 p.

Arndt, J. L., and J. L. Richardson. 1986. The effects of groundwater hydrology on salinity in a recharge-flowthrough-discharge wetland system in North Dakota. Pages 269-277 in G. Van der Kamp and H. Maathuis (eds.), Proc. Third Can. Hydrogeol. Conf. Saskatchewan Res. Counc., Saskatoon.

Arnold, J. G., M. D. Bircket, J. R. Williams, W. F. Smith, and H. N. McGill. 1987. Modeling the effects of urbanization on basin water yield and reservoir sedimentation. Water Resour. Bull. 23:1101-1107.

Arvola, L. 1984. Vertical distribution of primary production and phytoplankton in two small lakes with different humus concentration in southern Finland. Holarct. Ecol. 7:390-398.

Askins, R. A., M. J. Philbrick, and D. S. Sugeno. 1987. Relationship between the regional abundance of forest and the composition of forest bird communities. Biol. Conserv. 39:129-152.

Atchue, J. A., III, F. P. Day, Jr., and H. G. Marshall. 1983. Algal dynamics and nitrogen and phosphorus cycling in a cypress stand in the seasonally flooded

Great Dismal Swamp. Hydrobiol. 106:115-122.

Aubertin, G. M., and J. H. Patric. 1974. Water quality after clearcutting a small watershed in West Virginia. J. Environ. Qual. 3:243-249.

Austin, G. T. 1970. Breeding birds of desert riparian habitat in southern Nevada. Condor 72:431-436.

Avnimelech, H., and J. D. McHenry, Jr. 1984. Decomposition of organic matter in lake sediments. Environ. Sci. Technol. 18:5-11.

Axelrod, D. M., K. A. Moore, and M. E. Bender. 1976. Nitrogen, phosphorus, and carbon flux in Chesapeake Bay marshes. Va. Water Resour. Ctr. Bull. 79. Virginia Polytechnic Inst., Blacksburg. 182 p.

Bach, S. C., G. W. Thayer, and M. W. LaCroix. 1986. Export of detritus from eelgrass (*Zostera marina*) beds near Beaufort, North Carolina, USA. Mar. Ecol. Prog. Ser. 28:265-278.

Bachman, R. A. 1984. Foraging behavior of free-ranging wild and hatchery brown trout in a stream. Trans. Am. Fish. Soc. 113:1-32.

Bahr, L. M., Jr., R. Costanza, J. W. Day, Jr., S. E. Bayley, C. Neill, S. G. Leibowitz, and J. Fruci. 1983. Ecological characterization of the Mississippi deltaic plain region: A narrative with management recommendations. FWS/OBS-82/69. US Fish Wildl. Serv., Washington, DC. 189 p.

Bailey, M. W. 1978. A comparison of fish populations before and after extensive grass carp stocking. Trans. Am. Fish. Soc. 107:181-206.

Baines, D. 1988. The effects of improvement of upland, marginal grasslands on the distribution and density of breeding wading bird (Charadriiformes) in Northern England. Biol. Conserv. 45:221-236.

Baker, J. A., and S. T. Ross. 1981. Spatial and temporal resource utilization by Southeastern cyprinids. Copeia 1981:178-189.

Baker, J. P., and C. L. Schofield. 1982. Aluminum toxicity to fish in acidic water. Water Air Soil Pollut. 18:289-309.

Balling, S. S., and V. H. Resh. 1982. Arthropod community response to mosquito control recirculation ditches in San Francisco Bay salt marshes. Environ. Entomol. 11:801-808.

Balling, S. S., Stoehr, T., and Resh, V. H. 1980. The effects of mosquito control recirculation ditches on the fish community of a San Francisco salt marsh. Calif. Fish Game 66:25-34.

Banoub, M. W. 1975. The effects of reeds on the water chemistry of Gnadensee (Bodensee). Arch. Hydrobiol. 75:500-521.

Barbour, C. D., and J. H. Brown. 1974. Fish species diversity in lakes. Am. Nat. 108:473-488.

Barko, J. W., and R. M. Smart. 1980. Mobilization of sediment phosphorus by submersed freshwater macrophytes. Freshw. Biol. 10:229-238.

Barko, J. W., and R. M. Smart. 1983. Effects of organic matter additions to sediment on the growth of aquatic plants. J. Ecol. 71:161-175.

Barko, J. W., D. G. Hardin, and M. S. Matthews. 1984. Interactive influences of light and temperature on the growth and morphology of submersed freshwater macrophytes. Tech. Rep. A-84-3. US Army Engineer Waterways Experiment Station, Vicksburg, MS. 24 p.

Barlocher, F., J. Gordon, and R. J. Ireland. 1988. Organic composition of seafoam and its digestion by *Corophium volutator* (Pallus). J. Exp. Mar. Biol. Ecol. 115:179-186.

Barnby, M. A., and V. H. Resh. 1980. Distribution of arthropod populations in relation to mosquito control recirculation ditches and natural channels in the Petaluma Salt Marsh of San Francisco Bay. Proc. Calif. Mosquito and Vector Control Assoc. 48:100-102.

Barnby, M. A., J. N. Collins, and V. H. Resh. 1985. Aquatic macroinvertebrate communities of natural and ditched potholes in a San Francisco Bay salt marsh. Est. Coast. Shelf Sci. 20:331-147.

Barnes, R. S. K. 1988. The faunas of landlocked lagoons: Chance differences and the problems of dispersal. Est. Coast. Shelf Sci. 26:309-318.

Barton, B. A. 1977. Short-term effects of highway construction on the limnology of a small stream in southern Ontario. Freshw. Biol. 7:99-108.

Barton, D. R., W. D. Taylor, and R. M. Biette. 1985. Dimensions of riparian buffer strips required to maintain trout habitat in Southern Ontario streams. N. Am. J. Fish. Manage. 5:364-378.

Batema, D. L., G. S. Henderson, and L. H. Frederickson. 1985. Wetland invertebrate distribution in bottomland hardwoods as influenced by forest type and flooding regime. Pages 196-202 in Proc. 5th Central Hardwood Conference. Univ. Illinois, Urbana.

Batten, L. A. 1977. Sailing on reservoirs and its effects on water birds. Biol. Conserv. 11:49-58.

Baumann, R. H., J. W. Day, Jr., and C. A. Miller. 1984. Mississippi delta wetland survival: Sedimentation versus coastal submergence. Science 224:1093-1095.

Bay, R. 1969. Runoff from small peatland watersheds. J. Hydrol. 9:90-103.

Bay, R. R. 1967. Ground water and vegetation in two peat bogs in northern Minnesota. Ecology 48:308-310.

Bayless, J., and N. B. Smith. 1967. The effects of channelization upon fish populations of lotic waters in eastern North Carolina. Proc. S.E. Assoc. Game Fish Comm. 18:230-238.

Bayley, S.E., J. Zoltek, Jr., A. J. Hermann, T. J. Dolan, and L. Tortora. 1985. Experimental manipulation of nutrients and water in a freshwater marsh; Effects on biomass, decomposition, and nutrient accumulation. Limnol. Oceanog. 30:500-512.

Baylor, E. R., and W. H. Sutcliffe. 1963. Dissolved organic matter in seawater as a source of particulate food. Limnol. Oceanogr. 8:369-371.

Beauchamp, S. T., and J. J. Kerekes. 1980. Comparative changes in water chemistry within impounded and natural freshwater marshes at the Tintamarre National Wildlife Area. Trans. N.E. Sec. Wildl. Soc. 37. Ellenville, NY.

Beck, D. E. 1977. Twelve-year acorn yield in Southern Appalachian oaks. US For. Serv. Res. Note ES-244. 8 p.

Beck, K. C., J. H. Reuter, and E. M. Perdue. 1974. Organic and inorganic geochemistry of some coastal plain rivers of the southeastern United States. Geochim. Cosmochim. Acta 3:341-364.

Becket, D. C., C. R. Bingham, and L. G. Sanders. 1983. Benthic macroinvertebrates of selected habitats of the Lower Mississippi River. J. Freshw. Ecol. 2:247-261.

Bedient, P. P., W. C. Huber, and J. P. Heaney. 1976. Modeling hydrologic-land use interactions in Florida. Pages 362-366 in W. R. Ott (ed.), Proc. Conf. on Environmental Modeling and Simulation. EPA 600/9-76-016. US Environmental Protection Agency, Washington, DC.

Beebee, T. J. C. 1987. Eutrophication of heathland ponds at a site in southern England: Causes and effects, with particular reference to the amphibia. Biol. Conserv. 42:39-52.

Bell, M. C. 1973. Fisheries handbook of engineering requirements and biological criteria. Fisheries-engineering research program. US Army Engineer Division, North Pacific, Portland, OR. 498 p.

Bella, D. A., A. F. Ramm, and P. E. Peterson. 1972. Effects of tidal flats on estuarine water quality. Water Pollut. Cont. Fed. J. 44:541-556.

Bellrose, F. C. 1976. Ducks, geese, and swans of North America, 2d ed. Stackpole Books, Harrisburg, PA. 543 p.

Bellrose, F. C. 1979. Species distribution, habitats, and characteristics of breeding dabbling ducks in North America. Pages 1-15 in T. A. Bookhout (ed.), Waterfowl and wetlands—An integrated review. Proc., 1977 Symp. N. Cent. Sect., Wildl. Soc., Madison, WI.

Bellrose, F. C., and N. M. Trudeau. 1988. Wetlands and their relationship to migrating and winter populations of waterfowl. Pages 183-194 in D. D. Hook, W. H. McKee, Jr., H. K. Smith, J. Gregory, V. G. Burrell, Jr., M. R. DeVoe, R. E. Sojka, S. Gilbert, R. Banks, L. H. Stolzy, D. Brooks, T. D. Matthews, and T. H. Shear (eds.), The Ecology and Management of Wetlands; Vol 1, Ecology of Wetlands. Croom Helm, London and Sydney.

Bendell, B. E., and D. K. McNicol. 1987. Fish predation, lake acidity and the composition of aquatic insect assemblages. Hydrobiol. 150:193-202.

Benke, A. C., D. M. Gillespie, F. K. Parrish, T. C. VanArsdall, Jr., R. J. Hunter, and R. L. Henry. 1979. Biological basis for assessing impacts of channel modification: Invertebrate production, drift, and fish feeding in a southeastern blackwater river. Envir. Resour. Cent. Publ. No. ERC 06-79. Georgia Inst. Technol., Atlanta. 187 p.

Benner, C. S., P. L. Knutson, R. A. Brochu, and A. R. Hurme. 1982. Vegetative erosion control in an oligohaline environment, Currituck Sound, North Carolina. Wetlands 2:105-117.

Benner, R., M. A. Moran, and R. E. Hodson. 1985. Effects of pH and plant source on lignocellulose biodegradation rates in two wetland ecosystems, the Okefenokee swamp and a Georgia salt marsh. Limnol. Oceanogr. 30:489-499.

Benson, N. G. 1953. The importance of ground water to trout populations in the Pigeon River, Michigan. Trans. N. Am. Wildl. Nat. Res. Conf. 18:269-281.

Benson, N. G., and P. L. Hudson. 1975. Effects of a reduced fall drawdown on benthos abundance in Lake Francis. Trans. Am. Fish. Soc. 104:526-528.

Berg, L., and T. G. Northcote. 1985. Changes in territorial, gill-flaring, and feeding behavior in juvenile coho salmon (*Oncorhynchus kisutch*) following short-term pulses of suspended sediment. Can. J. Fish. Aquat. Sci. 42:1410-1417.

Bergman, R. D. 1973. Use of southern boreal lakes by postbreeding canvasbacks and redheads. J. Wildl. Manage. 37:160-170.

Bernard, J. M., and B. A. Solsky. 1977. Nutrient cycling in a *Carex lacustris* wetland. Can. J. Bot. 55:630-638.

Bernard, J. M., D. Solander, and J. Kvet. 1988. Production and nutrient dynamics in *Carex* wetlands. Aquat. Bot. 30:125-147.

Bertani, A., I. Brambilla, and R. Reggiani. 1987. Effect of exogenous nitrate on anaerobic root metabolism. Pages 255-264 in R. M. M. Crawford, ed., Plant Life in Aquatic and Amphibious Habitats. Spec. Publ. 5, British Ecological Society. Blackwell Scientific Publ., Oxford.

Bertness, M. D., and A. M. Ellison. 1987. Determinants of pattern in a New England salt marsh plant community. Ecol. Monogr. 57:129-147.

Bertulli, J. A. 1982. Influence of a forested wetland on a Southern Ontario watershed. Pages 33-47 in A. Champagne (ed.), Proc. Ontario Wetlands Conf. Federation of Ontario Naturalists, Don Mills.

Bhomik, N. G., and M. Demissie. 1982. Carrying capacity of flood plains. Am. Soc. Civil Eng. J. Hydraul. 108:443-453.

Biddinger, G. R., and S. P. Gloss. 1984. The importance of trophic transfer in the bioaccumulation of chemical contaminants in aquatic ecosystems. Residue Reviews 91:103-145.

Bilby, R. E., and G. E. Likens. 1979. Effect of hydrologic fluctuations on the transport of fine particulate organic carbon in a small stream. Limnol. Oceanogr. 24:69-74.

Bilby, R. E. 1984. Characteristics and frequency of cool-water areas in a western Washington stream. J. Freshw. Ecol. 2:593-602.

Binns, N. A., and F. M. Eiserman. 1979. Quantification of fluvial trout habitat in Wyoming. Trans. Am. Fish. Soc. 108:215-228.

Birge, E. A., and C. Juday. 1934. Particulate and dissolved organic matter in inland lakes. Ecol. Mongr. 4:44-474.

Birmingham, B. C., and B. Colman. 1983. Potential phytotoxicity of diquat accumulated by aquatic plants and sediments. Water Air Soil Pollut. 19:123-131.

Bjorkland, R. G. 1975. On the death of a midwestern heronry. Wilson Bull. 87:284-287.

Blake, J. G., and J. R. Karr. 1984. Species composition of bird communities and the conservation benefit of large versus small forests. Biol. Conserv. 30:173-187.

Blake, J. G., and J. R. Karr. 1987. Breeding birds of isolated woodlots: Area and habitat relationships. Ecology 68:1724-1734.

Blancher, P. J., and D. K. McNicol. 1988. Breeding biology of tree swallows in relation to wetland acidity. Can. J. Zool. 66:842-849.

Blem, C. R., and L. B. Blem. 1975. Density, biomass, and energetics of the bird and mammal populations of an Illinois deciduous forest. Tran. Ill. Acad. Sci. 68:156-164.

Boecklen, W. J. 1986. Effects of habitat heterogeneity on the species-area relationships of forest birds. J. Biogeogr. 13:59-68.

Boelter, D. H., and E. S. Verry. 1977. Peatland and water in the northern lake states. US For. Serv. Tech. Rpt. NC-31. 22 p.

Boesch, D. F., and R. E. Turner. 1984. Dependence of fishery species on salt marshes: The role of food and refuges. Estuaries 7:460-468.

Bohlool, B. B., and W. J. Wiebe. 1978. Nitrogen-fixing communities in an intertidal ecosystem. Can. J. Microbiol. 24:932-938.

Bokuniewicz, H. 1980. Groundwater seepage into Great South Bay, NY. Est. Coast. Mar. Sci. 10:437-444.

Boon, J. D., III. 1975. *Tidal discharge* asymmetry in a salt marsh drainage system. Limnol. Oceanog. 20(1):71.

Borey, R. B., P. A. Harcombe, and F. M. Fisher. 1983. Water and organic carbon fluxes from an irregularly flooded brackish marsh on the upper Texas coast, USA. Est. Coast. Shelf Sci. 16:379-402.

Bormann, F. H., and G. E. Likens. 1979. Pattern and Process in a Forested Ecosystem: Disturbance, Development and the Steady State Based on the Hubbard Brook Ecosystem Study. Springer-Verlag, N.Y. 253 p.

Bormann, F. H., G. E. Likens, T. G. Siccama, R. S. Pierce, and J. S. Eaton. 1974. The export of nutrients and recovery of stable conditions following deforestation at Hubbard Brook. Ecol. Monogr. 44:255-277.

Born, S. M., S. A. Smith, and D. A. Stephenson. 1979. Hydrogeology of glacial-terrain lakes, with management and planning applications. J. Hydrol. 43:7-43.

Bostrom, U., and S. G. Nilsson. 1983. Latitudinal gradients and local variations in species richness and structure of bird communities on raised peat-bogs in Sweden. Ornis Scand. 14:213-226.

Boto, K. G., and W. H. Patrick, Jr. 1979. The role of wetlands in the removal of suspended sediments. Pages 479-489 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Bottorff, R. L. 1974. Cottonwood habitat for birds in Colorado. Am. Birds 28:975-979.

Bovee, K. D. 1978. Probability of use criteria for the family Salmonidae. IFIP No. 4, FWS/OBS-78/07. US Fish Wildl. Serv., Washington, DC. 90 p.

Bowden, W. B. 1986. Nitrification, nitrate reduction, and nitrogen immobilization in a tidal freshwater marsh sediment. Ecology 67:88-99.

Bowden, W. B. 1984. Nitrogen and phosphorus in the sediments of a tidal freshwater marsh in Massachusetts. Estuaries 7:108-118.

Bowen, S. H. 1980. Detrital non-protein amino acids are the key to rapid growth of Tilapia in Lake Valencia, Venezuela. Science 207:1216-1218.

Bowen, S. H. 1981. Digestion and assimilation of periphytic detrital aggregate by *Tilapia mossambica*. Trans. Am. Fish. Soc. 110:238-245.

Bowen, S. H. 1984. Evidence of a detritus food chain based on consumption of organic precipitates. Bull. Mar. Sci. 35:440-448.

Bowmer, K. H. 1987. Nutrient removal from effluents by an artificial wetland: Influence of rhizosphere aeration and preferential flow studied using bromide and dye tracers. Water Res. 21:591-599.

Boyd, C. E. 1971. The limnological role of aquatic macrophytes and their relationship to reservoir management. Pages 153-166 in G. E. Hall (ed.), Reservoir Fisheries and Limnology. Am. Fish. Soc. Spec. Pub. No. 8.

Boyd, C. E. 1978. Chemical composition of wetland plants. Pages 15-16 in R. E. Good, D. G. Whigham, and R. L. Simpson (eds.), Freshwater wetlands: Ecological processes and management potential. Academic Press, New York.

Boynton, W. R., W. M. Kemp, and G. G. Osborne. 1980. Nutrient fluxes across the sediment water interface in the turbid zone of a coastal plain estuary. Pages 93-109 in V. S. Kennedy (ed.), Estuarine Perspectives. Academic Press, New York.

Boyt, F. L., S. E. Bayley, and J. Zoltek, Jr. 1976. Removal of nutrients from treated municipal wastewater by wetland vegetation. Water Pollut. Cont. Fed. J. 49:789-799.

Brezonik, P. L., and G. F. Lee. 1968. Denitrification as a nitrogen sink in Lake Mendota. Wisc. Env. Sci. Tech. 1:120-125.

Bridge, J. S., and M. R. Leeder. 1979. A simulation model of alluvial stratigraphy. Sedimentology 26:617-644.

Briggs, K. T., D. B. Lewis, W. B. Tyler, and G. L. Hunt. 1981. Brown pelicans in southern California: Habitat use and environmental fluctuations. Condor 83:1-15.

Briggs, K. B., K. R. Tenore, and R. B. Hanson. 1979. The role of macrofauna in detrital utilization by the polychaete *Nereis succinea* (Frey and Leuckart). J. Exp. Mar. Biol. Ecol. 36:225-234.

Brinson, M. M. 1977. Decomposition and nutrient exchange of litter in an alluvial swamp forest. Ecology 58:601-609.

Brinson, M. M., and G. J. Davis. 1976. Primary productivity and mineral cycling in aquatic macrophyte communities of the Chowan River, North Carolina. Water Resour. Res. Inst. Rep. No. 120. Univ. of North Carolina, Raleigh. 137 p.

Brinson, M. M., A. E. Lugo, and S. Brown. 1981. Primary productivity, decomposition, and consumer activity in freshwater wetlands. Annu. Rev. Ecol. Syst. 12:123-161.

Brinson, M. M., H. D. Bradshaw, and R. N. Holmes. 1983. Significance of floodplain sediments in nutrient exchange between a stream and its flood plain. Pages 199-221 in T. D. Fontaine III and S. M. Bartell (eds.), Dynamics of Lotic Ecosystems. Ann Arbor Science, Ann Arbor, MI.

Brinson, M. M., H. D. Bradshaw, and E. S. Kane. 1984. Nutrient assimlative capacity of an alluvial floodplain swamp. J. Appl. Ecol. 21:1041-1057.

Brinson, M., B. Swift, R. Plantico, and J. Barclay. 1981. Riparian ecosystems: Their ecology and status. FWS/OBS-81/17. US Fish Wildl. Serv., Washington, DC. 151 p.

Broadfoot, W. M., and H. L. Williston. 1973. Flooding effects on southern forests. J. For. 71:584-587.

Brock, T. C. M. 1984. Aspects of the decomposition of *Nymphoides peltata* (Menyanthaceae). Aquat. Bot. 19:131-156.

Brodrick, S. J., P. Cullen, and W. Maher. 1988. Denitrification in a natural wetland receiving secondary treated effluent. Wat. Res. 22:431-439.

Bronmark, C., J. Herrmann, B. Malmqvist, C. Otto, and P. Sjostrom. 1984. Animal community structure as a function of stream size. Hydrobiol. 112:73-79.

Broome, S. W., W. W. Woodhouse, and E. D. Seneca. 1975. The relationship of mineral nutrients on growth of *S. alterniflora* in North Carolina; II. The effects of nitrogen, phosphorous, and iron fertilizers. Soil Sci. Soc. Am. Proc. 39:301-307.

Brown, J. M. 1975. Geographical ecology of desert rodents. Pages 315-341 in M. L. Cody and J. M. Diamond (eds.), Ecology and evolution of communities. Belknap Press, Cambridge, MA. 545 p.

Brown, M., and J. J. Dinsmore. 1986. Implications of marsh size and isolation for marsh bird management. J. Wildl. Manage. 50:392-397.

Brown, M., and J. J. Dinsmore. 1988. Habitat islands and the equilibrium theory of island biogeography: Testing some predictions. Oecologia 75:426-429.

Brown, M. T., and M. F. Sullivan. 1988. The value of wetlands in low relief landscapes. Pages 133-145 in D. D. Hook, W. H. McKee, Jr., H. K. Smith, J. Gregory, V. G. Burrell, Jr., M. R. DeVoe, R. E. Sojka, S. Gilbert, R. Banks, L. H. Stolzy, D. Brooks, T. D. Matthews, and T. H. Shear (eds.), The Ecology and Management of Wetlands; Vol 1, Ecology of Wetlands. Croom Helm, London and Sydney.

Brown, R. G. 1985. Effects of wetlands on quality of runoff entering lakes in the twin cities metropolitan area, Minnesota. Water-Resources Investig. Rep. 85-4170. US Geological Survey, Reston, VA. 32 p.

Brown, R. G. 1988. Effects of precipitation and land use on storm runoff. Water Resour. Bull. 24:421-426.

Brown, S. 1981. A comparison of the structure, primary productivity and transpiration of cypress ecosystems in Florida. Ecol. Monogr. 551:403-427.

Brown, S., and A. E. Lugo. 1982. A comparison of structural and functional characteristics of saltwater and freshwater forested wetlands. Pages 109-130 in B. Gopal, R. E. Turner, R. G. Wetzel, and D. F. Whigham, (eds.), Wetlands Ecology and Management. National Institute of Ecology and International Scientific Publications, Jaipur, India. 514 p.

Brown, S. L., and D. L. Peterson. 1983. Structural characteristics and biomass production of two Illinois bottomland forests. Am. Midl. Natur. 110:107-117.

Brown, S., M. M. Brinson, and A. E. Lugo. 1979. Structure and function of riparian wetlands. Pages 17-31 in R. R. Johnson and J. F. McCormick (tech. coords.), Strategies for protection and management of floodplain wetlands and other riparian ecosystems. Gen. Tech. Rep. WO-12. US For. Serv., Washington, DC.

Brun, L. J., J. L. Richardson, J. W. Enz, and J. K. Larsen. 1981. Stream flow changes in the southern Red River Valley of North Dakota. North Dakota Farm Res. 38(5):11-14.

Buchsbaum, R., I. Valiela, and J. M. Teal. 1981. Grazing by Canada geese and related aspects of the chemistry of salt marsh grasses. Col. Waterbirds 4:126-131.

Buchsbaum, R., J. Wilson, and I. Valiela. 1986. Digestibility of plant constituents by Canada geese and Atlantic Brant. Ecology 67:386-393.

Buckley, F. G., and P. A. Buckley. 1980. Habitat selection and marine birds. Pages 69-113 in J. Burger, B. L. Olla, and H. E. Winn (eds.), Behavior of Marine Animals; Vol 4, Marine Birds. Plenum Press, New York.

Bulkley, R. V., R. W. Bachmann, K. D. Carlander, H. L. Fierstine, L. R. King, B. W. Menzel, A. L. Witten, and D. W. Zimmer. 1976. Warmwater stream alteration in Iowa: Extent, effects on habitat, fish, and fish food, and evaluation of stream improvement structures. FWS/OBS-76/16. US Fish Wildl. Serv., Washington, DC. 39 p.

Bulthuis, D. A., and W. J. Woelkerling. 1983. Biomass accumulation and shading effects of epiphytes of the seagrass *Heterozostera tasmanica*. Aquat. Bot. 16:137-148.

Bulthuis, D. A., G. W. Brand, and M. C. Mobley. 1984. Suspended sediments and nutrients in water ebbing from seagrass-covered and denuded tidal mudflats in a southern Australian embayment. Aquat. Bot. 20:257-266.

Buresh, R. J., and W. H. Patrick, Jr. 1981. Nitrate reduction to ammonium and organic nitrogen in an estuarine sediment. Soil Biol. Biochem. 13:279-283.

Burger, J. 1981. The effect of human activity on birds at a coastal bay. Biol. Conserv. 32:231-241.

Burger, J., J. Shisler, and F. Lesser. 1982. Avian utilization in six salt marshes in New Jersey. Biol. Conser. 23:187-212.

Burkham, D. E. 1976. Hydraulic effects of changes in bottomland vegetation on three major floods, Gila River in southeastern Arizona. Paper 655-J. US Geological Survey, Reston, VA. 14 p.

Burns, J. W. 1972. Some effects of logging and associated road construction on northern California streams. Trans. Am. Fish Soc. 101:1-17.

Burns, L. A. 1978. Productivity, biomass, and water relations in a Florida cypress forest. Ph.D. dissertation. Univ. of North Carolina, Chapel Hill. 170 p.

Burns, L. A. 1985. Models for predicting the fate of synthetic chemicals in aquatic ecosystems. Pages 176-190 in T. P. Boyle (ed.), Validation and Predictability of Laboratory Methods for Assessing the Fate and Effects of Contaminants in Aquatic Ecosystems, ASTM STP 865. American Society for Testing and Materials, Philadelphia, PA.

Burris, R. H. 1976. Nitrogen fixation by blue-green algae of the Lizard Island area of the Great Barrier Reef. Aust. J. Plant Physiol. 3:15-51.

Burton, T. M., R. C. Harris, M. Tripp, and D. Taylor. 1979. The influence of bird rookeries on nutrient cycling and organic matter production in the Shark River, Florida Everglades. Pages 73-79 in R. R. Johnson and J. F. M. McCormick (tech. coords.), Strategies for protection and management of floodplain wetlands and other riparian ecosystems. Gen. Tech. Rep. WO-12. US For. Serv., Washington, DC.

Bustard, D. R., and D. W. Narver. 1975. Aspects of the winter ecology of juvenile coho salmon and steelhead trout. J. Fish. Res. Bd. Can. 32:667-680.

Callender, E. 1982. Benthic phosphorus regeneration in the Potomac River Estuary. Hydrobiol. 92:431-446.

Callender, E., and D. E. Hammond. 1982. Nutrient exchange across sediment-water interface in the Potomac River Estuary. Est. Coastal Sci. 15:395-413.

Cameron, G. N. 1972. Analysis of insect trophic diversity in two salt marsh communities. Ecology 53:58-73.

Camfield, F. E. 1977. Wind-wave propagation over flooded, vegetated land. Tech. Pap. No. 77-12. US Army Engineer Coast. Eng. Res. Cent., Fort Belvoir, VA. 42 p.

Campbell, L. H. 1984. The impact of changes in sewage treatment on seaducks wintering in the Firth of Forth, Scotland. Biol. Conserv. 28:173-180.

Campbell, K. L., and H. P. Johnson. 1975. Hydrologic simulation of watersheds with artificial drainage. Water Resour. Res. 11:120-126.

Canfield, D. E., Jr., K. A. Langeland, M. J. Maceina, W. T. Haller, J. V. Shireman, and J. R. Jones. 1983. Trophic state classification of lakes with aquatic macrophytes. Can. J. Fish. Aquat. Sci. 40:1713-1718.

Capone, D. G. 1982. Nitrogen fixation (acetylene reduction) by rhizosphere sediments of the eelgrass, *Zostera marina*. Mar. Ecol. Prog. Ser. 10:67-75.

Capone, D. G., and B. F. Taylor. 1980. Microbial nitrogen cycling in a seagrass community. Pages 153-161 in V. S. Kennedy (ed.), Estuarine Perspectives. Academic Press, New York.

Carlander, K. D., J. S. Campbell, and R. J. Muncy. 1978. Inventory of percid and esocid habitat in North America. Am. Fish. Soc. Spec. Publ. 11:27-38.

Carothers, S. W., R. R. Johnson, and S. W. Aitchison. 1974. Population structure and social organization of Southwestern riparian birds. Am. Zool. 14:97-108.

Carpenter, E. J., C. D. Van Raalte, and I. Valiela. 1978. Nitrogen fixation by algae in a Massachusetts salt marsh. Limnol. Oceanogr. 23:318-327.

Carpenter, S. R. 1980. Enrichment of Lake Wingra, Wisconsin, by submersed macrphyte decay. Ecology 61:1145-1155.

Carpenter, S. R., and M. S. Adams. 1979. Effects of nutrients and temperature on decomposition of *Myriophyllum spicatum* in a hardwater eutrophic lake. Limnol. Oceanogr. 24:520-528.

Carpenter, S. R., and J. K. Greenlee. 1981. Lake deoxygenation after herbicide use: A simulation model analysis. Aquat. Bot. 11:173-186.

Carpenter, S. R., and D. M. Lodge. 1986. Effects of submersed macrophytes on ecosystem processes. Aquat. Bot. 26:341-370.

Carper, G. L., and R. W. Bachman. 1984. Wind resuspension of sediments in a prairie lake. Can. J. Fish. Aquat. Sci. 41:1763-1767.

Carter, V., and R. P. Novitzki. 1988. Some comments on the relation between ground water and wetlands. Pages 68-86 in D. D. Hook, W. H. McKee, Jr., H. K. Smith, J. Gregory, V. G. Burrell, Jr., M. R. DeVoe, R. E. Sojka, S. Gilbert, R. Banks, L. H. Stolzy, D. Brooks, T. D. Matthews, and T. H. Shear (eds.), The Ecology and Management of Wetlands; Vol 1, Ecology of Wetlands. Croom Helm, London and Sydney.

Carter, V., M. S. Bedinger, R. P. Novitzki, and W. O. Wilen. 1979. Water resources and wetlands. Pages 344-376 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Casey, H., and I. S. Farr. 1982. The influence of within-stream disturbance on dissolved nutrient levels during spates. Hydrobiol. 92:447-462.

Casselman, M. E., W. H. Patrick, Jr., and R. D. DeLaune. 1981. Nitrogen fixation in a Gulf coast salt marsh. Soil Sci. Soc. Am. J. 45:51-56.

Cattaneo, A., and J. Kalff. 1980. The relative contribution of aquatic macrophytes and their epiphytes to the production of macrophyte beds. Limnol. Oceanogr. 25:280-289.

Cernohous, L. 1979. The role of wetlands in providing flood control benefits. US Fish Wildl. Serv., Bismark, ND. 7 p.

Chabreck, R. H., R. K. Yancy, and L. McNease. 1975. Duck usage of management units in the Louisiana coastal marsh. Proc. S. E. Assoc. Game Fish Comm. 38:507-516.

Chabreck, R. H. 1979. Winter habitat of dabbling ducks—physical, chemical, and biological aspects. Pages 133-142 in T. A. Bookout (ed.), Waterfowl and wetlands—An integrated review. Wildl. Soc. N. Cent. Sect., Madison, WI.

Chabreck, R. H. 1979. Wildlife harvest in wetlands of the United States. Pages 618-631 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Chamberlain, T. W. 1982. Timber Harvest. Part 3 of W. R. Meehan (ed.), Influence of forest and rangeland management on anadromous fish habitat in western North America. GTR-PNW-136. USDA Forest Service, Portland, OR.

Chamie, J. P. M., and C. J. Richardson. 1978. Decomposition in northern wetlands. Pages 115-130 in R. E. Good, D. F. Whigham, and R. L. Simpson (eds.), Freshwater Wetlands. Academic Press, New York.

Chaney, A. H., B. R. Chapman, J. P. Karges, D. A. Nelson, R. R. Schmidt, and L. C. Thebeau. 1978. Use of dredged material islands by colonial seabirds and wading birds in Texas. Tech. Rep. D-78-8. US Army Engineer Waterways Exp. Sta., Vicksburg, MS. 170 p.

Chang, M., and S. P. Watters. 1984. Forests and other factors associated with streamflows in east Texas. Water Resour. Bull. 20:713-719.

Chang, M., J. D. McCullough, and A. B. Granillo. 1983. Effects of land use and topography on some water quality variables in forested east Texas. Water Resour. Bull. 19:191-196.

Chapman, D. W. 1966. The relative contributions of aquatic and terrestrial primary producers to the trophic relations of stream organisms. Pages 116-130 in Organism-substrate relationships in streams. Pymatuning Lab. Ecol. Spec. Publ. No. 4. Univ. Pittsburgh, Pittsburgh, PA.

Chapman, R. R., and H. F. Hemond. 1982. Denitrogen fixation by surface peat and *Sphagnum* in an ombrotrophic bog. Can. J. Bot. 60:538-543.

Chapman, V. J. 1976. Coastal Vegetation. Pergamon Press, New York. 292 p.

Chapman, V. J. (ed.). 1977. Ecosystems of the World; Vol 1, Wet Coastal Ecosystems. Elsevier Scientific Publishing, New York. 428 p.

Chen, R. L., and J. W. Barko. 1988. Effects of freshwater macrophytes on sediment chemistry. J. Freshw. Ecol. 4:279-289.

Chescheir, G. M., J. W. Gilliam, R. W. Skaggs, and R. G. Broadhead. 1987. The hydrology and pollutant removal effectiveness of wetland buffer areas receiving pumped agricultural drainage water. WRRI Project No. 70028/70029. North Carolina Water Resour. Res. Inst., Raleigh. 170 p.

Chow, V. T. 1959. Open-Channel Hydraulics. McGraw-Hill, New York. 680 p.

Christiansen, L. E., and J. B. Low. 1970. Water requirements of waterfowl marshlands in northern Utah. Utah Div. Fish Game Publ. 69-12. 108 p.

Chrzanowski, T. H., and J. D. Spurrier. 1987. Exchange of microbial biomass between a *Spartina alterniflora* marsh and the adjacent tidal creek. Estuaries 10:118-125.

Chubb, S. L., and C. R. Liston. 1986. Density and distribution of larval fishes in Pentwater Marsh, a coastal wetland on Lake Michigan. J. Great Lakes Res. 12:332-343.

Chui, T. W., B. W. Mar, and R. R. Horner. 1982. Pollutant loading model for highway runoff. J. Environ. Eng. Div., ASCE 108(EE6):1193-1210.

Chutter, F. M. 1969. The effects of silt and sand on the invertebrate fauna of streams and rivers. Hydrobiol. 34:57-76.

Clark, J. R., and J. A. Zinn. 1978. Cumulative effects in environmental assessment. Pages 2481-2492 in Coastal Zone '78 Proc., ASCE, New York, NY.

Clark, J. R., and J. E. Clark. 1979. Scientists' report: The national symposium on wetlands, Lake Buena Vista, FL. Nat. Wetlands Tech. Council, The Conservation Foundation, Washington, DC. 128 p.

Clarke, J. A., B. A. Harrington, T. Hruby, and F. E. Wasserman. 1984. The effect of ditching for mosquito control on salt marsh use by birds in Rowley, Massachusetts. J. Field. Ornithol. 55:160-180.

Clymo, R. S. 1983. Peat. Pages 159-224 in A. J. P. Gore (ed.), Ecosystems of the World; Vol 4A, Mires: Swamp, bog, fen and moor. Elsevier Scientific, Amsterdam, The Netherlands.

Coates, V. T. 1981. Methodology for the analysis of cumulative impacts of permit activities regulated by the US Army Corps of Engineers. US Army Corps of Eng. Inst. Water Resour., Fort Belvoir, VA.

Cobb, S. P., C. H. Pennington, J. A. Baker, and J. E. Scott. 1984. Fishery and ecological investigations of main stem levee borrow pits along the Lower Mississippi River, Report 1; Lower Mississippi River Environmental Program. Mississippi River Commission, Vicksburg, MS. 120 p.

Cole, D. N., and J. L. Marion. 1988. Recreation impacts in some riparian forests of the eastern United States. Envir. Manage. 12:99-107.

Cole, G. A. 1975. Textbook of Limnology. C. V. Mosby, St. Louis, MO. 283 p.

Colle, D. E., and J. V. Shireman. 1980. Coefficients of condition for largemouth bass, bluegill, and red-ear sunfish in *Hydrilla*-infested lakes. Trans. Am. Fish. Soc. 109:521-531.

Collins, J. O. 1961. Ten year acorn mast production study in Louisiana. PR Rep. W-29-R-8. Louisiana Wildl. Fish. Comm. 33 p.

Conger, D. H. 1971. Estimating magnitude and frequency of floods in Wisconsin. Open-File Rep. US Geol. Surv., Water Resour. Div., Madison, WI. 200 p.

Conine, K. H., B. W. Anderson, R. D. Ohmart, and J. F. Drake. 1978. Responses of riparian species to agricultural habitat conversions. Pages 248-263 in R. R. Johnson and J. F. McCormick (tech. coords.), Strategies for Protection and Management of Floodplain, Wetlands and Other Riparian Ecosystems. Gen. Tech. Rep. WO-12, US For. Serv., Washington, DC.

Conner, W. H., and T. W. Day. 1976. Productivity and composition of a baldcypress-water tupelo site and bottomland hardwood site in Louisiana swamp. Am. J. Bot. 63:1354-1364.

Conroy, M. J., G. R. Costanzo, and D. B. Stotts. 1987. Winter movements of American black ducks in relation to natural and impounded wetlands in New Jersey. Pages 31-44 in W. R. Whitman and W. H. Meredith (eds.), Proc. of a Symp. on Waterfowl and Wetland Management in the Coastal Zone of the Atlantic Flyway. Delaware Dep. Nat. Resour. and Environ. Cont., Dover.

Cook, A. H., and C. F. Powers. 1958. Early biochemical changes in the soils and waters of artificially created marshes in New York. NY Fish Game Jour.:9-65.

Cooke, G. D., M. McComas, T. N. Bhargava, and R. Heath. 1973. Monitoring and nutrient inactivation in two glacial lakes (Ohio) before and after nutrient diversion. Cent. for Urban Reg. Interim Res. Rep., Kent State Univ., Kent, OH. 92 p.

Cooper, C. M. 1987. Benthos in Bear Creek, Mississippi: Effects of habitat variation and agricultural sediments. J. Freshw. Ecol. 4:101-113.

Cooper, J. R., J. W. Gilliam, and T. C. Jacobs. 1986. Riparian areas as a control of nonpoint pollutants. Pages 166-192 in D. L. Correll (ed.), Watershed Research Perspectives. Smithsonian Institution Press, Washington, DC.

Cope, O. B., E. M. Woodard, and G. H. Wallen. 1970. Some chronic effects of 2,4-D on the bluegill (*Lepomis macrochirus*). Trans. Am. Fish. Soc. 99:1-12.

Copeland, B. J., and T. Dickens. 1969. Systems resulting from dredging spoil. Pages 1084-1100 in H. T. Odum, B. J. Copeland, and E. A. McMahan (eds.), Coastal Ecosytstems of the United States. The Conservation Foundation, Washington, DC.

Copeland, B. J., and S. W. Nixon. 1974. Hypersaline lagoons. Pages 312-330 in H. T. Odum, B. J. Copeland, and E. A. McMahan (eds.), Coastal Ecosystems of the United States. The Conservation Foundation, Washington, DC.

Correll, D. L. 1978. Estuarine productivity. BioScience 28:646-650.

Correll, D. W., and D. Ford. 1982. Comparison of precipitation and land runoff as sources of estuarine nitrogen. Est. Coast. Shelf Sci. 15:45-56.

Cosby, B. J., G. M. Hornberger, and M. G. Kelly. 1984. Identification of photosynthesis-light models for aquatic systems; II, Application to a macrophyte dominated stream. Ecol. Model. 23:25-51.

Costa, J. E. 1977. Sediment concentration and duration in stream channels. J. Soil Water Conserv. 32:168-170.

Cottam, C., J. L. Lynch, and A. L. Nelson. 1944. Food habits and management of American sea brant. J. Wildl. Manage. 8:36-56.

Coulter, M. W., and W. R. Miller. 1968. Nesting biology of black ducks and mallards in northern New England. Vermont Fish Game Dep. Bull. No. 68-2. 74 p.

Cowardin, L. M. 1969. Use of flooded timber by waterfowl at the Montezuma National Wildlife Refuge. J. Wildl. Manage. 33:829-842.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. FWS/OBS-79/31. US Fish Wildl. Serv., Washington, DC. 193 p.

Craig, N. J., R. E. Turner, and J. W. Day, Jr. 1980. Wetland losses and their consequences in coastal Louisiana. Zeitschrift Geomorph. N.F. 34(Supp.):225-241.

Craighead, F. C., Jr., and J. J. Craighead. 1949. Nesting Canada geese on the upper Snake River. J. Wildl. Manage. 13:51-64.

Cranford, P. J., P. Schwinghamer, and D. C. Gordon, Jr. 1987. Identification of microdetritus derived from *Spartina* and its occurrence in the water column and intertidal sediments of Cumberland Basin, Bay of Fundy. Estuaries 10:108-117.

Croft, A. R., and L. V. Monninger. 1953. Evapotransporative and other water losses in some aspen forest types in relation to water available for stream flow. Trans. Am. Geophys. Union 34:563-574.

Cronan, J. M., and B. F. Halla. 1968. Fall and winter foods of Rhode Island waterfowl. Pam. No. 7. Rhode Island Dep. Nat. Resour., Div. Conserv. Wildl. 39 p.

Cross, R. D., and D. L. Williams (eds.). 1981. Proc., national symposium on freshwater inflow to estuaries, 3 vols. FWS/OBS-81/04. US Fish Wildl. Serv., Washington, DC. 1053 p.

Crow, J. H., and K. B. MacDonald. 1978. Wetland values: Secondary productivity. Pages 146-161 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Crowder, A. A., and J. M. Bristow. 1988. The future of waterfowl habitats in the Canadian lower Great Lakes wetlands. J. Great Lakes Res. 14:115-127.

Crowder, L. B., and W. E. Cooper. 1982. Habitat structural complexity and the interaction between bluegills and their prey. Ecology 63:1802-1813.

Cudlip, L. S., and J. A. Perry. 1988. Is in-lake carbon processing phased to correlate with availability? Decomposition of *Decodon verticillatus* (L.) Ell. and *Ceratophyllum demersum* in Cedar Bog Lake, Minnesota, USA. Arch. Hydrobiol. 111:383-396.

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

Culp, J. M., and R. W. Davies. 1985. Responses of benthic macroinvertebrate species to manipulation of interstitial detritus in Carnation Creek, British Columbia. Can. J. Fish. Aquat. Sci. 42:139-146.

Cummins, K. W. 1974. Structure and function of stream ecosystems. BioScience 24:631-641.

Cushing, D. E. 1964. Plankton and water chemistry in the Montreal River lake-stream system, Saskatchewan. Ecology 45:306-313.

Cushman, R. M. 1985. Review of ecological effects of rapidly varying flows downstream from hydroelectric facilities. N. Am. J. Fish. Manage. 5:330-339.

Cyr, H., and J. A. Downing. 1988. Empirical relationships of phytomacrofaunal abundance to plant biomass and macrophyte bed characteristics. Can. J. Fish. Aquat. Sci. 45:976-984.

Cyrus, D. P., and S. J. M. Blaber. 1987. The influence of turbidity on juvenile marine fishes in estuaries; Part 1, Field studies at Lake Lucia on the southeastern coast of Africa. J. Exp. Mar. Biol. Ecol. 109:53-70.

Cyrus, D. P., and S. J. M. Blaber. 19[illegible] The influence of turbidity on juvenile marine fishes in estuaries; Part 2, Laboratory studies, comparisons with field data and conclusions. J. Exp. Mar. Biol. Ecol. 109:71-91.

Dahm, C. N., E. H. Trotter, and J. R. Sedell. 1987. Role of anaerobic zones and processes in stream ecosystem productivity. Pages 157-178 in R. C. Averett and D. M. McKnight (eds.), Chemical Quality of Water and the Hydrologic Cycle. Lewis Publishers, Chelsea, MI.

Daniel, J. F. 1971. Channel movement of meandering Indiana streams. Pages A1-A18 in Prof. Paper No. 732-A. US Geological Survey, Reston, VA.

Daniel, J. F. 1976. Estimating groundwater evapotranspiration from streamflow records. Water Resour. Res. 12:360-364.

Darnell, R. M. 1961. Trophic spectrum of an estuarine community, based on studies of Lake Ponchartrain, Louisiana. Ecology 42:553-568.

Darnell, R. M. 1967. Organic detritus in relation to the estuarine ecosystem. Pages 376-382 in G. H. Lauff (ed.), Estuaries. Am. Assoc. Adv. Sci. Publ. No. 83, Washington, DC.

Darnell, R. M., W. E. Pequegnat, B. M. James, F. J. Benson, and R. E. Defenbaugh. 1976. Impacts on construction activities in wetlands of the United States. EPA-600/3-76-45, Ecol. Res. Ser. US Environmental Protection Agency, Washington, DC. 396 p.

Davis, C. B., and A. G. van der Valk. 1978. Litter decomposition in prairie glacial marshes. Pages 99-114 in R. E. Good, D. F. Whigham, and R. L. Simpson (eds.), Freshwater Wetlands. Academic Press, New York.

Davis, G. J., and M. M. Brinson. 1980. Responses of submersed vascular plant communities to environmental change. FWS/OBS-79/33. US Fish Wildl. Serv., Washington, DC. 79 p.

Dawe, N. K., and B. D. Davis. 1975. A nesting study of Canada geese on the George C. Reifel Migratory Bird Sanctuary, British Columbia. Syesis 8:1-7.

Dawson, F. H. 1980. The origin, composition and downstream transport of plant material in a small chalk stream. Freshw. Biol. 19:419-435.

Day, D. S., and W. G. Pearcy. 1968. Some associations of benthic fishes on the continental shelf and slope off Oregon. J. Fish. Res. Bd. Can. 25:2665-2675.

Day, J. W., Jr., W. G. Smith, P. R. Wagner, and W. C. Stowe. 1973. Community structure and carbon budget of a salt marsh and shallow bay estuarine system in Louisiana. Center Wetland Resour., Louisiana State Univ., Baton Rouge. 79 p.

Day, J. W., Jr., W. H. Conner, and G. M. Kemp. 1980. Contribution of wooded swamps and bottomland forests to estuarine productivity. Pages 33-50 in P. L. Fore and R. D. Peterson (eds.), Proceedings, Gulf of Mexico Coastal Ecosystems Workshop. FWS/OBS-80/30. US Fish Wildl. Serv., Washington, DC.

Day, J. W., Jr., W. H. Conner, G. P. Kemp, and D. G. Chambers. 1981. The relationship of estuarine productivity to wooded swamps and bottomland forests in the southeastern United States. Pages 193-213 in R. C. Carey, P. S. Markovits, and J. B. Kirkwood (eds.), Proc., Coastal Ecosystems of the Southeastern United States. FWS/OBS-80/59. US Fish Wildl. Serv., Washington, DC.

Day, R. T., R. A. Keddy, J. McNeill, and T. Carleton. 1988. Fertility and disturbance gradients: A summary model for riverine marsh vegetation. Ecology 69:1044-1054.

De Jong, J. 1976. The purification of wastewater with the aid of rush or reed ponds. Pages 133-139 in J. Tourbier and R. W. Pierson, Jr. (eds.), Biological Control of Water Pollution. Univ. Pennsylvania Press, Philadelphia.

de la Cruz, A. A. 1979. Production and transport of detritus in wetlands. Pages 162-174 in P. E. Greeson, J. R. Clark, and J. S. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

de la Cruz, A. A. 1980. Recent advances in our understanding of salt marsh ecology. Pages 51-66 in P. L. Fore and R. D. Peterson (eds.), Proc., Gulf of Mexico Coastal Ecosystems Workshop. FWS/OBS-80/30. US Fish Wildl. Serv., Washington, DC.

de la Cruz. A. A. 1975. Proximate nutritive value changes during decomposition of salt marsh plants. Hydrobiol. 57:475-480.

Dean, R. G. 1979. Effects of vegetation on shoreline erosional processes. Pages 415-426 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

DeBoer, D., and H. Johnson. 1971. Simulation of runoff from depression characterized watersheds. Trans. Am. Soc. Agr. Eng. 14:615-620.

DeBusk, W. F., and K. R. Reddy. 1987. Removal of floodwater nitrogen in a cypress swamp receiving primary wastewater effluent. Hydrobiol. 153:79-86.

DeLaune, R. D., W. H. Patrick, Jr., and J. M. Brannon. 1976. Nutrient transformation in Louisiana salt marsh soils. Sea Grant Publ. No. LSU-T-76-009. Louisiana State Univ., Center Wetland Resour., Baton Rouge.

DeLaune, R. D., W. H. Patrick, Jr., and R. J. Buresh. 1978. Sedimentation rates determined by ^{137}Cs dating a rapidly accreting salt marsh. Nature 175:532-533.

DeLaune, R. D., and W. H. Patrick, Jr. 1980. Nitrogen and phosphorous cycling in a Gulf Coast salt marsh. Pages 143-151 in V. S. Kennedy (ed.), Estuarine Perspectives. Academic Press, New York.

DeLaune, R. D., C. N. Reddy, and W. H. Patrick, Jr. 1981. Effect of pH and redox potential on concentration of dissolved nutrients in an estuarine sediment. J. Environ. Qual. 10:276-278.

DeLaune, R. D., C. J. Smith, and M. N. Sarafyan. 1986. Nitrogen cycling in a freshwater marsh of *Panicum hemitomon* on the deltaic plain of the Mississippi River. J. Ecol. 74:249-256.

Delucchi, C. M. 1988. Comparison of community structure among streams with different temporal flow regimes. Can. J. Zool. 66:579-586.

Demers, S., J. C. Therriault, E. Bourget, and A. Bah. 1987. Resuspension in the shallow sublittoral zone of a macrotidal estuarine environment: Wind influence. Limnol. Oceanogr. 32:327-339.

Dendy, F. E. 1974. Sediment trap efficiency of small reservoirs. Trans. A.S.A.E. 17:898-901.

Dendy, F. E., and G. C. Bolton. 1976. Sediment yield-runoff-drainage area relationships in the United States. J. Soil Water Conserv. (Nov/Dec):264-266.

Dennison, W. C., and R. S. Alberte. 1985. Role of daily light period in depth distribution of *Zostera marina* L. (eelgrass). Mar. Ecol. Prog. Ser. 25:51-61.

Dewey and Kropper Engineers. 1964. Effect of loss of valley storage due to encroachment—Connecticut River. Connecticut Water Resources Comm., Hartford. 15 p.

Diaz, R. J., R. J. Orth, G. Markwith, W. Rizzo, R. Wetzel, and K. Storey. 1982. Examination of tidal flats; Vol 2, A review of unidentified values. FHWA/RD-80/181. US Dept. Transportation, Washington, DC.

Dickman, C. R. 1987. Habitat fragmentation and vertebrate species richness in an urban environment. J. Appl. Ecol. 24:337-351.

Dickson, J. G. 1978. Forest bird communities of the bottomland hardwoods. Pages 66-73 in R. M. DeGraaf (tech. coord.), Proc. of the Workshop on Management of Southern Forests for Nongame Birds. Gen. Tech. Rep. WE-14. US For. Serv., Atlanta, GA.

Diefenbach, D. R., J. D. Nichols, and J. E. Hines. 1988. Distribution patterns during winter and fidelity to wintering areas of American black ducks. Can. J. Zool. 66:1506-1513.

Dierberg, F. E., and P. L. Brezonik. 1978. The effect of secondary sewage effluent on the surface water and ground water quality of cypress domes. Pages 178-270 in H. T. Odum and K. C. Ewel (eds.), Cypress wetlands for water management, recycling and conservation. Center for Wetlands, Gainesville, FL.

Dierberg, F. E., and P. L. Brezonik. 1981. Nitrogen fixation (acetylene reduction) associated with decaying leaves of pond cypress in a natural and sewage-enriched cypress dome. Appl. Environ. Microbiol. 41:1413-1418.

Dierberg, F. E., and P. L. Brezonik. 1983. Tertiary treatment of municipal wastewater by cypress domes. Water Res. 17:1027-1040.

Dillon, P. J., and F. H. Rigler. 1975. A simple method for predicting the capacity of a lake for development. J. Fish Res. Bd. Can. 32:1519-1531.

Dodd, C. K., Jr. 1978. Amphibians and reptiles: The declining species. Water Spectrum 10:24-32.

Dolan, T. J., A. J. Hermann, S. E. Bayley, and J. Zoltek, Jr. 1984. Evapotranspiration of a Florida, USA, fresh water wetland. J. Hydrol. 74:355-371.

Dolan, T. J., S. E. Bayley, J. Zolteck, and A. Herman. 1978. The Clermont project: Renovation of treated effluent by a freshwater marsh. Pages 132-152 in M. A. Drew (ed.), Environmental quality through wetlands utilization. Coord. Counc. Kissimmee R. Val. and Taylor Cr.-Nubbin Slough Basin, Tallahassee, FL.

Doremus, C., and L. S. Cleseri. 1982. Microbial metabolism in the surface sediments and its role in the immobilization of phosphorus in oligotrophic lake sediments. Hydrobiol. 91:261-268.

Driver, E. A. 1977. Chironomid communities in small prairie ponds: Some characteristics and controls. Freshw. Biol. 7:121-133.

Duarte, C. M., and J. Kalff. 1986. Littoral slope as a predictor of the maximum biomass of submerged macrophyte communities. Limnol. Oceanogr. 31:1072-1080.

Duarte, C. M., and J. Kalff. 1988. Influence of lake morphometry on the response of submerged macrophytes to sediment fertilization. Can. J. Fish. Aquat. Sci. 45:216-221.

Duarte, C. M., J. Kalff, and R. H. Peters. 1986. Patterns in biomass and cover of aquatic macrophytes in lakes. Can. J. Fish. Aquat. Sci. 43:1900-1908.

Dubinski, B. J., R. L. Simpson and R. E. Good. 1986. The retention of heavy metals in sewage sludge applied to a freshwater tidal wetland. Estuaries 9:102-111.

Duda, A. M. 1982. Municipal point source and agricultural nonpoint source contributions to coastal eutrophication. Water Resour. Bull. 18:397-406.

Dudley, T. L., S. D. Cooper, and N. Hemphill. 1986. Effects of macroalgae on a stream invertebrate community. J. N. Am. Benthol. Soc. 5:93-106.

Duebbert, H. F., and J. T. Lokemoen. 1976. Duck nesting in fields of undisturbed grass-legume cover. J. Wildl. Manage. 40:39-49.

Duebbert, H. F., J. T. Lokemoen, and D. E. Sharp. 1983. Concentrated nesting of mallards and gadwalls on Miller Lake Island, North Dakota. J. Wildl. Manage. 47:2729-740.

Duebbert, H. F., and A. M. Frank. 1984. Value of prairie wetlands to duck broods. Wildl. Soc. Bull. 12:27-34.

Dunbabin, J. S., J. Pokorny, and K. H. Bowmer. 1988. Rhizosphere oxygenation by *Typha domingensis* Pers. in miniature artificial wetland filters used for metal removal from wastewaters. Aquat. Bot. 29:303-317.

Dunne, T., and L. B. Leopold 1978. Water in Environmental Plannin. W. H. Freeman and Co., San Francisco. 818 p.

Durocher, P. P., W. C. Provine, and J. E. Kraai. 1984. Relationship between abundance of largemouth bass and submerged vegetation in Texas Reservoirs. N. Am. J. Fish. Manage. 4:84-88.

Dvorak, J., and Best, E. P. H. 1982. Macroinvertebrate communities associated with the macrophytes of Lake Vechten: Structural functional relationships. Hydrobiol. 95:115-126.

Dwyer, T. J., G. L Krapu, and D. M. Janke. 1979. Use of prairie pothole habitat by breeding mallards. J. Wildl. Manage. 43:526-531.

Dybvig, W. L., and J. R. Hart. 1977. The effects of agricultural drainage on flood flows at Moose Jaw. Pages 311-321 in Proc. 1977 Canadian Hydrology Symposium. National Res. Counc. Canada, Ottawa.

Dzubin, A. 1969. Comments on carrying capacity of small ponds for ducks and possible effects of density on mallard production. Pages 239-267 in J. T. Ratti, L. D. Flake, and W. A. Wentz (eds.), Waterfowl ecology and management: Selected readings. Allen Press, Inc., Lawrence, KS.

Eckbald, J. W., N. L. Peterson, and K. Ostlie. 1977. The morphometry, benthos and sedimentation rates of a floodplain lake in Pool 9 of the upper Mississippi River. Am. Midl. Nat. 97:433-443.

Ehrenfeld, J. G. 1983. The effects of changes in land use on swamps of the New Jersey pine barrens. Biol. Cons. 25:353-375.

Ehrenfeld. J. G. 1987. Wetlands of the New Jersey pine barrens: The role of species composition in community function. Am. Midl. Nat. 115:301-313.

Eilers, H. P., III. 1975. Plants, plant communities, net production and tide levels: The ecological biography of the Nehalem salt marshes, Tillamook County, Oregon. Ph.D. dissertation, Oregon State Univ., Corvallis. 387 p.

Eisenlohr, W. S., Jr. 1966. Water loss from a natural pond through transpiration by hydrophytes. Water Resour. Res. 2:443-453.

Eisenlohr, W. S., Jr., C. E. Sloan, and J. B. Shieflo. 1972. Hydrologic investigations of prairie potholes in North Dakota, 1959-68. Prof. Pap. 585-A. US Geological Survey, Reston, VA. 102 p.

Elder, J. F. 1985. Nitrogen and phosphorus speciation and flux in a large Florida river wetland system. Water Resour. Res. 21:724-732.

Elder, J. F., and D. J. Cairns. 1982. Production and decomposition of forest litter fall on the Apalachicola River flood plain, Florida. Water-Supply Paper 2196, Chap. B. US Geological Survey, Reston, VA. 42 p.

Elder, J. F., and H. C. Mattraw, Jr. 1982. Riverine transport of nutrients and detritus to the Apalachicola Bay Estuary, Florida. Water Resour. Res. 18:849-856.

Eli, R. N., and H. W. Rauch. 1982. Fluvial hydrology of wetlands in Preston County, West Viginia. Pages 11-28 in Symp. on Wetlands of the Unglaciated Appalachian Region. West Virginia Univ., Morgantown.

Elser, A. A. 1968. Fish populations of a trout stream in relation to major habitat zones and channel alterations. Trans. Am. Fish. Soc. 97:389-397.

Engel, S. 1988. The role and interactions of submersed macrophytes in a shallow Wisconsin Lake. J. Freshw. Ecol. 4:329-341.

Engler, R. M., and Patrick, W. H. 1974. Nitrate removal from floodwater overlying flooded soils and sediments. J. Environ. Qual. 3:490.

Erman, D. C., and W. C. Chouteau. 1979. Fine particulate organic carbon output from fens and its effect on benthic macroinvertebrates. Oikos 32:409-415.

Erman, D. C., and F. K. Ligon. 1988. Effects of discharge fluctuation and the addition of fine sediment of stream fish and macroinvertebrates below a water-filtration facility. Envir. Manage. 12:85-97.

Erskine, A. J. 1971. Bird communities in and around Cape Breton wetlands. Can. Field-Natur. 85:129-140.

Erwin, R. M., J. A. Spendelow, P. H. Geissler, and B. K. Williams. 1987. Relationships between nesting populations of wading birds and habitat features along the Atlantic coast. Pages 56-69 in W. R. Whitman, and W. H. Meredith (eds.), Proc. of a Symp. on Waterfowl and Wetland Management in the Coastal Zone of the Atlantic Flyway. Delaware Dep. Nat. Resour. and Environ. Cont., Dover.

Euliss, N. H., Jr., and G. Grodhaus. 1987. Management of midges and other invertebrates for waterfowl wintering in California. Calif. Fish and Game 73:238-243.

Evans, C. D., and K. E. Black. 1956. Duck production studies on the prairie potholes of South Dakota. Spec. Rep. (Wildl.) No. 32. US Fish Wildl. Serv., Washington, DC. 59 p.

Evans, R. D., and C. W. Wolfe, Jr. 1967. Waterfowl production in the rainwater basin area of Nebraska. J. Wildl. Manage. 31:788-794.

Evans, R. D., and F. H. Rigler. 1983. A test of lead-210 dating for the measurement of whole lake soft sediment accumulation. Can. J. Fish. Aquat. Sci. 40:506-515.

Ewel, K. 1976. Seasonal changes in distribution of water fern and duckweed in cypress domes receiving sewage. Pages 164-170 in H. T. Odum, K. C. Ewel, J. W. Ordway, and M. K. Johnston (eds.), Cypress Wetlands for Water Management, Recycling and Conservation, Third Annual Report. Univ. of Florida, Gainesville.

Faanes, C. A. 1982. Avian use of Sheyenne Lake and associated habitats in central North Dakota. Resour. Publ. No. 144. US Fish Wildl. Serv., Washington, DC. 24 p.

Fairchild, J. F., T. P. Boyle, E. Robinson-Wilson, and J. R. Jones. 1984. Effects of inorganic nutrients on microbial leaf decomposition and mitigation of chemical perturbation. J. Freshw. Ecol. 2:405-416.

Fannin, T. E., M. Parker, and T. J. Maret. 1985. Multiple regression analysis for evaluating non-point source contributions to water quality in the Green River, Wyoming. Pages 201-205 in R. R. Johnson et al. (eds.), Riparian Ecosystems and Their Management. Gen. Tech. Rep. RM-120. US For. Serv., Fort Collins, CO.

Farnworth, E. G., M. C. Nichols, C. N. Vann, L. G. Wolfson, R. W. Bosserman, P. R. Hendrix, F. B. Gallery, and J. L. Cooley. 1979. Impacts of sediment and nutrients on biota in surface waters of the United States. EPA-600/3-79-105. US Environmental Protection Agency. 331 p.

Faye, R. E., W. P. Carey, J. K. Stamer, R. L. Kleckner. 1980. Erosion, sediment discharge and channel morphology in the Upper Chatanoochee River Basin, Georgia. Prof. Paper 1107. US Geological Survey, Reston, VA.

Featherly, H. I. 1940. Silting and forest succession in Deep Fork in Southwestern

Creek County, Oklahoma. Proc. Oklahoma Acad. Sci. 24:63-65.

Fefer, S. I. 1977. Waterfowl populations as related to habitat changes in bog wetlands of the Moosehorn National Wildlife Refuge. Tech. Bull. 86. Life Sci. Agric. Exp. Stn., Univ. Maine 16 p.

Fenchel, T. 1970. Studies on the decomposition of organic detritus derived from the turtle grass (*Thalassia testudinum*). Limnol. Oceanogr. 15:14-20.

Fenchel, T. 1972. Aspects of decomposer food chains in marine benthos. Verh. Dtsch. Zool. Ges. 65:14-22.

Fenchel, T., and T. H. Blackburn. 1979. Bacteria and Mineral Cycling. Academic Press, New York.

Fetter, C. W., Jr., W. E. Sloey, and F. L. Spangler. 1978. Use of a natural marsh for wastewater polishing. J. Water Pollut. Cont. Fed. 50:290-307.

Fetter, C. W., Jr. 1980. Applied Hydrogeology. Bell and Howell Co., Columbus, OH. 488 p.

Findlay, S., and K. Tenore. 1982. Nitrogen source for a detritivore: Detritus substrate versus associated microbes. Science 218:371-373.

Fisher, S. G., and A. LaVoy. 1972. Differences in littoral fauna due to fluctuating water levels below a hydroelectric dam. J. Fish. Res. Bd. Can. 29:1472-1476.

Fisher, S. G., and S. R. Carpenter. 1976. Ecosystem and macrophyte primary production of the Fort River, Massachusetts. Hydrobiol. 47:175-187.

Fisher, S. G., L. J. Gray, N. B. Grimm, and D. E. Busch. 1982. Temporal succession in a desert stream ecosystem following flash flooding. Ecol. Monogr. 52:93-110.

Fisher, T. R., P. R. Carlson, and R. T. Barber. 1982. Carbon and nitrogen primary productivity in three North Carolina estuaries. Est. Coast. Shelf Sci. 15:621-644.

Flake, L. D. 1979. Wetland diversity and waterfowl. Pages 312-319 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Res. Assoc., Minneapolis, MN.

Flood, B. S., M. E. Sangster, R. D. Sparrowe, and R. S. Baskett. 1979. A handbook for habitat evaluation procedures. Resour. Publ. 132. US Fish Wildl. Serv., Washington, DC. 77 p.

Flores, A. C., P. B. Bedient, and L. W. Mays. 1981. Method for optimizing size and location of urban detention storage. Pages 357-365 in Proc. of the Internatl. Symp. on Urban Hydrology, Hydraulics and Sediment Control. ASCE, New York, NY.

Floyd, K. P., R. D. Hoyt, and S. Timbrook. 1984. Chronology of appearance and habitat partitioning by stream larval fishes. Trans. Am. Fish. Soc. 113:216-223.

Fonseca, M. S., and J. S. Fisher. 1986. A comparison of canopy friction and sediment movement between four species of seagrass with reference to their ecology and restoration. Mar. Ecol. Prog. Ser. 29:15-22.

Fonseca, M. S., W. J. Kenworthy, and G. W. Thayer. 1982. A low-cost transplanting procedure for sediment stabilization and habitat development using eelgrass. Wetlands 2:138-151.

Fonseca, M. S., J. C. Zieman, G. W. Thayer, and J. S. Fisher. 1983. The role of current velocity in structuring eelgrass

(*Zostera marina*) land meadows. Est. Coast. Shelf Sci. 17:367-380.

Francis, M. M., R. J. Naiman, and J. M. Melillo. 1985. Nitrogen fixation in subarctic streams influenced by beaver (*Castor canadensis*). Hydrobiol. 121:193-202.

Franz, D. R., and W. H. Harris. 1988. Seasonal and spatial variability in macrobenthos communities in Jamaica Bay, NY— An urban estuary. Estuaries 11:15-28.

Fredrickson, L. H. 1979. Floral and faunal changes in lowland hardwood forests in Missouri resulting from channelization, drainage and impoundment. FWS/OBS-78/91. US Fish Wildl. Serv. 130 p.

Freemark, K. E., and H. G. Merriam. 1986. Importance of area and habitat heterogeneity to bird assemblages in temperate forest fragments. Biol. Conserv. 36:115-141.

Freeze, R. A., and P. A. Witherspoon. 1967. Theoretical analysis of regional groundwater flow; 2. Effect of water-table configuration and subsurface permeability variation. Water Resour. Res. 3:623-634.

Friday, L. E. 1987. The diversity of macroinvertebrate and macrophyte communities in ponds. Freshw. Biol. 18:87-104.

Fried, E. 1974. Priority rating of wetlands for acquisition. Trans. Northeast Fish Wildl. Conf. 31:15-30.

Friedman, R. M., and C. B. Dewitt. 1978. Wetlands as carbon and nutrient reservoirs; A spatial, historical, and societal perspective. Pages 175-185 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Fritz, W. R., and S. C. Helle. 1978. Cypress wetlands as a natural treatment method for secondary effluents. Pages 69-81 in M. A. Drew (ed.), Symp. on Freshwater Wetlands. Coord. Counc. Restor. Kissimmee R. Val. and Taylor Cr.-Nubbin Slough Basin, Tallahassee, FL.

Froelich, P. N. 1988. Kinetic control of dissolved phosphate in natural rivers and estuaries: A primer on the phosphate buffer mechanism. Limnol. Oceanogr. 33:649-668.

Fryer, G. 1980. Acidity and species diversity in freshwater crustacean faunas. Freshw. Biol. 10:41-45.

Fryer, G. 1985. Crustacean diversity in relation to the size of water bodies: Some facts and problems. Freshw. Biol. 15:347-361.

Fukuhara, H., and M. Sakamoto. 1987. Enhancement of inorganic nitrogen and phosphate release from lake sediment by tubificid worms and chironomid larvae. Oikos 48:312-320.

Gaines, D. A. 1974. A new look at the nesting riparian avifauna of the Sacramento Valley, California. West. Birds 5:61-80.

Gallagher, J. L., W. J. Pfeiffer, and L. R. Pomeroy. 1976. Leaching and microbial utilization of dissolved organic carbon from leaves of *Spartina alterniflora*. Est. Coast. Mar. Sci. 4:467-471.

Gallagher, J. L. 1978. Decomposition processes: Summary and recommendations. Pages 145-151 in R. E. Good, D. F. Whigham, and R. L. Simpson (eds.), Freshwater wetlands: Ecological processes and management potential. Academic Press, New York.

Gallagher, J. L., and H. V. Kibby. 1980. Marsh plants as vectors in trace metal transport in Oregon tidal marches. Am. J. Bot. 67:1069-1074.

Gallep, G. W. 1979. Chironomid influence on phosphorus release in sediment-water microcosms. Ecology 60:547-556.

Galli, A. E., C. F. Leck, and R. T. Forman. 1976. Avian distribution patterns in forest islands of different sizes in central New Jersey. Auk 63:356-365.

Gammon, J. R. 1970. The effect of inorganic sediment on stream biota. Rep. No. 18050 DWC 12/70. US Environmental Protection Agency, Washington, DC. 141 p.

Garbisch, E. W., Jr. 1977. Marsh development for shore erosion. Pages 77-94 in Proc. of the workshop on the role of vegetation in stabilization of the Great Lakes shoreline. Great Lakes Basin Commission, Ann Arbor, MI.

Garie, H. L., and A. McIntosh. 1986. Distribution of benthic macroinvertebrates in a stream exposed to urban runoff. Water Resour. Bull. 22:447-455.

Gates, J. M. 1965. Duck nesting and production on Wisconsin farmlands. J. Wildl. Manage. 29:515-523.

Gauthreaux, S. A. 1978. The structure and organization of avian communities in forests. Pages 17-37 in R. M. DeGraaf (tech. coord.), Management of southern forests for nongame birds. Gen. Tech. Rep. SE-14. US For. Serv., Washington, DC.

George, U. S., and A. D. Antoine. 1982. Denitrification potential of a salt marsh soil: Effect of temperature, pH and substrate concentration. Soil Biol. Biochem. 14:117-125.

Gersberg, R. M., B. V. Elkins, and C. R. Goldman. 1983. Nitrogen removal in artificial wetlands. Water Resour. Res. 17:1009-1014.

Gersberg, R. M., B. V. Elkins, and C. R. Goldman. 1984. Use of artificial wetlands to remove nitrogen from wastewater. J. Water Pollut. Control Fed. 56:152-156.

Gersberg, R. M., B. V. Elkins, S. R. Lyon, and C. R. Goldman. 1986. Role of aquatic plants in wastewater treatment by artificial wetlands. Water Res. 20:363-368.

Gessner, F. 1959. Hydrobotanik die physiologischen granlagen dir pflanzenvebreiting in Wasser; II, Stuffhanhalt, Berlin, VEB Deutsch Verlag der Wissenschaften. 701 p.

Giblin, A. E. 1985. Comparisons of the processing of elements by ecosystems; II. Metals. Pages 158-179 in P. J. Godfrey et al. (eds.), Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. Van Nostrand Reinhold, New York.

Gibson, R. J., and D. Galbraith. 1975. The relationships between invertebrate drift and salmonoid populations in the Matamek River, Quebec, below a lake. Trans. Am. Fish. Soc. 104:529-535.

Gilinsky, E. 1984. The role of fish predation and spatial heterogeneity in determining benthic community structure. Ecology 65:455-468.

Gillion, R. J. 1985. Pesticides in rivers of the United States. Pages 85-92 in National Water Summary 1984. Pages 85-92 in Water-Supply Paper 2275. US Geological Survey, Reston, VA.

Gilmer, D. S., I. J. Ball, L. M. Cowardin, J. H. Riechmann, and J. R. Tester. 1975. Habitat use and home range of mallards breeding in Minnesota. J. Wildl. Manage. 39:781-789.

Gilmore, R. G., D. W. Cook, and C. J. Donohoe. 1981. A comparison of the fish populations and habitat in open and closed salt marsh impoundments in east-central Florida. N. E. Gulf Sci. 5:25-37.

Gislason, J. C. 1985. Aquatic insect abundance in a regulated stream under fluctuating and stable diel flow patterns. N. Am. J. Fish. Manage. 5:39-46.

Gleason, M. C., D. A. Elmer, N. C. Pien, and J. S. Fisher. 1979. Effects of stem density upon sediment retention by salt marsh cordgrass, *Spartina alterniflora* Loisel. Estuaries 2:271-273.

Glover, F. 1956. Nesting and production of the blue-winged teal in northwest Iowa. J. Wildl. Manage. 20:28-46.

Godshalk, G. L., and R. G. Wetzel. 1978. Decomposition in the littoral zone of lakes. Pages 131-144 in R. E. Good, D. F. Whigham, and R. L. Simpson (eds.), Freshwater Wetlands. Academic Press, New York.

Goldstein, R. M. 1981. Longitudinal succession in impact assessment of river system fish communities. Water Resour. Bull. 17:75-81.

Golet, F. C., and J. S. Larson. 1974. Classification of freshwater wetlands in the glaciated Northeast. Resour. Publ. 116. US Fish Wildl. Serv., Washington, DC. 56 p.

Goodwin, R. H., and W. A. Neiring. 1975. Inland wetlands of the United States. National Park Serv., Washington, DC. 550 p.

Gordon, D. C., P. J. Cranford, and C. Desplanque. 1985. Observations on the ecological importance of salt marshes in the Cumberland Basin, a macrotidal estuary in the Bay of Fundy. Estuarine Coastal Shelf Sci. 20:205-227.

Gordon, D. H., B. T. Gray, and R. M. Kaminski. 1987. A preliminary analysis of habitat use by dabbling ducks wintering in coastal wetlands of South Carolina. Pages 13-25 in W. R. Whitman, and W. H. Meredith (eds.), Proc. of a Symp. on Waterfowl and Wetland Management in the Coastal Zone of the Atlantic Flyway. Delaware Dep. Nat. Resour. and Environ. Cont., Dover.

Gore, J. A. 1982. Benthic invertebrate colonization: Source distance effects on community composition. Hydrobiol. 94:183-193.

Gorman, O. T., and J. R. Karr. 1978. Habitat structure and stream fish communities. Ecology 59:507-515.

Gosselink, J. G., and R. E. Turner. 1978. The role of hydrology in freshwater ecosystems. Pages 63-78 in R. E. Good, D. F. Whigham, and R. L. Simpson (eds.), Freshwater Wetlands. Academic Press, New York.

Gosselink, J. G., and L. C. Lee. 1987. Cumulative impact assessment in bottomland hardwood forests. Rep. No. LSU-CEI-86-09. Center for Wetland Resour., Louisiana State Univ., Baton Rouge. 55 p.

Grant, R. R., and R. Patrick. 1970. Tinicum Marsh as a water purifier. Pages 105-123 in Two Studies of Tinicum Marsh. The Conserv. Found., Washington, DC.

Gray, B. T., D. H. Gordon, and R. M. Kaminski. 1987. Functional attributes of coastal wetlands for waterfowl: Perspectives for research and management. Pages 205-222 in W. R. Whitman, and W. H. Meredith (eds.), Proc. of a

Symp. on Waterfowl and Wetland Management in the Coastal Zone of the Atlantic Flyway. Delaware Dep. Nat. Resour. and Environ. Cont. Dover.

Gray, D. H. 1977. The influence of vegetation on slope processes in the Great Lakes Region. Pages 5-29 in Proc., Workshop on the Role of Vegetation in Stabilization of the Great Lakes Shoreline. Great Lakes Basin Comm., Ann Arbor, MI.

Gray, D. M. 1974. Reinforcement and stabilization of soil by vegetation. J. Geotech. Eng. Div. 100(GT6):696-699.

Greij, E. D. 1976. The effects of a marsh on water quality. Inst. Water Res. Tech., Michigan State Univ., Lansing. 188 p.

Grimm, N. B. 1987. Nitrogen dynamics during succession in a desert stream. Ecology 68:1157-1170.

Griswold, B. L. 1978. Some effects of stream channelization on fish populations, macroinvertebrates, and fishing in Ohio and Indiana. FWS/OBS-77/46. US Fish Wildl. Serv., Columbus, MO.

Grue, C. E., L. R. DeWeese, P. Mineau, G. A. Swanson, J. R. Foster, P. M. Arnold, J. N. Huckins, P. J. Sheehan, W. K. Marshall, and A. P. Ludden. 1986. Potential impacts of agricultural chemicals on waterfowl and other wildlife inhabiting prairie wetlands: An evaluation of research needs and approaches. Trans. N. Am. Wildl. Nat. Res. Conf. 51:357-383.

Grushevsky, M. S. 1967. Floodplain influence on flood wave propagation along a river. Pages 745-754 in Int. Symp. on Floods and Their Computation, Vol 2. IASH-UNESCO-WMO Publ. 85.

Guenzi, W. D., and W. E. Beard. 1968. Anaerobic conversion of DDT to DDD and aerobic stability of DDT in soil. Proc. Soil Sci. Soc. Am. 32:522-524.

Guenzi, W. D., W. E. Beard, and F. G. Viets, Jr. 1971. Influence of soil treatment on persistence of six chlorinated hydrocarbon insecticides in the field. Proc. Soil. Sci. Soc. Am. 35:910-913.

Guildford, S. J., F. P. Healey, and R. E. Hecky. 1987. Depression of primary production by humic matter and suspended sediment in limnocorral experiments at Southern Indian Lake, Northern Manitoba. Can. J. Fish. Aquat. Sci. 44:1408-1417.

Guillory, V. 1979. Utilization of an inundated floodplain by Mississippi River fishes. Fla. Sci. 42:222-228.

Gurtz, M. E., and J. B. Wallace. 1984. Substrate-mediated response of stream invertebrates to disturbance. Ecology 65:1556-1569.

Gurtz, M. E., and C. M. Tate. 1988. Hydrologic influences on leaf decomposition in a channel and adjacent bank of a gallery forest stream. Am. Midl. Nat. 120:11-21.

Guthery, F. S., S. M. Obenberger, and F. A. Stormer. 1984. Predictors of site use by ducks on the Texas High Plains. Wildl. Soc. Bull. 12:35-40.

Hackney, C. T. 1977. Energy flux in a tidal creek draining an irregularly flooded Juncus marsh. Ph.D. dissertation, Mississippi State Univ., Starkville. 83 p.

Hackney, C. T. 1987. Factors affecting accumulation or loss of macroorganic matter in salt marsh sediments. Ecology 68:1109-1113.

Hackney, C. T., and A. A. de la Cruz. 1980. In situ decomposition of roots and rhizomes of two tidal marsh plants. Ecology 61:226-231.

Hackney, C. T., and T. D. Bishop. 1981. A note on relocation of marsh debris during a storm surge. Est. Coast. Mar. Sci. 12:621-624.

Haines, E. B. 1977. The origins of detritus in Georgia salt marsh estuaries. Oikos 29:254-260.

Haines, E. B., A. G. Chalmers, R. B. Hanson, and B. Sherr. 1977. Nitrogen pools and fluxes in a Georgia salt marsh. Pages 241-254 in M. Wiley (ed.), Estuarine Processes. Academic Press, New York.

Haines, E. B., and G. L. Montague. 1979. Food sources of estuarine invertebrates analyzed using $^{13}C/^{12}C$ ratios. Ecology 60:48-56.

Haines, T. A. 1981. Acidic precipitation and its consequence for aquatic ecosystems: A review. Trans. Am. Fish. Soc. 110:669-707.

Hair, J. D., G. T. Hepp, L. M. Luckett, K. P. Reese, and D. K. Woodward. 1978. Beaver pond ecosystems and their relationships to multi-use natural resource management. Pages 80-92 in R. R. Johnson and J. F. McCormick (tech. coords.), Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. Gen. Tech. Rep. WO-12. US For. Serv., Washington, DC.

Hale, S. S., T. E. McMahon, and P. C. Nelson. 1985. Habitat suitability index models: Chum salmon. Biol. Rep. 82(10.108). US Fish Wildl. Serv., Fort Collins, CO. 48 p.

Hall, D. J., and E. E. Werner. 1977. Seasonal distribution and abundance of fishes in the littoral zone of a Michigan Lake. Tran. Am. Fish. Soc. 106:545-555.

Hammer, T. 1972. Stream channel enlargement due to urbanization. Water Resour. Res. 8:1530-1540.

Hammond, M. C., and G. E. Mann. 1956. Waterfowl nesting on islands. J. Wildl. Manage. 20:345-352.

Hanks, K. S., D. E. Wallace, and H. A. Schreiber. 1981. Bacteriological ground water quality characteristics of the Walnut Gulch Experimental Watershed. Conserv. Res. Rep. No. 28. Arizona Sci. Educ. Admin., Phoenix, AZ. 15 p.

Hanlon, R. D. G. 1982. The breakdown and decomposition of allochthonous and autochthonous plant litter in an oligotrophic lake. Hydrobiol. 88:281-288.

Hansen, E. A. 1971. Stream channelization effects of fishes and bottom fauna in the Little Sioux River, Iowa. Pages 29-51 in E. Schneberger and J. L. Funk (eds.), Stream Channelization: A Symposium. N. Cen. Div. Am. Fish. Soc., Spec. Publ. No. 2. Omaha, NE.

Hanson, H. C., M. Rogers, and E. S. Rogers. 1949. Waterfowl of the forest portions of the Canadian Pre-Cambrian shield and Paleozoic Basin. Can. Field-Nat. 63:183-204.

Hanson, R. B. 1977. Nitrogen fixation (acetylene reduction) in a salt marsh amended with sewage sludge + organic carbon + nitrogen compounds. Appl. Environ. Microbiol. 33:846-852.

Hanson, R. B. 1983. Nitrogen fixation activity (acetylene-reduction) in the rhizophere of salt marsh angiosperms, Georgia, USA. Bot. Mar. 26:49-59.

Hardin, D. L. 1987. An evaluation of impoundments and ponds created for waterfowl in Delaware tidal marshes. Pages 121-126 in W. R. Whitman and W. H. Meredith (eds.), Proc. of a Symp. on Waterfowl and Wetland Management in the Coastal Zone of the Atlantic Flyway. Delaware Dep. Nat. Resour. and Environ. Cont., Dover.

Hargeby, A., and R. C. Petersen, Jr. 1988. Effects of low pH and humus on the survivorship, growth and feeding of *Grammarus pulex* (L.) (Amphipoda). Freshw. Biol. 19:235-247.

Hargrave, B. T., N. J. Prouse, G. A. Phillips, and P. A. Neame. 1983. Primary production and respiration in pelagic and benthic communities at two intertidal sites in the upper Bay of Fundy. Can. J. Fish. Aquat. Sci. 40 (Supp. 1):229-243.

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

Harms, W. R. 1973. Some effects of soil type and water regime on growth of tupelo seedlings. Ecology 54:188-193.

Harr, R. D. 1976. Hydrology of small forested streams in Western Oregon. Gen. Tech. Rep. PNW-55. US For. Serv., Washington, DC. 15 p.

Harr, R. D., W. C. Harper, J. T. Krygier, and F. S. Hsieh. 1975. Changes in storm hydrographs after road building and clear-cutting in the Oregon Coast Range. Water Resour. Res. 11:436-444.

Harrell, R. C., B. J. Davis, and T. C. Dorris. 1967. Stream order and species diversity of fishes in an intermittent Oklahoma stream. Am. Midl. Nat. 78:428-436.

Harrington, R. W., Jr., and E. S. Harrington. 1982. Effects on fishes and their forage organisms of impounding a Florida salt marsh to prevent breeding by salt marsh mosquitoes. Bull. Mar. Sci. 32:523-531.

Harris, H. J., M. S. Milligan, and G. A. Fewless. 1983. Diversity: Quantification and ecological evaluation in freshwater marshes. Biol. Conserv. 27:99-110.

Harris, L. D. 1984. The fragmented forest-island biogeography theory and the preservation of biotic diversity. Univ. of Chicago Press. 211 p.

Harris, L. D., and C. R. Vickers. 1984. Some faunal community characteristics of cypress ponds and the changes induced by perturbations. Pages 171-185 in K. C. Ewel and H. T. Odum (eds.), Cypress Swamps. Univ. Florida Press, Gainesville.

Harrison, S. S., and L. Clayton. 1970. Effects of groundwater on fluvial processes. Geol. Soc. of Am. Bull. 81:1217-1226.

Hart, C. W., Jr., and S. L. H. Fuller. 1972. Environmental degradation in the Patuxent River Estuary, Maryland. No. 1. Philadelphia Acad. Nat. Sci., Dep. Limnol. 14 p.

Hart, J. T. 1982. Uptake of trace metals by sediments and suspended particulates: A review. Hydrobiol. 91:299-313.

Hartland-Rowe, R., and P. P. Wright. 1975. Effects of sewage effluent on a swampland stream. Verh. Internat. Verein. Limnol. 19:1575-1583.

Hartman, F. E. 1960. Ecology of black ducks wintering in the Penobscot estuary. M.S. thesis, Univ. of Maine, Orono. 142 p.

Harvey, J. W., P. F. Germann, and W. E. Odum. 1987. Geomorphological control of subsurface hydrology in the creekbank zone of tidal marshes. Est. Coast. Shelf Sci. 25:677-691.

Hatcher, R. M. 1973. Floodwater retarding structures as fish and wildlife habitat. Pages 35-37 in Wildlife and Water Management: Striking a Balance. Soil Conserv. Soc. Am., Ankeny, IA.

Hawkins, C. P., M. L. Murphy, and N. H. Anderson. 1982. Effects of canopy, substrate composition, and gradient on the structure of macroinvertebrate communities in Cascade Range streams of Oregon. Ecology 63:1840-1856.

Heal, O. W., and D. D. French. 1974. Decomposition of organic matter in tundra. Pages 279-304 in A. J. Holding, O. W. Heal, S. F. Maclean, and P. W. Flanagan (eds.), Soil Organisms and Decomposition in Tundra. The Univ. of Alaska Press, College, AK.

Heaton, T. H. E. 1986. Isotopic studies of nitrogen pollution in the hydrosphere and the atmosphere: A review. Chem. Geol. (Isotope Geoscience Sec.) 59:87-102.

Heck, K. L., and T. A. Thoman. 1984. The nursery role of seagrass meadows in the upper and lower reaches of the Chesapeake Bay. Estuaries 7:70-92.

Heckman, C. W. 1986. The role of marsh plants in the transport of nutrients as shown by a quantitative model for the freshwater section of the Elbe Estuary. Aquat. Bot. 25:139-151.

Heifetz, J., M. L. Murphy, and K. V. Koski. 1986. Effects of logging on winter habitat of juvenile salmonids in Alaskan streams. N. Am. J. Fish. Manage. 6:52-58.

Heinemann, H. G. 1981. A new sediment trap efficiency curve for small reservoirs. Water Resour. Bull. Am. Res. Assoc. 17:825-830.

Heinle, D. R., and D. A. Flemer. 1976. Flows of material between poorly flooded tidal marshes and an estuary. Marine Biol. 35:349-373.

Heinselman, M. L. 1970. Landscape evolution, peatland types and the environment in the Lake Agassiz Natural Area, Minnesota. Ecol. Monogr. 40:235-261.

Heitmeyer, M. E., and L. H. Fredrickson. 1981. Do wetland conditions in the Mississippi Delta hardwoods influence mallard recruitment? Trans. N. Am. Wildl. Nat. Resour. Conf. 46:44-57.

Heliotis, F. D., and C. B. DeWitt. 1983. A conceptual model of nutrient cycling in wetlands used for wastewater treatment: A literature analysis. Wetlands 3:134-152.

Heliotis, F. D., and C. B. De Witt. 1987. Rapid water table responses to rainfall in a northern peatland ecosystem. Water Resour. Bull. 23:1011-1016.

Hemond, H. F. 1983. The nitrogen budget of Thoreau's Bog. Ecology 64:99-109.

Henbry, M. S., and J. Cairns, Jr. 1984. Protozoan colonization rates and trophic status of some freshwater wetland lakes. J. Protozool. 31:456-467.

Hendricks, A. C., F. Taylor, and G. S. Simmons. 1984. Leaf litter processing in a Virginia reservoir—the role of temperature and fish. Verhandlungen der Internationale Vereinigung fur Theoretische und Angewandte Limnologie 22:1482-1485.

Hendrickson, G. E., R. L. Knutilla, and C. J. Doonan. 1973. Hydrology and recreation on selected coldwater rivers of the

St. Lawrence River basin in Michigan, New York, and Wisconsin. Rep. USGS-WRD-73-009. US Geological Survey, Reston, VA. 73 p.

Hennings, R. G. 1978. The hydrogeology of a sand plain seepage lake, Portage County, Wisconsin. M.S. thesis, Univ. of Wisconsin, Madison. 69 p.

Hepp, G. R., and J. D. Hair. 1979. Wood duck brood mobility and utilization of beaver pond habitats. Proc. S. E. Assoc. Game Fish Comm. 31:216-225.

Herke, W. H. 1971. Use of natural, and semi-impounded, Louisiana tidal marshes as nurseries for fishes and crustaceans. Ph.D. dissertation, Louisiana State Univ., Baton Rouge. 264 p.

Herlong, D. D., and M. A. Mallin. 1985. The benthos-plankton relationship upstream and downstream of a blackwater impoundment. J. Freshw. Ecol. 3:47-59.

Herman, S. S., J. A. Mihursky, and A. J. McErlean. 1968. Zooplankton and environmental characteristics of the Patuxent River estuary, 1963-1965. Ches. Sci. 9:67-82.

Herron, R. C. 1985. Phosphorus dynamics in Dingle Marsh, Idaho. Ph.D. dissertation, Utah State Univ., Logan. 153 p.

Herron, R. C., V. A. Lamarra, and V. D. Adams. 1984. The nitrogen, phosphorus, and carbon budgets of a large riverine marsh and their impact on the Bear Lake ecosystem. Pages 223-228 in Lake and Reservoir Management. EPA 440/5/84-001. US Environmental Protection Agency, Washington, DC.

Hewlett, J., and A. Hibbert. 1965. Factors affecting the response of small watersheds to precipitation in humid areas. Pages 275-290 in W. T. Stopper and H. W. Lull (eds.), Forest Hydrology. Pergamon Press, New York.

Hickok, E. A., M. C. Hannaman, and N. C. Wenck. 1977. Urban runoff treatment methods; Vol 1, Nonstructural wetlands treatment. EPA-600/1-77-217. US Environmental Protection Agency, Washington, DC. 131 p.

Higgins, K. F. 1977. Duck nesting in intensively farmed areas of North Dakota. J. Wildl. Manage. 41:232-242.

Hildebrand, H. H. 1954. A study of the fauna of the pink shrimp (*Penaeus duorarum* Burkenroad) grounds in the Gulf of Campeche. Publ. Inst. Mar. Sci. 4:169-232.

Hill, A. R. 1979. Denitrification in the nitrogen budget of a river ecosystem. Nature 281:291-292.

Hill, A. R., and K. Sanmugadas. 1985. Denitrification rates in relation to stream sediment characteristics. Water Res. 19:1579-1586.

Hill, B. H. 1985. The breakdown of macrophytes in a reservoir wetland. Aquat. Bot. 21:23-31.

Hill, B. H., and J. R. Webster. 1983. Aquatic macrophyte contribution to the New River organic matter budget. Pages 273-281 in T. D. Fontaine III and S. M. Bartell (eds.), Dynamics of Lotic Ecosystems, Ann Arbor Sci., Ann Arbor, MI.

Hindall, S. M. 1975. Measurement and prediction of sediment yields in Wisconsin streams. Water Resour. Invest. 54-75. US Geological Survey, Reston, VA. 27 p.

Hines, J. E., and G. J. Mitchell. 1983. Breeding ecology of the gadwall at Waterhen Marsh, Saskachewan, Canada. Can. J. Zool. 61:1532-1539.

Ho, C. L., and B. B. Barrett. 1977. Distribution of nutrients in Louisiana's coastal waters influenced by the Mississippi River. Est. Coast. Mar. Sci. 5:173-195.

Hoffman, E. J., J. S. Latimer, C. D. Hunt, and J. G. Quinn. 1985. Storm water runoff from highways. Water, Air, and Soil Pollution 25:449-464.

Holland, L. E. 1987. Effect of brief navigation-related dewaterings on fish and egg larvae. N. Am. J. Fish Manage. 7:145-147.

Holt, S. A., C. L. Kitting, and C. R. Arnold. 1983. Distribution of young red drums among different sea-grass meadows. Trans. Am. Fish. Soc. 112:267-271.

Hook, D. D. 1984. Waterlogging tolerance of lowland tree species of the south. South. J. Appl For. 8:136-149.

Hooke, J. M. 1979. An analysis of the processes of river bank erosion. J. Hydrol. 42:39-62.

Hooke, J. M. 1980. Magnitude and distribution of rates of river bank erosion. Earth Surface Processes and Landforms 5:143-157.

Hooper, F. F., and L. S. Morris. 1982. Mat-water phosphorus exchange in an acid bog lake. Ecology 63:1411-1421.

Hooper, R. G. 1967. The influence of habitat disturbances on bird populations. M.S. thesis, Virginia Polytechnic Inst., Blacksburg. 134 p.

Hopkinson, C. S., Jr., and J. P. Schubauer. 1984. Static and dynamic aspects of nitrogen cycling in the salt marsh graminoid *Spartina alterniflora*. Ecology 65:961-969.

Horner, R. R., and E. B. Welch. 1981. Stream periphyton development in relation to current velocity and nutrients. Can. J. Fish. Aquat. Sci. 38:449-457.

Howarth, R.W., R. Marino, J. Lane, and J.J. Cole. 1988. Nitrogen fixation in freshwater, estuarine, and marine ecosystems; 1. Rates and importance. Limnol. Oceanogr. 33(4):669-687.

Howes, B. L., R. W. Howarth, J. M. Teal, and I. Valiela. 1981. Oxidation-reduction potentials in a salt marsh: Spatial patterns and interactions with primary production. Limnol. Oceanogr. 26: 350-360.

Hoy, M. D. 1987. Waterfowl use of a north-central Texas reservoir. M.S. thesis, Texas A&M Univ., College Station. 90 p.

Hsieh, Y. P., and O. J. Weber. 1984. Net aerial primary production and dynamics of soil organic matter formation in a tidal marsh ecosystem. Soil Sci. Soc. Am. J. 48:65-72.

Hubbard, D. E., and R. L. Linder. 1986. Spring runoff retention in prairie pothole wetlands. J. Soil Water Conserv. 41:122-125.

Hubert, J. F., and J. N. Krull. 1973. Seasonal fluctuations of aquatic macroinvertebrates in Oakwood Bottoms greentree reservoir. Am. Midl. Nat. 90:177-185.

Hudson, H. R. 1982. A field technique to directly measure river bank erosion. Can. J. Earth Sci. 19:381-383.

Hudson, M. S. 1983. Waterfowl production on three age-classes of stock ponds in Montana. J. Wildl. Manage. 47:112-117.

Hughes, E. H., and E. B. Sherr. 1983. Subtidal food webs in a Georgia estuary: ^{13}C Analysis. J. Exp. Mar. Biol. Ecol. 67:227-242.

Hughes, R. M., and J. R. Gammon. 1987. Longitudinal changes in fish assemblages and water quality in the Wilamette River, Oregon. Trans. Am. Fish. Soc. 116(2):196-209.

Huish, M. T., and G. B. Pardue. 1978. Ecological studies of one channelized and two unchannelized swamp streams in North Carolina. FWS/OBS-78/85, Biological Services Program. US Fish and Wildlife Service, Washington, DC. 71 p.

Hull, R. W., J. E. Dysert, and W. B. Mann IV. 1981. Quality of surface water in the Suwannee River Basin, Florida, August 1968-December 1977. Water Resour. Invest. 80-110. US Geological Survey, Reston, VA. 97 p.

Hunt, P. B., and C. R. Lee. 1976. Land treatment of wastewater by overland flow for improved water quality. Pages 151-160 in J. Toubier and R.W. Pierson, Jr. (eds.), The biological control of water pollution. Univ. of Pennsylvania Press, Philadelphia.

Huryn, A. D., and J. B. Wallace. 1987. Local geomorphology as a determinant of macrofaunal production in a mountain stream. Ecology 68:1932-1942.

Hussey, M. F., Q. D. Skinner, J. C. Adams, and A. J. Harvey. 1985. Denitrification and bacterial numbers in riparian soils of a Wyoming mountain watershed. J. Range Manage. 38(6):492-496.

Hutchinson, G. E. 1975. A Treatise on Limnology; Vol 3, Limnological Botany. John Wiley and Sons, New York. 660 p.

Hutchinson, G. E. 1957. A Treatise on Limnology; Vol 1, Geography, Physics, and Chemistry. John Wiley and Sons, New York. 1015 p.

Hynes, H. B. N. 1970. The Ecology of Running Waters. Univ. Toronto Press. 555 p.

Idso, S. B. 1981. Relative rates of evaporative water losses from open and vegetation covered water bodies. Water Resour. Bull. 17:46-48.

Iizumi, H., A. Hattori, and C. P. McRoy. 1980. Nitrate and nitrite in interstitial waters of eelgrass beds in relation to the rhizosphere. J. Exp. Mar. Biol. Ecol. 47:191-201.

Imberger, J., T. Berman, R. R. Christian, E. B. Sherr, D. E. Whitney, L. R. Pomeroy, R. Wiegart, and W. J. Wiebe. 1983. The influence of water motion on the distribution and transport of materials in a saltmarsh estuary. Limnol. Oceanogr. 28:201-214.

Inman, E. J. 1987. Simulation of flood hydrographs for Georgia streams. Water-Supply Paper 2317. US Geological Survey, Reston, VA. 26 p.

Irving, J. R. 1985. Effects of successive flow perturbations on stream invertebrates. Can. J. Fish. Aquat. Sci. 42:1922-1927.

Isirimah, N. D. 1972. Nitrogen cycling in Lake Wingra. Ph.D. dissertation, Univ. of Wisconsin, Madison.

Isirimah, N., and D. Keeney. 1973. Contribution of developed and natural marshland soils to surface and subsurface water quality. Tech. Rept. WIS WRC 73-

09. Univ. Wis. Water Resour. Cent., Madison.

Jackson, H. O., and W. C. Starrett. 1959. Turbidity and sedimentation at Lake Chatuqua, Illinois. J. Wildl. Manag. 23(2):157-168.

Jackson, T. A., and R. E. Hecky. 1980. Depression of primary productivity by humic matter in lake and reservoir waters of the boreal forest zone. Can. J. Fish. Aquat. Sci. 37:2300-2317.

Jacobs, T. C., and J. W. Gilliam. 1985. Riparian losses of nitrate from agricultural drainage waters. J. Environ. Qual. 14:472-478.

Jacobs, T. C., and J. W. Gilliam. 1985. Headwater stream losses of nitrogen from two coastal plain watersheds. J. Environ. Qual. 14:467-471.

Jarvis, R. L., and S. W. Harris. 1971. Land use patterns and duck production at Malheur National Wildlife Refuge. J. Wildl. Manage. 35:767-773.

Jaynes, M. L., and S. R. Carpenter. 1980. Effects of vascular and nonvascular macrophytes on sediment redox and solute dynamics. Ecology 67:875-882.

Jeffries, H. P. 1982. Fatty-acid ecology of a tidal marsh. Limnol. Oceanogr. 17:433-440.

Jenkins, R. M. 1967. The influence of some environmental factors on standing crop and harvest of fishes in U.S. reservoirs. Pages 298-321 in Reservoir Fishery Resources Symposium, Athens, GA, April 1967. So. Div., Am. Fisheries Soc.

Jenkins, R. M. 1982. The morphedaphic index and reservoir fish production. Trans. Am. Fish. Soc. 111:133-140.

Jenkins, R. M., and I. Morais. 1971. Reservoir sport fishing effort and harvest in relation to environmental variables. Pages 371-384 in G. E. Hall (ed.), Reservoir Fisheries and Limnology. Am. Fish. Soc. Spec. Publ. No. 8. 511 p.

Johannes, R. E., and C. J. Hearn. 1985. The effect of submarine groundwater discharge on nutrient and salinity regimes in a coastal lagoon off Perth, Western Australia. Estuarine Coastal Shelf Sci. 21:801-815.

Johnson, B. H., and P. K. Senter. 1977. Effect of loss of valley storage in the Cannelton Pool on Ohio River flood heights. Misc. Paper H-77-7. US Army Engineer Waterways Experiment Station, Vicksburg, MS. 33 p.

Johnson, M. C., and J. B. Moyle. 1969. Management of a large shallow winterkill lake in Minnesota for the production of pike. Trans. Am. Fish. Soc. 98:691-697.

Johnson, R. P. 1963. Studies on the life history and ecology of the bigmouth buffalo. J. Fish. Res. Bd. Can. 20:1397-1429.

Johnson, R. J., and M. M. Beck. 1988. Influences of shelterbelts on wildlife management and biology. Agriculture, Ecosystems and Environment 22/23:301-307.

Johnson, W. C., R. J. Burgess, and W. R. Kreammerer. 1976. Forest overstory vegetation and environment on the Missouri River flood plains in North Dakota. Ecol. Monogr. 46:59-84.

Johnston, C. A., G. D. Bubenzer, G. B. Lee, F. W. Madison, and J. R. McHenry. 1984. Nutrient trapping by sediment deposition in a seasonally flooded lakeside wetland. J. Environ. Qual. 13:283-290.

Jonasson, P. M., and C. L. Lindegaard. 1979. Zoobenthos and its contribution to the metabolism of shallow lakes. Arch. Hydrobiol. Beih. Ergebn. Limnol. 13:162-180.

Jones, J. R., and M. V. Hoyer. 1982. Sportfish harvest predicted by summer chlorophyll-a concentration in midwestern lakes and rservoirs. Trans. Am. Fish. Soc. 111:176-179.

Jones, J. R., B. P. Borofka, and R. W. Bachmann. 1976. Factors affecting nutrient loads in some Iowa streams. Water Res. 10:117-122.

Jones, K. 1974. Nitrogen fixation in a salt marsh. J. Ecol. 62:553-565.

Jones, R. C. 1984. Application of a primary production model to epiphytic algae in a shallow eutrophic lake. Ecology 65:1895-1903.

Jones, R. C., and C. C. Clark. 1987. Impact of watershed urbanization on stream insect communities. Water Resour. Bull. 23(6):1047-1055.

Jordan, T. E., and I. Valiela. 1982. A nitrogen budget of the ribbed mussel, *Geukensia demissa*, and its significance in nitrogen flow in a New England salt marsh. Limnol. Oceanogr. 27:75-90.

Jordan, T. E., D. L. Correll, and D. F. Whigham. 1983. Nutrient flux in the Rhode River: Tidal exchange of nutrients by brackish marshes. Est. Coast. Shelf Sci. 17:651-667.

Jordan, T. E., D. L. Correll, W. T. Peterjohn, and D. E. Weller. 1986. Nutrient flux in a landscape: The Rhode River Watershed and receiving waters. Pages 57-76 in D. L. Correll (ed.), Watershed Research Perspectives. Smithsonian Institution Press, Washington DC.

Josselyn, M. (ed.). 1982. Wetland restoration and enhancement in California. Calif. Sea Grant Rep. No. T-CSGCP-007. 110 p.

Josselyn, M. 1983. The ecology of San Francisco Bay tidal marshes: A community profile. FWS/OBS-83/23. US Fish Wildl. Serv., Washington, DC. 102 p.

Joyner, D. E. 1980. Influence of invertebrates on pond selection by ducks on Ontario. J. Wildl. Manage. 44(3):700-705.

Judy, R. D., Jr., P. N. Seeley, T. M. Murray, S. C. Svirsky, M. R. Whitworth, and L. S. Ischinger. 1984. The 1982 National Fisheries Survey; Vol I, Technical report: Initial findings. FWS/OBS-84/06. US Fish Wildl. Serv., Washington, DC. 140 p.

Jupp, B. P., and D. H. N. Spence. 1977. Limitations of macrophytes in a eutrophic lake, Loch Leven; II, Wave action, sediments and waterfowl grazing. J. Ecol. 65:431-446.

Kadlec, J. A. 1986. Input-output budgets for small diked marshes. Can. J. Fish. Aquat. Sci. 43:2009-2016.

Kadlec, J. A. 1987. Nutrient dynamics in wetlands. Pages 393-419 in K. R. Reddy and W. H. Smith (eds.), Aquatic Plants for Water Treatment and Resource Recovery. Magnolia Publishing, Inc.

Kadlec, R. H., and J. A. Kadlec. 1979. Wetlands and water quality. Pages 436-456 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Kadlec, R. H., and D. L. Tilton. 1979. The use of freshwater wetlands as a tertiary wastewater treatment alternative. CRC Critical Reviews in Environmental Control:185-212.

Kadlec, R. H., and D. E. Hammer. 1988. Modeling nutrient behavior in wetlands. Ecol. Modelling 40:37-66.

Kadlec, R. H., R. B. Williams, and R. D. Scheffe. 1988. Wetland evapotranspiration in temperate and arid climates. Pages 146-160 in D. D. Hook, W. H. McKee, Jr., H. K. Smith, J. Gregory, V. G. Burrell, Jr., M. R. DeVoe, R. E. Sojka, S. Gilbert, R. Banks, L. H. Stolzy, D. Brooks, T. D. Matthews, and T. H. Shear (eds.), The Ecology and Management of Wetlands; Vol 1, Ecology of Wetlands. Croom Helm, London and Sydney.

Kaiser, M. S., and E. K. Fritzell. 1984. Effects of river recreationists on green-backed heron behavior. J. Wildl. Manage. 48:561-568.

Kaminski, R. M., and H. H. Prince. 1981. Dabbling duck and aquatic macroinvertebrate responses to manipulated wetland habitat. J. Wildl. Manage. 45:1-15.

Kantrud, H. A., and R. F. Stewart. 1984. Ecological distribution and crude density of breeding birds on prairie wetlands. J. Wildl. Manage. 48:426-437.

Kantrud, H. A., and R. E. Stewart. 1977. Use of natural basin wetlands by breeding waterfowl in North Dakota. J. Wildl. Manage. 41:243-253.

Kaplan, E., J. Walker, and M. Draus. 1974. Some effects of dredging on populations of macrobenthos organisms. Fish. Bull. 72:445-480.

Kaplan, W., L. Valiela, and J. M. Teal. 1979. Denitrification in a salt marsh ecosystem. Limnol. Oceanogr. 24:726-734.

Karickhoff, W. W., D. S. Brown, and T. A. Scott. 1979. Sorption of hydrophobic pollutants on natural sediments. Water Res. 13:241-248.

Karr, J. R., and R. R. Roth. 1971. Vegetation structure and avian diversity in several new world areas. Am. Nat. 105:423-435.

Karr, J. R., and I. J. Schlosser. 1977. Impact of nearstream vegetation and stream morphology on water quality and stream biota. EPA 600/3-77-097. US Environmental Protection Agency, Washington, DC. 103 p.

Karr, J. R., and I. J. Schlosser. 1978. Water resources at the land-water interface. Science 201:229-234.

Kaster, J. L., and G. Z. Jacobi. 1978. Benthic macroinvertebrates of a fluctuating reservoir. Freshwater Biol. 8:283-290.

Keddy, P. A. 1983. Shoreline vegetation in Axe Lake, Ontario: Effects of exposure on zonation patterns. Ecology 64:331-344.

Kelley, B. J., Jr., W. D. Marshall, H. N. McKellar, Jr., R. D. Porcher, and R. Zingmark. 1985. Primary production and community metabolism in coastal salt water impoundments (Abstr.). Estuaries 8:40A.

Kelley, J. C., T. M. Burton, and W. R. Enslin. 1984. The effects of natural water level fluctuations on N and P cycling in a Great Lakes marsh. Wetlands 4:159-176.

Kenworthy, W. J., J. C. Zieman, and G. W. Thayer. 1982. Evidence for the influence of seagrasses on the benthic nitrogen cycle in a coastal plain estuary near Beaufort, North Carolina (USA). Oecologia 54:152-158.

Kenworthy, W. J., and G. W. Thayer. 1984. Production and decomposition of roots and rhizomes of seagrasses, *Zostera marina* and *Thalassia testidum*, in temperate and subtropical marine ecosystems. Bull. Mar. Sci. 35:364-379.

Ketchum, B. H. 1967. Phytoplankton nutrients in estuaries. Pages 329-335 in G. H. Lauff (ed.), Estuaries. Am. Assoc. Advance. Sci., Washington, DC.

Kibby, H. V. 1978. Effects of wetlands on water quality. Pages 289-298 in R. R. Johnson and J. F. McCormick (tech. coord.), Strategies for protection and management of floodplain wetlands and other riparian ecosystems. Gen. Tech. Rep. No. CTR-WO-12. US For. Serv., Washington, DC.

Kimble, L. A., and T. A. Wesche. 1965. Relationships between selected physical parameters and benthic community structure in a small mountain stream. Water Resour. Ser. No. 55. Water Resour. Res. Inst., Raleigh, NC. 65 p.

King, D. L., and R. C. Ball. 1967. Comparative energetics of a polluted stream. Limnol. Oceanogr. 12:27-33.

King, G. M., M. J. Klug, R. G. Wiegert, and A. G. Chalmers. 1982. Relation of soil water movement and sulfide concentration to *Spartina alterniflora* production in a Georgia salt marsh. Science 218:61-63.

King, L. R. 1973. Comparison of the distribution of minnows and darters collected in 1947 and 1972 in Boone County, Iowa. Proceedings of the Iowa Academy of Science 80:133-135.

King, L. R., and K. D. Carlander. 1976. A study of the effects of stream channelization and bank stabilization on warmwater sport fish in Iowa; Subproject No. 3, Some effects of short-reach channelization on fishes and fish-food organisms in central Iowa warmwater streams. FWS/OBS-76/13. US Fish and Wildlife Service, Biological Services Program. 217 p.

Kiorboe, T. 1980. Distribution and production of submerged macrophytes in Tipper Grund (Ringkobing Fjord, Denmark), and the impact of waterfowl grazing. J. Appl. Ecol. 17:675-688.

Kirby-Smith, W. W. 1976. The detritus problem and the feeding and digestion of an estuarine organism. Pages 469-479 in M. Wiley (ed.), Estuarine Processes, Vol 1. Academic Press, New York.

Kistritz, R. U., K. J. Hall, and I. Yesaki. 1983. Productivity, detritus flux, and nutrient cycling in a *Carex lyngbyei* tidal marsh. Estuaries 6:227-236.

Kitchen, D. W., and G. W. Hunt. 1969. Brood habitat of the hooded merganser. J. Wildl. Manage. 33:605-609.

Kitchens, W. M., Jr., J. M. Dean, L. H. Stevenson, and J. H. Cooper. 1975. The Santee Swamp as a nutrient sink. Pages 349-366 in E. G. Howell, J. B. Gentry, and M. H. Smith (eds.), Mineral cycling in southeastern ecosystems. ERDA Symp. Series, CONF-740513.

Klapwijk, K., and W. J. Snodgrass. 1982. Experimental measurement of sediment nitrification and denitrification in Hamilton Harbour, Canada. Hydrobiol. 91:207-216.

Klett, A. T., T. L. Shaffer, and D. H. Johnson. 1988. Duck nest success in the prairie pothole region. J. Wildl. Manage. 52(3):431-440.

Klimas, C. V. 1987. Baldcypress response to increased water levels, Caddo Lake, Louisiana-Texas. Wetlands 7:25-38.

Klimas, C. V., C. O. Martin, and J. W. Teaford. 1981. Impacts of flooding regime modification on wildlife habitats of bottomland and hardwood forests in the lower Mississippi Valley. Tech. Rep. EL-81-13. US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Klingeman, P. C., and J. B. Bradley. 1976. Willamette River Basin streambank stabilization by natural means. Water Resour. Res. Inst., Oregon State Univ., Corvallis. 238 p.

Klopatek, J. M. 1975. The role of emergent macrophytes in mineral cycling in a freshwater marsh. Pages 367-393 in F. G. Howell, J. G. Gentry, and M. H. Smith (eds.), Mineral cycling in southeastern ecosystems. ERDA Symp. Series, CONF-740513.

Klopatek, J. M. 1978. Nutrient dynamics of freshwater riverine marshes and the role of emergent macrophytes. Pages 195-216 in R. E. Good, D. F. Whigham, and R. L. Simpson (eds.), Freshwater Wetlands. Academic Press, New York.

Knight, R. L., and S. K. Knight. 1984. Responses of wintering bald eagles to boating activity. J. Wildl. Manage. 48:999-1004.

Knight, R. L., B. R. Winchester, and J. C. Higman. 1984. Carolina Bays—Feasibility for effluent advanced treatment and disposal. Wetlands 4:177-204.

Knight, R. L. 1987. Effluent distribution and basin design for enhanced pollutant assimilation by freshwater wetlands. Pages 913-921 in K. R. Reddy and W. H. Smith (eds.), Aquatic Plants for Water Treatment and Resource Recovery. Magnolia Publishing, Inc.

Knopf, F. L. 1986. Changing landscapes and the cosmopolitism of the eastern Colorado avifauna. Wildl. Soc. Bull. 14:132-142.

Knowles, R. 1982. Denitrification. Microbiol. Rev. 46:43-70.

Knudsen, E. E., W. H. Herke, and J. M. Mackler. 1977. The growth rate of marked juvenile brown shrimp, *Penaeus aztecus*, in a semi-impounded Louisiana coastal marsh. Proc. Gulf Caribb. Fish. Inst. 29:144-159.

Knutson, P. L., J. C. Ford, and M. R. Inskeep. 1981. National survey of planted salt marshes (vegetative stabilization and wave stress). Wetlands 3:129-153.

Knutson, P. L., R. A. Brochu, W. N. Seelig, and M. Inskeep. 1982. Wave dampening in *Spartina alterniflora* marshes Wetlands 2:87-104.

Ko, W. H., and J. L. Lockwood. 1968. Conversion of DDT to DDD in soil and the effect of these compounds on soil organisms. Can. J. Microbiol. 14:1069-1073.

Koenings, J. P., and F. F. Hooper. 1976. The influence of colloidal organic matter on iron and iron phosphorus cycling in an acid bog lake. Limnol. Oceanogr. 21:684-696.

Kondratieff, P. F., and G. M. Simmons, Jr. 1984. Seston microbial activity in a river-reservoir system. J. Freshw. Ecol. 2:487-497.

Korschgen, C. E., L. S. George, and W. L. Green. 1985. Disturbance of diving ducks by boaters on a migrational staying area. Wildl. Soc. Bull. 13:290-296.

Kranck, K. 1984. The role of flocculation in the filtering of particulate matter in estuaries. Pages 159-175 in V. S. Kennedy, ed., The Estuary as a Filter. Academic Press, Inc., Orlando, FL.

Krapu, G. L. 1974. Foods of breeding pintails in North Dakota. J. Wildl. Manage. 38:408-417.

Krebs, C. T., and K. A. Burns. 1977. Long-term effects of an oil spill on populations of the salt-marsh crab, *Uca pugnax*. Science 197:484-487.

Krebs, C. T., and I. Valiela. 1978. Effects of experimentally applied chlorinated hydrocarbons on the biomass of the fiddler crab, *Uca pugnax*. Est. Coast. Mar. Sci. 6:375-386.

Krecker, F. H. 1939. A comparative study of the animal population of certain submerged aquatic plants. Ecology 20:553-562.

Kroodsma, D. E. 1979. Habitat values for nongame wetlands birds. Pages 320-329 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Kuenzler, E. J., and N. J. Craig. 1986. Land use and nutrient yields of the Chowan River watershed. Pages 77-107 in D. L. Correll (ed.), Watershed Research Perspectives. Smithsonian Institution Press, Washington, DC.

Kuenzler, E. J., P. J. Mulholland, L. A. Yarbro, and L. A. Smock. 1980. Distributions and budgets of carbon, phosphorus, iron and manganese in a floodplain swamp ecosystem. Rep. No. 157. Water Resour. Res. Inst., Univ. N. Carolina, Raleigh. 234 p.

Kushlan, J. A. 1974. Effects of a natural fish kill on the water quality, plankton, and fish population of a pond in the Big Cypress Swamp, Florida. Trans. Am. Fish. Soc. 103:235-243.

LaBaugh, J. W., T. C. Winter, V. A. Adomaitis, and G. A. Swanson. 1987. Hydrology and chemistry of selected prairie wetlands in the Cottonwood Lake area, Stutsman County, North Dakota. Prof. Paper 1431. US Geological Survey, Reston, VA. 10 p.

Lackey, T. B., G. B. Morgan, and O. H. Hart. 1959. Turbidity effects in natural waters in relation to organisms and the uptake of radioisotopes. Tech. Pap. 167 13(8). Univ. Fla. Eng. Indust. Exp. Stn. 9 p.

Lahti, T., and E. Ranta. 1985. The SLOSS principle and conservation practice: An example. Oikos 44:369-370.

Lahti, T., and E. Ranta. 1986. Island biogeography and conservation: A reply to Murphy and Wilcox. Oikos 47:388-389.

Lambou, V. M. 1985. Aquatic carbon and nutrient fluxes, water quality, and aquatic productivity in the Atchafalaya Basin, Louisiana. Pages 180-186 in Riparian Ecosystems and Their Management: Reconciling Conflicting Uses. USDA Forest Service Gen. Tech. Rep. RM-120. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Landers, J. L., A. S. Johnson, P. H. Morgan, and W. P. Baldwin. 1976. Duck foods in managed tidal impoundments in South Carolina. J. Wild. Manage. 40:721-728.

Landers, J. L., T. T. Fendley, and A. Johnson. 1977. Feeding ecology of wood ducks in South Carolina. J. Wildl. Manage. 41:118-127.

Lantz, K. E., J. T. Davis, J. S. Hughes, and H. E. Schafer, Jr. 1967. Water level fluctuation—Its effect on vegetation control and fish population management. Proc. S. E. Assoc. Game Fish Comm. 18:483-494.

Lantz, K. E., J. T. Orvis, J. S. Hughes, and H. E. Schafer. 1965. Water level fluctuation—Its effect on vegetation control and fish population management. Proc. Ann. Conf. Southeast. Assoc. Game Fish Comm. 25:570-583.

Larson, J. S. 1976. Models for assessment of freshwater wetlands. Rep. No. 32. Univ. of Mass. Water Resour. Res. Cent., Amherst. 91 p.

Laser, K. D., C. G. Rausch, C. L. Olson, and K. D. Carlewder. 1969. Fish distribution in the Skunk River below Ames, Iowa. Proceedings of the Iowa Academy of Science 76:196-204.

Lassen, J. J. 1975. The diversity of freshwater snails in view of the equilibrium theory of island biogeography. Oecologia 19:1-8.

Laxen, D. P. H., and R. M. Harrison. 1977. The highway as a source of water pollution: An appraisal with the heavy metal lead. Water Res. 11:1-11.

Lazrus, A. L., E. Larange, and J. P. Lodge, Jr. 1970. Lead and other metal ions in United States precipitation. Environmental Science and Technology 4:55-58.

Lee, G. F., E. Bentley, and R. Amundson. 1975. Effects of marshes on water quality. In A. D. Hasler (ed.), Coupling of Land and Water Systems. Springer-Verlag, New York.

Lee, R. 1980. Forest Hydrology. Columbia Univ. Press, New York. 349 p.

Lee, V., and S. Olsen. 1985. Eutrophication and management initiatives for the control of nutrient inputs to Rhode Island coastal lagoons. Estuaries 8:191-202.

Leidy, R. A., and P. L. Fiedler. 1985. Human disturbance and patterns of fish species diversity in the San Francisco Bay drainage, California. Biol. Conserv. 33:247-267.

Leitch, W. G., and R. M. Kaminski. 1985. Long-term wetland-waterfowl trends in Saskatchewan grassland. J. Wildl. Manage. 49(1)212-222.

Leonard, P. M., and D. J. Orth. 1986. Application and testing of an index of biotic integrity in small, coolwater streams. Trans. Am. Fish. Soc. 115:401-414.

Leopold, A. 1933. Game Management. Charles Scribner's Sons, New York. 481 p.

Leopold, L. B., and J. P. Miller. 1961. Ephemeral streams—Hydraulic factors and their relation to the drainage net. Prof. Pap. 282A. US Geological Survey, Reston, VA. 38 p.

Levy, D. A., and T. G. Northcote. 1981. The distribution and abundance of juvenile salmon in marsh habitats of the Fraser River estuary. Tech. Rep. 25. Westwater Research Center, Univ. British Columbia, Vancouver, Canada. 117 p.

Lewis, V. P., and D. S. Peters. 1984. Menhaden—A single step from vascular plant to fisheries harvest. J. Exp. Mar. Biol. Ecol. 84:95-100.

Lewke, R. E. 1975. Pre-impoundment study of vertebrate populations and riparian habitat behind Lower Granite Dam on the Snake River in Southeastern Washington. Ph.D. dissertation, Washington State Univ., Pullman, WA. 242 p.

Lichtler, W. F., G. H. Hughes, and F. L. Pfischner. 1976. Hydrologic relations between lake and aquifers in a recharge area near Orlando, FL. Water Resour.

Invest. 76-65. US Geological Survey, Reston, VA. 61 p.

Likens, G. E., and F. H. Bormann. 1974. Linkages between terrestrial and aquatic systems. Bioscience 24:447-456.

Lindau, C .W., R. D. DeLaune, and G. L. Jones. 1988. Fate of added nitrate and ammonium-nitrogen entering a Louisiana gulf coast swamp forest. J. Water Poll. Contr. Fed. 60(3):386-390.

Linthurst, R. A., and E. D. Seneca. 1980. Dieback of salt-water cordgrass (*Spartina alterniflora* Loisel) in the lower Cape Fear estuary of North Carolina: An experimental approach to reestablishment. Environ. Conserv. 7:59-66.

Lissey, A. 1968. Surficial mapping of groundwater flow systems with application to the Oak River Basin, Manitoba. Ph.D. dissertation, Univ. of Saskatchewan, Saskatoon. 141 p.

Lissey, A. 1971. Depression-focused transient groundwater flow patterns in Manitoba. Geol. Assoc. Can. Spec. Pap. No. 9:333-341.

Livingston, R. J. 1979. Food chain value. Pages 8-30 in J. R. Clark and J. E. Clark (eds.), Scientist's Report. The Conserv. Found. Nat. Wetl. Tech. Counc., Washington, DC.

Livingston, R. J., and O. L. Loucks. 1979. Productivity, trophic interactions, and food-web relationships in wetlands and associated systems. Pages 101-119 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Livingston, R. J. 1984. The ecology of the Apalachicola Bay system: An estuarine profile. FWS/OBS-82/05. US Fish Wildl. Serv. 148 p.

Lloyd, D. S. 1987. Turbidity as a water quality standard for salmonid habitats in Alaska. N. Am. J. Fish. Man. 7:34-45.

Lloyd, D. S., J. P. Koenings, and J. D. LaPerriere. 1987. Effects of turbidity in fresh waters of Alaska. N. Am. J. of Fish. Man. 7:18-33.

Lodge, D. M. 1985. Macrophyte-gastropod associations: Observations and experiments on macrophyte choice by gastropods. Freshw. Biol. 15:695-708.

Lodge, D. M., A. L. Beckel, and J. J. Magnuson. 1985. Lake bottom tyrant. Nat. Hist. 94:32-37.

Loeb, S. L., and C. R. Goldman. 1979. Water and nutrient transport via groundwater from Ward Valley into Lake Tahoe. Limnol. Oceanogr. 24:1146-1154.

Lonard, R. I., E. J. Clairain, Jr., R. T. Huffman, J. W. Hardy, L. D. Brown, P. E. Ballard, and J. W. Watts. 1981. Analysis of methodologies used for the assessment of wetlands values. US Water Resour. Counc., Washington, DC. 68 p.

Lonard, R. I., E. J. Clairain, Jr., R. T. Huffman, J. W. Hardy, L. D. Brown, P. E. Ballard, and J. W. Watts. 1984. Wetland functions and values study plan; Appendix A: Analysis of methodologies for assessing wetlands values. Technical Report Y-83-2. US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Lorenz, J. S., and D. D. Biesboer. 1987. Nitrification, denitrification, and ammonia diffusion in a cattail marsh. Pages 525-534 in K. R. Reddy and W. H. Smith (eds.), Aquatic Plants for Water Treatment and Resource Recovery. Magnolia Publishing, Inc.

Lotrich, V. A. 1973. Growth, production and community composition of fishes inhabiting a first-, second-, and third-order stream of eastern Kentucky. Ecol. Monogr. 43:337-397.

Loucks, O. L., and V. Watson. 1978. The use of models to study wetland regulation of nutrient loading to Lake Mendota. Pages 242-252 in C. B. DeWitt and E. Soloway (eds.), Wetlands: Ecology, values and impacts. Univ. Wis. Inst. Environ. Stud., Madison.

Lowe, R. L., S. W. Golladay, and J. R. Webster. 1986. Periphyton response to nutrient manipulation in streams draining clearcut and forested watersheds. J. N. Am. Benthol. Soc. 5(3):221-229.

Lowrance, R., J. K. Sharpe, and J. M. Sheridan. 1986. Long-term sediment deposition in the riparian zone of a coastal plain watershed. J. Soil Water Conserv. (Jul/Aug):266-271.

Lowrance, R., R. Todd, J. Fail, Jr., O. Hendrickson, Jr., R. Leonard, and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience 34:374-377.

Ludden, A. P., D. F. Frink, and D. H. Johnnson. 1983. Water storage capacity of natural wetland depressions in the Devils Lake Basin of North Dakota. J. Soil Water Conserv. 38:45-48.

Luey, J. E., and J. R. Adelman. 1980. Downstream natural areas as refuges for fish in drainage-development watersheds. Trans. Am. Fish. Soc. 109:332-335.

Lugo, A. E., S. Brown, and M. M. Brinson. 1988. Forested wetlands in freshwater and salt-water environments. Limnol. Oceanogr. 33(4):894-909.

Lyford, J. H., and S. V. Gregory. 1975. The dynamics and structure of periphyton communities in three Cascade Mountain streams. Int. Assoc. of Theor. Appl. Limnol. 19:1610-1616.

Lynch, J. F., and D. F. Whigham. 1984. Effects of forest fragmentation on breeding bird communities in Maryland, USA. Biol. Conserv. 28:287-324.

Maas, R. P., S. A. Dressing, J. Spooner, M. D. Smolen, and F. J. Humenik. 1984. Best Management Practices for Agricultural Nonpoint Source Control; IV, Pesticides. Biol. and Agric. Eng. Dept., North Carolina State Univ., Raleigh, NC.

Macan, T. T. 1949. A survey of a moorland fishpond. J. Anim. Ecol. 18:160-187.

MacArthur, R. H., and J. W. MacArthur. 1961. On bird species diversity. Ecology 42:594-598.

MacArthur, R. H., J. W. MacArthur, and J. Preer. 1964. On bird species diversity; II, Prediction of bird census from habitat measurements. Am. Nat. 96:167-171.

MacArthur, R. H., and E. O. Wilson. 1967. The theory of island biogeography. Princeton Univ. Press, Princeton, NJ. 203 p.

MacCrimmon, H. R. 1980. Nutrient and sediment retention in a temperate marsh ecosystem. Inst. Revue Ges. Hydrobiol. 65:719-744.

Macdonald, J. S., I. K. Birtwell, and G. M. Kruzynski. 1987. Food and habitat utilization by juvenile salmonids in the Campbell River estuary. Can. J. Fish. Aquat. Sci. 44:1133-1246.

Maciolek, J. A., and M. G. Tunzi. 1968. Microseston dynamics in a simple Sierra Nevada lake-stream system. Ecology 49:695-699.

Mack, G. D., and L. D. Flake. 1980. Habitat relationships of waterfowl broods on South Dakota stock ponds. J. Wildl. Manage. 44:695-699.

Mackay, R. J., and T. F. Waters. 1986. Effects of small impoundments on hydropsychid caddisfly production in Valley Creek, Minnesota. Ecol. 67(6):1680-1686.

Madsen, J. D., and M. S. Adams. 1988. The seasonal biomass and productivity of the submerged macrophytes in a polluted Wisconsin stream. Freshw. Biol. 20:41-50.

Magnuson, J. J., J. P. Baker, and E. J. Rahel. 1984. A critical assessment of effects of acidification on fisheries in North America. Philosophical Trans. Royal Soc. London-B, Biol. Sci. 305:501-516.

Mahaffy, L. A. 1987. Effects of open marsh water management on submerged aquatic vegetation utilized by waterfowl in Delaware. Pages 323-332 in W. R. Whitman and W. H. Meredith (eds.), Proceedings of a Symposium on Waterfowl and Wetlands Management in the Coastal Zone of the Atlantic Flyway. Delaware Dept. of Natural Resources and Environmental Control, Dover.

Major, D. W., C. I. Mayfield, and J. F. Barker. 1988. Biotransformation of benzene by denitrification in aquifer sand. Ground Water 26(1):8-14.

Mann, K. H. 1988. Production and use of detritus in various freshwater, estuarine, and coastal marine ecosystems. Limnol. Oceanogr. 33(4):910-930.

Marcus, M. D. 1980. Periphytic community response to chronic nutrient enrichment by a reservoir discharge. Ecology 61:387-399.

Maret, T. J., M. Parker, and T. E. Fannin. 1987. The effect of beaver ponds on the nonpoint source water quality of a stream in southwestern Wyoming. Wat. Res. 21(3):263-268.

Marinucci, A. C., and R. Bartha. 1982. A component model of decomposition of *Spartina alterniflora* in a New Jersey salt marsh. Can. J. Bot. 60:1618-1624.

Markham, B. J., and S. H. Brechtel. 1979. Status and management of three colonial waterbird species in Alberta. Proc. 1978 Conf. Colonial Waterbird Group 2:55-64.

Marron, D. C. 1987. Floodplain storage of metal-contaminated sediments downstream of a gold mine at Lead, South Dakota. Pages 193-209 in R. C. Averett and D. M. McKnight (eds.), Chemical Quality of Water and the Hydrologic Cycle. Lewis Publishers, Chelsea, MI.

Martens, L. A. 1968. Flood inundation and effects of urbanization in metropolitan Charlotte, North Carolina. Water-Supply Paper 1591-C. US Geological Survey, Reston, VA. 60 p.

Martin, A. C., H. S. Zim, and A. L. Nelson. 1951. American Wildlife and Plants: A Guide to Wildlife Food Habits. Dover Publications, New York. 500 p.

Martin, D. B., and W. A. Hartman. 1987. Correlations between selected trace elements and organic matter and texture in sediments of northern prairie wetlands. J. Assoc. Off. Anal. Chem. 70(5):916-919.

Martin, T. E. 1986. Competition in breeding birds: On the importance of considering processes at the level of the individual. Current Ornithology 4:181-210.

Martz, G. F. 1967. Effects of nesting cover reduction on breeding puddle ducks. J. Wildl. Manage. 31:236-247.

Marzolf, G. R. 1978. The potential effects of clearing and snagging on stream ecosystems. FWS/OBS-78/14. US Fish and Wildl. Serv., Washington, DC. 32 p.

Mason, C. F., and R. J. Bryant. 1975. Production, nutrient content and decomposition of *Phragmites communis* and *Typha angustifolia*. J. Ecol. 63:71-96.

Mathias, J. A., and J. Barica. 1980. Factors controlling oxygen depletion in ice-covered lakes. Can. J. Fish. Aquat. Sci. 37:185-194.

Matson, E. A., and R. L. Klotz. 1983. Organic carbon supply and demand in the Shetucket River of Eastern Connecticut. Pages 247-269 in T. D. Fontaine III and S. M. Bartell (eds.), Dynamics of Lotic Ecosystems. Ann Arbor Sci., Ann Arbor, MI.

Mattraw, H. C., Jr., and Elder, J. F. 1984. Nutrient and detritus transport in the Apalachicola River, Florida. Water-Supply Paper 2196-C. US Geological Survey, Reston, VA. 62 p.

Maxey, G. B. 1968. Hydrogeology of desert basins. Ground Water 6:10-22.

May, R. M. 1986. The search for patterns in the balance of nature: Advances and retreats. Ecology 67:1115-1126.

McBride, M. S., and H. O. Pfannkuch. 1975. The distribution of seepage within lakebeds. J. Res. US Geol. Surv. 3:505-512.

McCormick, J., and H. A. Somes. 1982. The coastal wetlands of Maryland. Coast. Zone Manage. Prog., Md. Dept. Nat. Resour., Annapolis.

McCrimmon, H. R. 1954. Stream studies on planted Atlantic salmon. J. Fish Res. Board Can. 11:362-403.

McDowell, D. M., and R. J. Naiman. 1986. Structure and function of a benthic invertebrate stream community as influenced by beaver. Oecologia 68:481-489.

McDowell, W. H., and G. E. Likens. 1988. Origin, composition, and flux of dissolved organic carbon in the Hubbard Brook Valley. Ecol. Monogr. 58(3):177-195.

McHugh, J. L. 1966. Management of estuarine fishes. Am. Fish. Soc. Spec. Publ. 3:133-154.

McIvor, C. C., and W. E. Odum. 1988. Food, refuges, and the associated risk of predation in structuring the fish assemblage of a tidal freshwater marsh. Ecology 69:1341-1351.

McIvor, C. C., L. P. Rozas, and W. E. Odum. 1989. Use of the marsh surface by fishes in tidal freshwater wetlands. In R. R. Sharitz and J. W. Gibbons (eds.), Freshwater Wetlands and Wildlife. US Dept. Energy, Off. Tech. Inform., Washington, DC.

McKay, E. D., A. Elzaftawy, and K. Cartwright. 1979. Groundwater of selected wetlands in Union and Alexander Counties, Illinois. Ill. Inst. Nat. Resour. Environ. Geol. Note No. 85, Urbana. 41 p.

McKee, K. L., and E. D. Seneca. 1982. The influence of morphology in determining the decomposition of two salt marsh macrophytes. Estuaries 5:302-309.

McKellar, H. N., Jr., B. J. Kelley, and R. G. Zingmark. 1987. Tidal nutrient exchange and primary production in South Carolina coastal impoundments. Pages 353-356 in W. R. Whitman and W. H. Meredith (eds.), Proceedings of a Symposium on Waterfowl and Wetlands Management in the Coastal Zone of the Atlantic Flyway. Delaware Dept. of Natural Resources and Environmental Control, Dover.

McNabb, C. D., Jr. 1976. The potential of submersed vascular plants for reclamation of wastewater in temperate zone ponds. Pages 123-132 in J. Tourbier and Pierson (eds.), Biological Control of Water Pollution. Univ. Pa. Press, Philadelphia.

McNaughton, S. J., and L. L. Wolf. 1973. General Ecology. Holt, Rinehart and Winston, New York. 710 p.

McPherson, B. F., and W. H. Sonntag. 1984. Transport and distribution of nutrients in the Loxahatchee River estuary, southeastern Florida, 1979-81. Water Resour. Bull. 20:27-34.

Meade, R. H. 1982. Sources, sinks, and storage of river sediment in the Atlantic drainage of the United States. J. of Geol. 90:235-252.

Mendall, H. L. 1949. Food habits in relation to black duck management in Maine. J. Wild. Manage. 31:64-101.

Mendall, H. L. 1958. The ring-necked duck in the Northeast. Univ. Maine Bull. 60(16). 320 p.

Mendelssohn, I. A., and E. D. Seneca. 1980. The influence of soil drainage on the growth of salt marsh cordgrass, *Spartina alterniflora*, in North Carolina. Est. Coast. Mar. Sci. 2:27-40.

Menzel, B. W., and H. L. Fierstine. 1976. A study of the effects of stream channelization and bank stabilization on warmwater sport fish in Iowa: Effects of long-reach stream channelization on distribution and abundance of fishes. FWS/OBS-76/15. US Fish Wildl. Serv., Washington, DC.

Menzie, C. A. 1980. The chironomid (Insecta:Diptera) and other fauna of a *Myriophyllum spicatum* L. plant bed in the lower Hudson River. Estuaries 3:38-54.

Meredith, W. H., and D. E. Saveikis. 1987. Effects of open marsh water management (OMWM) on bird populations of a Delaware tidal marsh, and OMWM's use in waterbird habitat restoration and enhancement. Pages 299-321 in W. R. Whitman and W. H. Meredith (eds.), Proceedings of a Symposium on Waterfowl and Wetlands Management in the Coastal Zone of the Atlantic Flyway. Delaware Dept. of Natural Resources and Environmental Control, Dover.

Meyboom, P. 1966. Unsteady groundwater flow near a willow ring in hummocky *moraine*. J. Hydrol. 4:38-62.

Meyboom, P. 1967. Mass-transfer studies to determine the groundwater regime of permanent lakes in hummocky moraine of western Canada. J. Hydrol. 5:117-142.

Mickle, A. M., and R. G. Wetzel. 1978. Effectiveness of submersed angiosperm-epiphyte complexes on exchange of nutrients and organic carbon in littoral systems. Aquat. Bot. 4:303-329.

Millar, J. B. 1971. Shoreline-area ratio as a factor in rate of water loss from small sloughs. J. Hydrol. 14:259-284.

Miller, E. G. 1965. Effect of Great Swamp, New Jersey, on streamflow during base-flow periods. Prof. Pap. 525-B. US Geological Survey, Reston, VA.

Miller, J. J., D. F. Acton, and R. J. St. Arnaud. 1985. The effect of groundwater on soil formation in a morainal landscape in Saskatchewan. Can. J. Soil Sci. 65:293-307.

Mills, D. H. 1971. Salmon and trout, a resource: Its ecology and management. St. Martin's Press, New York. 351 p.

Mills, J. G., and M. A. Zwarich. 1986. Transient groundwater flow surrounding a recharge slough in a till plain. Can. J. Soil Sci. 66:121-134.

Minello, T. J., R. J. Zimmerman, and E. X. Martinez. 1987. Fish predation on juvenile brown shrimp, *Penaeus aztecus* Ives: Effects of turbidity and substratum on predation rates. Fish. Bull. 85:59-70.

Minkley, W. L. 1963. The ecology of a spring stream, Doe Run, Meade County, Kentucky. Wildl. Monogr. 2:1-124.

Minshall, G. W. 1968. Community dynamics in a woodland spring brook. Hydrobiol. 32:305-339.

Minshall, G. W., R. C. Peterson, K. W. Cummins, T. L. Bott, J. R. Sedell, C. E. Cushing, and R. L. Vannote. 1983. Interbiome comparison of stream ecosystem dynamics. Ecol. Monogr. 53:1-25.

Minshall, G.W., R. C. Petersen, Jr., and C. F. Nimz. 1985. Species richness in streams of different size from the same drainage basin. Am. Nat. 125:16-38.

Mitsch, W. J., C. L. Dorge, and J. R. Wiemhoff. 1977. Forested wetlands for water resources management in southern Illinois. Ill. Univ. Water Resour. Cent., Urbana-Champaign.

Mitsch, W. J., and J. G. Gosselink. 1986. Wetlands. Van Nostrand Reinhold, New York. 537 p.

Mitsch, W. J., C. L. Dorge, and J. R. Wiemhoff. 1979. Ecosystem dynamics and a phosphorus budget of an alluvial cypress swamp in southern Illinois. Ecology 60:1116-1124.

Mittelbach, G. G. 1981. Foraging efficiency and body size: A study of optimal diet and habitat use by bluegills. Ecology 62:1370-1386.

Mock, C. R. 1967. Natural and altered estuarine habitats of penaeid shrimp. Gulf Caribb. Fish. Inst. 19:86-98.

Molinas, A., G. T. Auble, C. A. Segelquist, and L. S. Ischinger (eds.). 1988. Assessment of the role of bottomland hardwoods in sediment and erosion control. NERC-88/11. US Fish and Wildlife Service, National Ecology Research Center, Fort Collins, CO. 116 p.

Moller, A. P. 1987. Breeding birds in habitat patches: Random distribution of species and individuals? J. Biogeogr. 14:225-236.

Montague, C. L., A. V. Zale, and H. F. Percival. Ecological effects of coastal marsh impoundments: A review. Envir. Manage. 11(6):743-756.

Moore, I. D., and D. L. Larson. 1979. Effects of drainage projects on surface runoff from small depressional watersheds in the north-central region. Univ. Minn. Water Resour. Res. Cent. Bull. 99. 225 p.

Moore, I., and C. Larson. 1980. Hydrologic impact of draining small depressional watersheds. J. Irrig. Drain. Div. 196 (IR4):345-363.

Moore, J. W., V. A. Beaubien, and D. J. Sutherland. 1979. Comparative effects of sediment and water contamination on benthic invertebrates in four lakes. Bull. Environ. Contam. Toxicol. 23:840-847.

Moore, K. A. 1974. Carbon transport into York River, Virginia, tidal marshes. M.A. thesis, Univ. Va., Charlottesville. 102 p.

Morgan, M. D., and K. R. Philip. 1986. The effect of agricultural and residential development on aquatic macrophytes in the New Jersey pine barrens. Biol. Conserv. 35:143-158.

Morin, A., and R. H. Peters. 1988. Effect of microhabitat features, seston quality, and periphyton on abundance of overwintering black fly larvae in southern Quebec. Limnol. Oceanogr. 33(3):431-446.

Morin, P. J. 1984. The impact of fish exclusion on the abundance and species composition of larval odonates: Results of short-term experiments in a North Carolina farm pond. Ecology 65:53-60.

Morris, F. A., and L. J. Paulson. 1982. .A desert wetland created by wastewater flows: Current trends and problems. Wetlands 2:191-206.

Morris, J. T. 1982. A model of growth responses by *Spartina alterniflora* to nitrogen limitation. J. Ecol. 70:25-42.

Morris, J. T. 1984. Effects of oxygen and salinity on ammonium uptake by *Spartina alterniflora* Loisel and *Spartina patens* (Aitch.) Muhl. J. Exp. Mar. Biol. Ecol. 78:87-98.

Morris, J. T., and J. W. H. Dacey. 1984. Effects of oxygen on ammonium uptake and root respiration by *Spartina alterniflora*. Am. J. Bot. 71:979-985.

Morris, J. T., and W. B. Bowden. 1986. A mechanistic, numerical model of sedimentation, mineralization, and decomposition for marsh sediments. Soil Sci. Soc. Am. J. 50:96-105.

Morris, P. F., and W. G. Barker. 1977. Oxygen transport rates through mats of *Lemna minor* and *Wolffia* sp., and oxygen tension within and below the mat. Can. J. Bot. 55:1926-1932.

Morton, J. M., R. L. Kirkpatrick, and M. R. Vaughan. 1987. Wetland use by black ducks wintering at Chincoteague, Virginia. Pages 27-29 in W. R. Whitman and W. H. Meredith (eds.), Proceedings of a Symposium on Waterfowl and Wetlands Management in the Coastal Zone of the Atlantic Flyway. Delaware Dept. of Natural Resources and Environmental Control, Dover.

Moulton, J. C. 1970. The fishery potential of four aquatic environments created by Interstate Route 91 construction in Massachusetts. M.S. thesis, Univ. Ma., Amherst. 86 p.

Moyle, J. B. 1945. Some chemical factors influencing the distribution of aquatic plants in Minnesota. Am. Midl. Nat. 34:402-420.

Moyle, J. B. 1956. Relationships between chemistry of Minnesota surface waters and wildlife management. J. Wildl. Manage. 20:303-320.

Moyle, P. B., and J. J. Cech. 1982. Fishes: An Introduction to Ichthyology. Prentice-Hall, Englewood Cliffs, NJ. 593 p.

Mudroch, A., and J. A. Capobianco. 1979. Effects of treated effluent on a natural marsh. J. Water Pollut. Cont. Fed. 51: 2243-2256.

Mulholland, P. J., and E. J. Kuenzler. 1979. Organic carbon export from upland and forested watersheds. Limnol. Oceanogr. 35:960-966.

Mulholland, P. J. 1981. Organic carbon flow in a swamp stream ecosystem. Ecol. Monog. 51:307-322

Muncy, R. J., G. J. Atchison, R. V. Bulkley, B. W. Menzel, L. G. Perry, and R. C. Summerfelt. 1979. Effects of suspended solids and sediment on reproduction and early life of warmwater fishes: A review. EPA-600/3-79-042. US Environmental Protection Agency, Washington, DC. 110 p.

Mundorff, J. B. 1950. Flood-plain deposits of North Carolina piedmont and mountain streams as a possible source of ground-water supply. Bull. 59, Div. Miner. Resour. N.C. Dept. Conserv. Dev. 20 p.

Murgatroyd, A. L., and J. L. Ternan. 1983. The impact of the afforestation on stream bank erosion and channel form. Earth Surface Processes and Landforms 8:357-369.

Murkin, H. R., R. M. Kaminski, and R. D. Titman. 1982. Responses by dabbling ducks and aquatic invertebrates to an experimentally manipulated cattail marsh. Can. J. Zool. 60:2324-2332.

Murkin, H. R., and J. A. Kadlec. 1986. Responses by benthic macroinvertebrates to prolonged flooding of marsh habitat. Can. J. Zool. 64:65-72.

Murphy, G. I. 1962. Effect of mixing depth and turbidity on the productivity of fresh-water impoundments. Trans. Am. Fish. Soc. 91:69-76.

Murphy, M. L., C. P. Hawkins, and N. H. Anderson. 1981. Effects of canopy modification and accumulated sediment on stream communities. Trans. Am. Fish. Soc. 110:469-478.

Murphy, M. L., and J. D. Hall. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams in the Cascade Mountains, Oregon. Can. J. Fish. Aquat. Sci. 38:137-145.

Murphy, T. P., and B. G. Brownlee. 1981. Ammonia volatilization in a hypertrophic prairie lake. Can. J. Fish. Aquat. Sci. 38:1035-1039.

Murray, R. E., and R. E. Hodson. 1984. Microbial biomass and utilization of dissolved organic matter in the Okefenokee Swamp ecosystem. Appl. Environ. Microbiol. 47:685-692.

Mussallem, K. E., and J. A. Lynch. 1980. Controlling nonpoint source pollution from commercial clearcuts. ASCE Symp. Watershed Manage. 2:669-681.

Naiman, R. J. 1983. The annual pattern and spatial distribution of aquatic oxygen metabolism in boreal forest watersheds. Ecol. Monogr. 53:73-94.

Naiman, R. J., and J. R. Sedell. 1979. A study of benthic organic matter as a function of stream order. Archiv. Hydrobiol. 87:404-472.

Naiman, R. J., and J. R. Sedell. 1980. Relationships between metabolic parameters and stream order in Oregon. Can. J. Fish. Aquat. Sci. 37:834-847.

Nakashima, B. S., and W. C. Leggett. 1980. The role of fishes in the regulation of phosphorus availability in lakes. Can. J. Fish. Aquat. Sci. 37:1540-1549.

Nanson, G. C. 1980. Point bar and floodplain formation of the meandering Beatton River, northeastern British Columbia, Canada. Sedimentology 27:3-29.

Nanson, G. C., and E. J. Hickin. 1986. A statistical analysis of bank erosion and channel migration in western Canada. Geol. Soc. Am. Bull. 97:497-504.

Nash, R. D. M. 1988. The effects of disturbance and severe seasonal fluctuations in environmental conditions on north temperate shallow-water fish assemblages. Estuarine, Coastal and Shelf Science 26:123-135.

Nedwell, D. B. 1975. Inorganic nitrogen metabolism in a eutrophicated tropical mangrove estuary. Water Res. 9:221-231.

Neill, C., and R. E. Turner. 1987. Comparison of fish communities in open and plugged backfilled canals in Louisiana coastal marshes. N. Am. J. Fish. Man. 7:57-62.

Nelson, J. W., and J. A. Kadlec. 1984. A conceptual approach to relating habitat structure and macroinvertebrate production in freshwater wetlands. Trans. N. Am. Wildl. Nat. Resour. Conf. 49:262-269.

Nessel, J. K., and S. E. Bayley. 1984. Distribution and dynamics of organic matter and phosphorus in a sewage-enriched cypress swamp. Pages 262-278 in K. C. Ewel and H. T. Odum (eds.), Cypress Swamps. Univ. Florida Press, Gainesville.

Newbold, J. D., D. C. Erman, and K. B. Roby. 1980. Effects of logging on macroinvertebrates in streams with and without buffer strips. Can. J. Fish. Aquat. Sci. 37:1076-1085.

Newell, R. 1965. The role of detritus in the nutrition of two marine deposit feeders, the prosobranch *Hydrobia ulvae* and the bivalve *Macoma balthica*. Proc. Zool. Soc. London 144:25-45.

Nichols, D. S. 1983. Capacity of natural wetlands to remove nutrients from wastewater. J. Water Pollut. Control Fed. 55:495-505.

Niemeier, P. E., and W. A. Hubert. 1986. The 85-year history of the aquatic macrophyte species composition in a eutrophic prairie lake (United States). Aquat. Bot. 25:83-89.

Nishio, T., I. Koike, and A. Hattori. 1982. Denitrification, nitrate reduction, and oxygen consumption in coastal and estuarine sediments. Appl. Environ. Microbiol. 43:648-653.

Nishio, T., I. Koike, and A. Hattori. 1983. Estimates of denitrification and nitrification in coastal and estuarine sediments. Appl. Environ. Microbiol. 45:444-450.

Nixon, S. W. 1980. Between coastal marshes and coastal waters: A review of twenty years of speculation and research on the role of salt marshes in estuarine productivity and water chemistry. Pages 437-525 in P. Hamilton and K. B. MacDonald (eds.), Estuarine and Wetland Processes. Plenum Publishing, New York.

Nixon, S. W. 1981. Freshwater inputs and estuarine productivity. Pages 31-557 in R. D. Cross and D. L. Williams (eds.), Proc. Nat. Symp. on Freshwater Inflow to Estuaries. FWS/OBS-81-04. US Fish Wildl. Serv., Washington, DC.

Nixon, S. W. 1982. The ecology of New England high salt marshes: A community profile. FWS/OBS-81/55. US Fish Wildl. Serv., Washington, DC. 70 p.

Nixon, S. W., and C. A. Oviatt. 1973. Ecology of a New England salt marsh. Ecol. Monogr. 43(4):463-498.

Nixon, S. W., and V. Lee. 1986. Wetlands and water quality: A regional review of recent research in the United States on the role of freshwater and salt water wetlands as sources, sinks, and transformers of nitrogen, phosphorus, and various heavy metals. Tech. Rep. Y-86-2, prepared by University of Rhode Island for US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Nixon, S. W. 1988. Physical energy inputs and the comparative ecology of lake and marine ecosystems. Limnol. Oceanogr. 33(4):1005.

Nolen, S. L., J. Wilhm, and G. Howick. 1985. Factors influencing inorganic turbidity in a Great Plains reservoir. Hydrobiol. 123:109-117.

Novitzki, R. P. 1978. Hydrology of the Nevin wetland near Madison, Wisconsin. US Geol. Surv. Water Resour. Invest. 78-43. 25 p.

Novitzki, R. P. 1979. The hydrologic characteristics of Wisconsin wetlands and their influence on floods, streamflow, and sediment. Pages 377-388 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Novitzki, R. P. 1981. Hydrology of Wisconsin wetlands. Univ. Wis. Exten., Geol. Nat. Hist. Surv. Inform. Circ. 40, Madison. 30 p.

Novotny, V. 1980. Delivery of suspended sediment and pollutants from nonpoint sources during overland flow. Water Resour. Bull. 16(6):1057-1065.

Nunnally, N., and E. Keller. 1979. Use of fluvial processes to minimize adverse effects of stream channelization. Water Resour. Res. Inst. Rep. No. 144. Univ. NC, Chapel Hill. 115 p.

NY State Department of Environmental Conservation. 1980. Freshwater wetlands maps and classification regulations; 6NYCRR, Part 664.

O'Brien, A. L. 1977. Hydrology of two small wetland basins in eastern Massachusetts. Water Res. Bull. 13:325-341.

Oberts, G. L. 1981. Impact of wetlands on watershed water quality. Pages 213-227 in B. Richardson (ed.), Selected proceedings of the Midwest conference on wetland values and management. Freshw. Soc., Navarre, MN.

Odum W. E., and J. E. Drifmeyer. 1978. Sorption of pollutants by plant detritus: A review. Envir. Health Perspect. 27:133-137.

Odum, E. P. 1971. Fundamentals of Ecology, 3rd ed. W. B. Saunders Co., Philadelphia, PA. 574 p.

Odum, E. P. 1979. Ecological importance of the riparian zone. Pages 2-4 in R. R. Johnson and J. F. McCormick (tech. coords.), Strategies for protection and management of floodplain wetlands and other riparian ecosystems. Gen. Tech. Rep. WO-12. US For. Serv., Washington, DC.

Odum, E. P. 1980. The status of three ecosystem-level hypotheses regarding salt marsh estuaries: Tidal subsidy, outwelling and detritus-based flood chains. Pages 485-495 in V. S. Kennedy (ed.), Estuarine Perspectives. Academic Press, New York.

Odum, E. P. 1981. A functional classification of wetlands. Pages 4-10 in Coastal ecosystems of the southeastern United

States. FWS/OBS 80/59. US Fish Wildl. Serv., Washington, DC.

Odum, E. P., and A. A. de la Cruz. 1967. Particulate organic detritus in a Georgia salt marsh estuarine ecosystem. Pages 383-388 in G. H. Lauff (ed.), Estuaries. Publ. No. 83. Am. Assoc. Ad. Sci., Washington, DC.

Odum, E. P. 1985. Trends expected in stressed ecosystems. Bioscience 35:419-422.

Odum, H. T. 1963. Productivity measurements in Texas turtle grass and the effects of dredging in an intracoastal channel. Publ. Inst. Mar. Sci. Texas 9:48-58.

Odum, H. T. 1978. Principles for interfacing wetlands with development. Pages 29-56 in M. A. Drew (ed.), Environmental quality through wetlands utilization. Coord. Counc. Kissimmee R. Val. Taylor Cr.-Nubbin Slough Bas., Tallahassee, FL.

Odum, W. E. 1970. Utilization of the direct grazing and plant detritus food chains by the striped mullet *Mulgil cephalus*. Pages 222-240 in J. H. Steele (ed.), Marine food chains. Univ. Calif. Press, Berkeley.

Odum, W. E. 1970. Insidious alteration of the estuarine environment. Trans. Am. Fish. Soc. 99:836-847.

Odum, W. E. 1984. Estuarine productivity: Unresolved questions concerning the coupling of primary and secondary production. Pages 231-253 in B. J. Copeland, K. Hart, and S. Friday (eds.), Research for Managing the Nation's Estuaries. Univ. NC Sea Grant, Raleigh.

Odum, W. E., and E. J. Heald. 1975. The detritus-based food web of an estuarine mangrove community. Pages 265-286 in L. E. Cronin (ed.), Estuarine Research, Vol 1. Academic Press, New York.

Odum, W. E., and M. A. Heywood. 1978. Decomposition of intertidal freshwater marsh plants. Pages 89-97 in R. Good, D. Whigham, and R. Simpson (eds.), Freshwater Wetlands. Academic Press, New York.

Odum, W. E., and T. T. Smith. 1981. Ecology of tidal, low salinity ecosystems. Pages 36-40 in Coastal ecosystems of the southeastern United States. FWS/OBS-80/59. US Fish Wildl. Serv., Washington, DC.

Odum, W. E., J. C. Zieman, and E. J. Heald. 1972. The importance of vascular plant detritus to estuaries. Pages 91-114 in R. H. Chabreck (ed.), Coastal marsh and estuary management. La. State Univ., Baton Rouge.

Odum, W. E., M. L. Dunn, and T. J. Smith III. 1978. Habitat value of tidal freshwater wetlands. Pages 248-255 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Odum, W. E., J. S. Fisher, and J. C. Pickral. 1979. Factors controlling the flux of particulate organic carbon from estuarine wetlands. Pages 69-80 in R. J. Livingston (ed.), Ecological Processes in Coastal and Marine Systems. Mar. Sci. Ser. No. 10. Plenum Press, New York.

Odum, W. E., T. J. Smith III, J. K. Hoover, and C. C. McIvor. 1984. The ecology of tidal freshwater marshes of the United States east coast: A community profile. FWS/OBS-83/17. US Fish Wildl. Serv., Washington, DC. 177 p.

Oetting, R. B., and J. F. Cassel. 1971. Waterfowl nesting on interstate highway right-of-way in North Dakota. J. Wildl. Manage. 35:774-781.

Ogan, M. T. 1983. Factors affecting nitrogenase activity associated with marsh grasses and their soils from eutrophic lakes. Aquat. Bot. 17:215-230.

Ogawa, H., and J. W. Male. 1983. The flood mitigation potential of inland wetlands. Univ. Ma. Water Resour. Res. Cent., Amherst.

Ohlendorf, H. M., R. L. Hothem, C. M. Bunck, T. W. Aldrich, and J. F. Moore. 1986. Relationships between selenium concentrations and avian reproduction. Trans. 51st N. Am. Wildl. Nat. Res. Conf. 330-343.

Olsen, L. A. 1984. Effects of contaminated sediment on fish and wildlife: Review and annotated bibliography. FWS/OBS-82/66. US Fish Wildl. Serv., Washington, DC. 103 p.

Omernik, J. M., A. R. Abernathy, and L. M. Male. 1981. Stream nutrient levels and proximity of agricultural and forest land to streams: Some relationships. J. Soil Water Conserv. (Jul/Aug):227-231.

Onuf, C. P., M. L. Quammen, G. P. Shaffer, C. H. Peterson, J. W. Chapman, J. Cermak, and R. W. Holmes. 1979. An analysis of the values of central and southern California coastal wetlands. Pages 186-199 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Onuf, C. P., J. M. Teal, and I. Valiela. 1977. Interactions of nutrients, plant growth, and herbivory in a mangrove ecosystem. Ecology 58:514-526.

Osenga, G. A., and B. C. Coull. 1983. *Spartina alterniflora* Loisel root structure and meiofaunal abundance. J. Exp. Mar. Biol. Ecol. 67:221-225.

Osterkamp, W. R., and E. R. Hedman. 1982. Perennial-streamflow characteristics related to channel geometry and sediment in Missouri River Basin. Prof. Paper 1242. US Geological Survey, Reston, VA. 20 p.

Ostry, R. C. 1982. Relationship of water quality and pollutant loads to land uses in adjoining watersheds. Water Res. Bull. 18:99-104.

Owens, N. J. P., and W. D. P. Stewart. 1983. Enteromorpha and the cycling of nitrogen in a small estuary. Est. Coast. Shelf Sci. 17:287-296.

Page, G. W., L. E. Stenzel, and C. M. Wolfe. 1979. Aspects of the occurrence of shorebirds on a central California estuary. Stud. Avian Biol. 2:15-32.

Palmisano, A. W. 1973. Habitat preference of waterfowl and fur animals in the northern Gulf coast marshes. Pages 163-190 in R. H. Chabreck (ed.), Proc. coastal marsh and estuary management symposium. La. State Univ., Baton Rouge.

Paragamian, V. L., and M. J. Wiley. 1987. Effects of variable streamflows on growth of smallmouth bass in the Maquoketa River, Iowa. N. Am. J. Fish. Manage. 7:357-362.

Pardue, G. B. 1983. Habitat suitability index models: Alewife and blueback herring. FWS/OBS-82/10.58. US Fish Wildl. Serv., Washington, DC. 22 p.

Parnell, J. F., and T. L. Quay. 1962. The populations, breeding biology, and environmental relations of the black duck, gadwall, and blue-winged teal and Pea and Bodie Islands, North Carolina. Proc. S. E. Assoc. Game Fish Comm. 16:53-66.

Parr, J. F., and S. Smith. 1976. Degradation of toxaphene in selected anaerobic soil environments. Soil Sci. 121:52-57.

Patric, J. H., J. O. Evans, and J. D. Helvey. 1984. Summary of sediment yield data from forested land in the United States. J. For. 82:101-103.

Patrick, R. 1967. Diatom communities in estuaries. Pages 311-315 in G. H. Lauff (ed.), Estuaries. American Association for the Advancement of Science, Washington, DC.

Patrick, W. H., Jr., and R. D. DeLaune. 1976. Nitrogen phosphorus utilization by *Spartina alterniflora* in a salt marsh in Barataria Bay, Louisiana. Est. Coast. Mar. Sci. 4:59-64.

Patrick, W. H., R. DeLaune, R. M. Engler, and S. Gotoh. 1976. Nitrate removal from water at the water-mud interface in wetlands. EPA 600/3-76-042. US Environmental Protection Agency, Corvallis, OR. 79 p.

Patterson, J. H. 1976. The role of environmental heterogeneity in the regulation of duck populations. J. Wildl. Manage. 40:22-32.

Patton, P., and V. Baker. 1976. Morphometry and floods in small drainage basins subject to diverse hydrogeomorphic controls. Water Resour. Res. 12:941-952.

Paulet, M., H. Kohnke, and L. J. Lund. 1972. An interpretation of reservoir sedimentation: Effect of watershed characteristics. J. Environ. Qual. 1:146-150.

Peddicord, R. K., and V. A. McFarland. 1978. Effects of suspended dredged material on aquatic animals. Tech. Rep. D-78-29. US Army Engineer Waterways Experiment Station, Vicksburg, MS. 118 p.

Pedersen. E. R., and M. A. Perkins. 1986. The use of benthic invertebrate data for evaluating impacts of urban runoff. Hydrobiol. 139:13-22.

Pennak, R. W., and E. D. Van Gerpen. 1947. Bottom fauna production and physical nature of the substrate in a northern Colorado trout stream. Ecology 28:42-48.

Percival, E., and H. Whitehead. 1929. A quantitative study of some types of streambeds. J. Ecol. 17:282-314.

Perry, S. A., and A. L. Sheldon. 1986. Effects of exported seston on aquatic insect faunal similarity and species richness in lake outlet streams in Montana, USA. Hydrobiol. 137:65-77.

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

Peterken, O. F. 1974. A method for assessing woodland flora for conservation using indicator species. Biol. Conserv. 6:239-245.

Peterson, B. J., and B. Fry. 1987. Stable isotopes in ecosystem studies. Ann. Rev. Ecol. Syst. 18:293-320.

Peterson, B. J., R. W. Howarth, and R. H. Garritt. 1985. Multiple stable isotopes used to trace the flow of organic matter in estuarine food webs. Science 227:1361-1363.

Peterson, C. H., and N. M. Peterson. 1979. The ecology of intertidal flats on North Carolina: A community profile. FWS/OBS-79/39. US Fish Wildl. Serv., Washington, DC. 73 p.

Peterson, C. H. 1981. The ecological role of mud flats in estuarine systems. Pages 184-191 in Coastal ecosystems of the southeastern United States. FWS/OBS-80/59. US Fish Wildl. Serv., Washington, DC.

Peterson, N. P. 1982. Immigration of juvenile coho salmon into riverine ponds. Can. J. Fish. Aquat. Sci. 39:1308-1310.

Peterson, N. P. 1982. Population characteristics of juvenile coho salmon overwintering in riverine ponds. Can. J. Fish. Aquatic Sci. 39:1303-1307.

Peterson, S. R., and J. B. Low. 1977. Waterfowl use of Uinta Mountain wetlands in Utah. J. Wildl. Manage. 41:112-117.

Pethick, J. S. 1981. Long-term accretion rates on tidal salt marshes. J. Sediment. Petrol. 51:571-577.

Petryk. D., and G. Bosmajian III. 1975. Analysis of flow through vegetation. Proc. Am. Soc. Civil Eng., J. Hydr. Div. 101(HY7): 871-884.

Peverly, J. H. 1985. Element accumulation and release by macrophytes in a wetland stream. J. Environ. Qual. 14:137-143.

Pfankuch, D. J. 1975. Stream reach inventory and channel stability evaluation: A watershed management procedure. US For. Serv., N. Reg., Missoula, MT. 25 p.

Phillips, G. L., D. Eminson, and B. Moss. 1978. A mechanism to account for macrophyte decline in progressively eutrophicated freshwaters. Aquat. Bot. 4:103-126.

Phillips, R. C. 1980. Role of seagrass in estuarine systems. Pages 67-96 in Proc. Gulf of Mexico coastal ecosystems workshop. FWS/OBS-80/30. US Fish Wildl. Serv., Washington, DC.

Phillips, R. C. 1984. The ecology of eelgrass meadows in the Pacific Northwest: A community profile. FWS/OBS-84/24. US Fish and Wildlife Service, Office of Biological Services, Washington, DC.

Phung, H. T., and E. B. Knipling. 1976. Photosynthesis and transpiration of citrus seedlings under flood conditions. Hort. Sci. 11:131-133.

Pickral, J. C., and W. E. Odum. 1977. Benthic detritus in a saltmarsh tidal creek. Pages 280-292 in M. Wiley (ed.), Estuarine Processes, Vol 2. Academic Press, New York.

Pieczynska, E. 1986. Sources and fate of detritus in the shore zone of lakes. Aquat. Bot. 25:153-166.

Piest, L. A., and L. K. Sowls. 1985. Breeding duck use of a sewage marsh in Arizona. J. Wildl. Manage. 49:580-585.

Pionke, H. B., and G. Chesters. 1973. Pesticide-sediment-water interactions. J. Envir. Qual. 2:29-45.

Pionke, H. B., R. R. Schnabel, J. R. Hoover, W. J. Gburek, J. B. Urban, and A. S. Rogowski. 1986. Mahantango Creek watershed—fate and transport of water and nutrients. Pages 108-134 in D. L. Correll (ed.), Watershed Research Perspectives. Smithsonian Institution Press, Washington, DC.

Pitt, R., and M. Bozeman. 1982. Sources of urban runoff pollution and its effects on an urban creek. EPA-600/S2-82-090. US Environmental Protection Agency, Municipal Environmental Research Lab., Cincinnati, OH. 7 p.

Platts, W. S., M. A. Shirazi, and D. H. Lewis. 1979. Sediment particle sizes used by salmon for spawning with methods for evaluation. EPA-600/3-79-043. US Environmental Protection Agency, Corvallis, OR.

Ploskey, G. R., J. M. Nestler, and L. R. Aggus. 1984. Effects of water levels and hydrology on fisheries in hydropower storage, hydropower mainstream and flood control reservoirs. Tech. Rep. E-84-8. US Army Engineer Waterways Experiment Station, Vicksburg, MS.
50 p + append.

Poe, T. P., C. O. Hather, C. L. Brown, and D. W. Schloesser. 1986. Comparison of species composition and richness of fish assemblages in altered and unaltered littoral habitats. J. Freshw. Ecol. 3:525-536.

Pollard, J. E., S. M. Melancon, and L. S. Blakey. 1983. Importance of bottomland hardwoods to crawfish and fish in the Henderson Lake area, Atchafalaya Basin, Louisiana. Wetlands 3:1-25.

Polunin, N. V. C. 1984. The decomposition of emergent macrophytes in freshwater. Advan. Ecol. Res. 14:115-166.

Pomerantz, G. A., D. J. Decker, G. R. Goff, and K. G. Purdy. 1988. Assessing impact of recreation on wildlife: A classification scheme. Wildl. Soc. Bull. 16(1):58-62.

Pomeroy, L. R., and R. G. Wiegert, ed. 1981. The ecology of a salt marsh. Springer-Verlag, New York. 271 p.

Portnoy, J. W. 1978. Colonial waterbird population status and management on the north Gulf of Mexico coast. Proc. 1977 Conf. Colonial Waterbird Group 1:38-43.

Portnoy, J. W., C. T. Roman, and M. A. Soukup. 1987. Hydrologic and chemical impacts of diking and drainage of a small estuary (Cape Cod National Seashore): Effects on wildlife and fisheries. Pages 254-267 in W. R. Whitman and W. H. Meredith (eds.), Proceedings of a Symposium on Waterfowl and Wetlands Management in the Coastal Zone of the Atlantic Flyway. Delaware Dept. of Natural Resources and Environmental Control, Dover.

Portt, C. B., E. K. Balon, and D. L. G. Noakes. 1986. Biomass and production of fishes in natural and channelized streams, Can. J. Fish. Aquat. Sci. 43:1926-1934.

Possardt, E. E., W. E. Dodge, W. P. MacConnell, and R. J. Reed. 1976. Channelization assessment, White River, Vermont: Remote sensing, benthos, and wildlife. FWS/OBS-76/07. US Fish and Wildlife Service, Office of Biological Services. 73 p.

Post, H. A., and A. A. del la Cruz. 1977. Litterfall, litter decomposition and flux of particulate organic material in a coastal plain stream. Hydrobiol. 55:201-207.

Powell, G. V. N. 1987. Habitat use by wading birds in a subtropical estuary: Implications of hydrography. The Auk 104:740-749.

Power, M. E., and W. J. Matthews. 1983. Algae-grazing minnows (*Campostoma anomalum*), piscivorous bass (*Micropterus* sp.) and the distribution of attached algae in a small prairie-margin stream. Oecologia 60:328-332.

Prairie, Y. T., and J. Kalff. 1988. Dissolved phosphorus dynamics in headwater streams. Can. J. Fish. Aquat. Sci. 45:200-209.

Prairie, Y., and J. Kalff. 1986. Effect of catchment size on phosphorus export. Water Resour. Bull. 22(2):465-470.

Prairie, Y. T., and J. Kalff. 1988. Particulate phosphorus dynamics in headwater streams. Can. J. Fish. Aquat. Sci. 45:210-215.

Pratt, J. M., and R. A. Coler. 1979. Ecological effects of urban stormwater runoff on benthic macroinvertebrates inhabiting the Green River, Massachusetts. Publ. No. 100. Water Resources Res. Center, Univ. Massachusetts, Amherst. 75 p.

Pregnall, A. M. 1983. Release of dissolved organic-carbon from estuarine intertidal macroalga *Enteromorpha prolifera*. Mar. Biol. 73:37-42.

Prentki, R. T., T. D. Gustafson, and M. S. Adams. 1978. Nutrient movements in lakeshore marshes. Pages 169-194 in R. E. Good, D. F. Whigham, and R. L. Simpson (eds.), Freshwater Wetlands. Academic Press, New York.

Prescott, F. W. 1968. The Algae. Houghton Mifflin Co., Boston. 195 p.

Prevost, M. B. 1987. Management of plant communities for waterfowl in coastal South Carolina. Pages 168-182 in W. R. Whitman and W. H. Meredith (eds.), Proceedings of a Symposium on Waterfowl and Wetlands Management in the Coastal Zone of the Atlantic Flyway. Delaware Dept. of Natural Resources and Environmental Control, Dover.

Price, M. 1977. Techniques for estimating flood-depth frequency relations on natural streams in Georgia. Water Resour.
Investig. Rep. 77-90. US Geol. Surv., Doraville, GA.

Prince, H. H. 1968. Nest sites used by wood ducks and common goldeneyes in New Brunswick. J. Wild. Manage. 32:489-500.

Prinslow, T. E., I. Valiela, and J. M. Teal. 1974. The effect of detritus and ration size on the growth of *Fundulus heteroclitus* (L.). J. Exp. Mar. Biol. Ecol. 16:1-10.

Prouse, N. J. 1986. Distribution and abundance of mysids in the Cumberland Basin, upper Bay of Fundy. Proc. N. S. Inst. Sci. 36:1-11.

Quennerstadt, N. 1958. Effect of water level fluctuation on lake vegetation. Verh. Internat. Verein. Limnol. 13:901-906.

Rabalais, N. N. 1980. Ecological values of selected coastal habitats. Pages 191-209 in Proc. Gulf of Mexico coastal ecosystems workshop. FWS/OBS-80/30. US Fish Wildl. Serv., Washington, DC.

Rabe, F. W., and F. Gibson. 1984. The effect of macrophyte removal on the distribution of selected invertebrates in a littoral environment. J. Freshw. Ecol. 2:359-366.

Rabeni, C. F., and G. W. Minshall. 1977. Factors affecting microdistribution of stream benthic insects. Oikos 29:33-43.

Racey, G. D., and D. L. Euler. 1982. Small mammal and habitat response to shoreline cottage development in central Ontario. Can. J. Zool. 60:865-880.

Rahel, F. S. 1984. Facors structuring fish assemblages along a bog lake successional gradient. Ecology 65:1276-1289.

Raleigh, R. F., and P. C. Nelson. 1985. Habitat suitability index models: Pink salmon. Biol. Rep. 82 (10.109). US Fish Wildl. Serv., Fort Collins, CO. 36 p.

Rannie, W. F. 1980. The Red River flood control system and recent flood events. Water Resour. Bull. 16:207-214.

Rappaport, R. A., N. R. Urban, P. S. Capel, J. E. Baker, B. B. Looney, S. J. Eisenreich, and E. Gorham. 1985. "New" DDT inputs to North America: Atmospheric deposition. Chemosphere 14:1167- 1173.

Rausch, D. L., and J. D. Schreiber. 1977. Callahan Reservoir; I. Sediment and nutrient trap efficiency. Trans. ASAE 20:281-284; 290.

Raveling, D. G. 1977. Canada geese of the Churchill River Basin in north-central Manitoba. J. Wildl. Manage. 41:35-47.

Ream, C. H. 1980. Impacts of backcountry recreationists on wildlife: An annotated bibliography. Gen. Tech. Rep. INT-81. USDA, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT. 62 p.

Reddy, K. R. 1981. Diel variations of certain physico-chemical parameters of water in selected aquatic system. Hydrobiol. 85:201-207.

Reddy, K. R., and D. L. Sutton. 1984. Water hyacinths for water quality improvement and biomass production. Journal of Environmental Quality 13:1-8.

Reddy, K. R., and W. H. Patrick. 1975. Effect of alternate aerobic and anaerobic conditions on redox potential, organic matter decomposition and nitrogen loss in a flooded soil. Soil. Biol. Biochem. 7:87-94.

Reddy, K. R., and W. H. Patrick, Jr. 1976. Effect of frequent changes in aerobic and anaerobic conditions in redox potential and nitrogen loss in a flooded soil. Soil Biol. Biochem. 8:491-495.

Reddy, K. R., P. D. Sacco, P. A. Graetz, K. L. Campbell, and L. R. Sinclair. 1982. Water treatment by aquatic ecosystems: Nutrient removal by reservoirs and flooded fields. Environ. Manage. 6:261-271.

Redfield, A. C. 1972. Development of a New England salt marsh. Ecol. Monogr. 42:201-237.

Reed, A. 1975. Reproductive output of black ducks on the St. Lawrence estuary. J. Wildl. Manage. 39:2243-2255.

Reed, A., and G. Moisan. 1971. The *Spartina* tidal marshes of the St. Lawrence estuary and their importance to aquatic birds. Nat. Can. 98:905-922.

Reed, J. P., J. M. Miller, D. F. Pence, and B. Schaich. 1983. The effects of low level turbidity on fish and their habitat. Report 190. Water Resour. Res. Inst., Univ. of N. Carolina, Raleigh.

Reice, S. R., and A. E. Stiven. 1983. Environmental patchiness, litter decomposition and associated faunal patterns in a *Spartina alterniflora* marsh. Est. Coast. Shelf Sci. 16:559-571.

Reichholf, V. J. 1976. The possible use of the aquatic bird communities as indicators for the ecological conditions of wetlands. Landschaft + Stadt 3:125-129.

Reimold, R. J. 1972. The movement of phosphorus through the salt marsh cordgrass, *Spartina alterniflora.* Limnol. Oceanogr. 17:606-611.

Reinecke, K. J., and R. B. Owen, Jr. 1980. Food use and nutrition of black ducks nesting in Maine. J. Wildl. Manage. 44:549-558.

Remane, A. 1971. Ecology of brackish water. Pages 1-210 in A. Remane and C. Schlieper (eds.), Biology of Brackish Water. John Wiley and Sons, New York.

Renouf, R. N. 1972. Waterfowl utilization of beaver ponds in New Brunswick. J. Wild. Manage. 36:740-744.

Renwick, W. H., and G. M. Ashley. 1984. Sources, storages, and sinks of fine-grained sediments in a fluvial-estuarine system. Geol. Soc. Am. Bull. 95:1343-1348.

Rey, J. R., and R. A. Crossman. 1987. Physical and biological studies of impounded marshes along the Indian River lagoon of east-central Florida. Pages 396-402 in W. R. Whitman and W. H. Meredith (eds.), Proceedings of a Symposium on Waterfowl and Wetlands Management in the Coastal Zone of the Atlantic Flyway. Delaware Dept. of Natural Resources and Environmental Control, Dover.

Reynoldson, T. B. 1987. Interactions between sediment contaminants and benthic organisms. Hydrobiol. 149:53-66.

Ribelin, B. W., and A. W. Collier. 1979. Ecological considerations of detrital aggregates in the salt marsh. Pages 47-69 in R. J. Livingston (ed.), Ecological processes in coastal and marine systems. Plenum Press, New York.

Ricci, E. D., W. A. Hubert, and J. J. Richard. 1983. Organochlorine residues in sediment cores of a midwestern reservoir. J. Environ. Qual. 12(3):418-421.

Rice, D. C., and K. R. Tenore. 1981. Dynamics of carbon and nitrogen during the decomposition of detritus derived from estuarine macrophytes. Estuarine Coastal Shelf Sci. 13:681-690.

Rice, D. L. 1982. The detritus nitrogen problem: New observations and perspectives from organic geochemistry. Mar. Ecol. Prog. Ser. 9:153-162.

Richards, G. A. 1978. Seasonal and environmental variations in sediment accretion in a Long Island salt marsh. Estuaries 1(1): 29-35.

Richards, J. F. 1934. The salt marshes of the Dovey Estuary; IV, The rates of vertical accretion, horizontal extensions and scarp erosion. Ann. Bot. 48:225-259.

Richardson, C. J., D. L. Tilton, J. A. Kadlec, J. P. M. Chamie, and W. A. Wentz. 1978. Nutrient dynamics of northern wetland ecosystems. Pages 217-242 in R. E. Good, D. F. Whigham, and R. L. Simpson (eds.), Freshwater Wetlands. Academic Press, New York.

Richardson, C. J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science 228:1424-1427.

Richardson, C. J., and D. S. Nichols. 1985. Ecological analysis of wastewater management criteria in wetland ecosystems. Pages 351-391 in P. J. Godfrey, E. R. Kaynor, S. Pelczarski, and J. Benforado (eds.), Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. Van Nostrand Reinhold, New York.

Richardson, D. J., and P. E. Marshall. 1986. Processes controlling movement, storage, and export of phosphorus in a fen peatland. Ecol. Monogr. 56:279-302.

Richardson, E. M., and E. Epstein. 1971. Retention of three insecticides on different size soil particles suspended in water. Soil Sci. Soc. Am. Proc. 35:884-887.

Richardson, J. L., and R. J. Bigler. 1984. Principal component analysis of prairie pothole soils in North Dakota. Soil Sci. Soc. Am. J. 48:1350-1355.

Rimmer, D. M. 1985. Effects of reduced discharge on production and distribution of age-0 rainbow trout in seminatural channels. Tran. Am. Fish. Soc. 114:388-396.

Ringelman, J. K., and J. R. Longcore. 1982. Movements and wetland selection by brood-rearing black ducks. J. Wildl. Manage. 46:615-621.

Ritter, D. F. 1986. Process Geomorphology. Wm. C. Brown Publ., Dubuque, IA. 579 p.

Robbins, C. S. 1979. Effects of forest fragmentation on bird populations. Pages 198-212 in R. M. DeGraaf and N. Tilghman (eds.), Proc. of the Workshop on Management of North-central and Northeastern Forests for Nongame Birds. Gen. Tech. Rep. NC-51. US Forest Service, Washington, DC.

Robel, R. J. 1961. The effects of carp populations on the production of waterfowl plants on western waterfowl marsh. Trans. N. Am. Wildl. Nat. Resour. Conf. 26:147-159.

Robertson, R. J., and N. J. Flood. 1980. Effects of recreational use of shorelines on breeding bird populations. Can. Field Nat. 94(2):131-138.

Rochford, D. J. 1953. Studies in Australian hydrology; I. Introductory and comparative features. Aust. J. Mar. Freshw. Res. 3:1-116.

Rogers, S. G., T. E. Targett, and S. B. Van Sant. 1984. Fish-nursery use in Georgia salt-marsh estuaries: The influence of springtime freshwater conditions. Trans. Am. Fish. Soc. 113:595-606.

Ross, S. T., and J. A. Baker. 1983. The response of fishes to periodic spring floods in a southeastern stream. Am. Midl. Nat. 109:1-14.

Roth, R. R. 1976. Spatial heterogeneity and bird species diversity. Ecology 57:773-782.

Rozas, L. P., and C. T. Hackney. 1983. The importance of oligohaline estuarine wetland habitats to fisheries resources. Wetlands 3:77-89.

Rozas, L. P., and C. T. Hackney. 1984. Use of oligohaline marshes by fishes and macrofaunal crustaceans in North Carolina. Estuaries 7:213-224.

Rozas, L. P., and W. E. Odum. 1987. Fish and macrocrustacean use of submerged plant beds in tidal freshwater marsh creeks. Mar. Ecol. Prog. Ser. 38:101-108.

Rozas, L. P., and W. E. Odum. 1987. Use of tidal freshwater marshes by fishes and macrofaunal crustaceans along a marsh stream-order gradient. Estuaries 10:36-43.

Rozas, L. P., and W. E. Odum. 1987. The role of submerged aquatic vegetation in influencing the abundance of nekton on contiguous tidal freshwater marshes. J. Exp. Mar. Biol. Ecol. 114:289-300.

Rubey, W. W. 1938. The force required to move particles on a stream bed. Prof. Paper 189-E. US Geological Survey, Reston, VA. 20 pp.

Rumble, M. A., and L. D. Flake. 1983. Management considerations to enhance use of stock ponds by waterfowl broods. J. Range Manage. 36:691-694.

Rundquist, D. C., M. P. Lawson, L. P. Queen, and R. S. Cerveny. 1987. The relationship between summer-season rainfall events and lake-surface area. Water Resour. Bull. 23(3):493-508.

Rusnak, G. A. 1967. Rates of sediment accumulation in modern estuaries. Pages

180-184 in G. H. Lauff (ed.), Estuaries. Am. Assoc. Adv. Sci., Washington, DC.

Ruwaldt, J. J., Jr., L. D. Flake, and J. M. Gates. 1979. Waterfowl pair use of natural and manmade wetlands in South Dakota. J. Wild. Manage. 43:375-383.

Ryder, R. A. 1965. A method for estimating the potential fish production of north-temperate lakes. Trans. Am. Fish. Soc. 94:214-218.

Ryder, R. A., W. D. Graul, and G. C. Miller. 1980. Status, distribution, and movement of Ciconiiforms in Colorado. Proc., 1979 Conf. Colonial Waterbird Group 3:49-57.

Sakamoto, M. 1971. Chemical factors involved in the control of phytoplankton production in the Experimental Lakes Area, northwestern Ontario. J. Fish. Res. Bd. Can. 28:203-213.

Sampou, P. 1985. The effect of sewage sludge loading on carbon and energy cycling in an estuarine sediment (Abstr.). Estuaries 8:26A.

Sanville, W. 1988. Response of an Alaskan wetland to nutrient enrichment. Aquat. Bot. 30:231-243.

Sather, J. H., and P. R. Stuber. 1984. Proceedings of the National Wetlands Values Assessment Workshop. FWS/OBS-84/12. US Fish and Wildlife Service, Washington, DC.

Savino, J. F., and R. A. Stein. 1982. Predator-prey interaction between largemouth bass and bluegills as influenced by simulated, submersed vegetation. Trans. Am. Fish. Soc. 111:255-266.

Scaife, W. W., R. E. Turner, and R. Costanza. 1983. Coastal Louisiana recent land loss and canal impacts. Envir. Manage. 7(5):433-442.

Scarnecchia, D. L. 1981. Effects of streamflow and upwelling on yield of wild coho salmon. Can. J. Fish. Aquat. Sci. 38:471-475.

Scarnecchia, D. L., and E. P. Bergersen. 1987. Trout production and standing crop in Colorado's small streams, as related to environmental features. N. Am. J. Fish. Manag. 7:315-330.

Scheffer, M., A. A. Achterberg, and B. Beltman. 1984. Distribution of macroinvertebrates in a ditch in relation to the vegetation. Freshw. Biol. 14:367-370.

Schindler, D. W. 1978. Factors regulating phytoplankton production and standing crop in the world's fresh waters. Limnol. Oceanogr. 23:478-486.

Schindler, D.W. 1987. Detecting ecosystem responses to anthropogenic stress. Can. J. Fish. Aquat. Sci. 44(Suppl. 1):6-25.

Schlosser I. J., and J. R. Karr. 1981. Riparian vegetation and channel morphology impact on spatial patterns of water quality in agricultural watersheds. Envir. Manage. 5:233-243.

Schlosser, I. J. 1987. The role of predation in age and size related habitat use by stream fishes. Ecology 68:651-659.

Schlosser, I. J., and J. R. Karr. 1981. Water quality in agricultural watersheds: Impact of riparian vegetation during base flow. Water Res. Bull. 17:233-240.

Schmal, R. N., and D. F. Sanders. 1978. Effects of stream channelization on aquatic macroinvertebrates, Buena Vista Marsh, Portage County, Wisconsin. FWS/OBS-78/92. US Fish and Wildlife Service, Office of Biological Services. 80 p.

Schnabel, R.R. 1986. Nitrate concentrations in a small stream as affected by chemical and hydrologic interactions in the riparian zone. Pages 263-282 in D. L. Correll (ed.), Watershed Research Perspectives. Smithsonian Institution Press, Washington, DC.

Schneider, D. C., and B. A. Harrington. 1981. Timing of shorebird migration in relation to prey depletion. Auk 98:901-811.

Schoenberg, S. A., and J. D. Oliver. 1988. Temporal dynamics and spatial variation of algae in relation to hydrology and sediment characteristics in the Okefenokee Swamp, Georgia. Hydrobiol. 162:123-133.

Schramm, H. L., Jr., K. J. Jirka, and M. V. Hoyer. 1987. Epiphytic macroinvertebrates on dominant macrophytes in two central Florida lakes. J. Freshw. Ecol. 4:151-176.

Schreiber, R. W. 1977. Maintenance behaviour and communication in the brown pelican. Am. Ornithologists Union Monogr. 22. 78 p.

Schreiber, R. W. 1979. Reproductive performance of the eastern brown pelican. Contrib. Sci. Nat. Hist. Mus. Los Angeles 317:1-43.

Schubel, J. R., and H. S. Carter. 1984. The estuary as a filter for fine-grained suspended sediment. Pages 81-105 in V. S. Kennedy (ed.), The Estuary as a Filter. Academic Press, Orlando, FL.

Schwan, M. W. 1985. A study of land use activities and their relationship to the sport fish resources in Alaska; Vol 26, Study D-I, Job D-I-B. Alaska Dept. Fish & Game, Juneau.

Schwartz, F. W., and W. A. Milne-Home. 1982. Watersheds in muskeg terrain. J. Hydrol. 57:267-305.

Schwartz, L. N., and G. K. Gruendling. 1985. The effects of sewage on a Lake Champlain wetland. J. Freshw. Ecol. 3:35-46.

Scrivener, J. C., and B. C. Andersen. 1984. Logging impacts and some mechanisms that determine the size of spring and summer populations of Coho salmon fry (*Oncorhynchus kisutch*) in Carnation Creek, British Columbia. Can. J. Fish and Aquat. Sci. 41:1097-1105.

Sculthorpe, C. D. 1967. The Biology of Aquatic Vascular Plants. Edward Arnold, London. 610 p.

Sedell, J. R., F. J. Triska, and N. S. Triska. 1975. The processing of conifer and hardwood leaves in two coniferous forest streams; I. Weight loss and associated invertebrates. Verh. Internat. Verein. Limnol. 19:1617-1627.

Seidel, K. 1976. Macrophytes and water purification. Pages 109-121 in J. Tourbier and R. W. Pierson, Jr. (eds.). Biological Control of Water Pollution. Univ. Pa. Press, Philadelphia.

Seitzinger, S. P., S. W. Nixon, and M. E. Q. Pilson. 1984. Denitrification and nitrous oxide production in a coastal marine ecosystem. Limnol. Oceanogr. 29:73-83.

Seitzinger, S. P. 1988. Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. Limnol. Oceanogr. 33(4):702-725.

Seliskar, D. M., and J. L. Gallagher. 1983. The ecology of tidal marshes of the Pacific northwest coast: A community profile. FWS/OBS-82/32. US Fish and Wildlife Service, Washington, DC. 65 p.

Serodes, J. B., and J. P. Troude. 1984. Sedimentation cycle of a fresh water tidal flat in the St. Lawrence estuary. Estuaries 7:119-127.

Settlemyre, J. L., and L. R. Gardner. 1977. Suspended sediment flux through a salt marsh drainage basin. Est. Coast. Mar. Sci. 5:653-663.

Sewell, P. L. 1982. Urban groundwater as a possible nutrient source for an estuarine benthic algal bloom. Est. Coast. Shelf Sci. 15:569-576.

Shaffer, G. P. 1988. K-systems analysis for determining the factors influencing benthic microfloral productivity in a Louisiana estuary, USA. Mar. Ecol. Prog. Ser. 43:43-54.

Shahane, A. 1982. Estimation of pre- and post-development non-point water quality loadings. Water Resources Bull. 18(2):231-233.

Sharma, P., L. R. Gardner, W. S. Moore, and M. S. Bollinger. 1987. Sedimentation and bioturbation in a salt marsh as revealed by ^{210}Pb, ^{137}Cs, and ^{7}Be studies. Limnol. Oceanogr. 32(2):313-326.

Sharom, M. S., J. R. W. Miles, C. R. Harris, and F. L. McEwen. 1980. Behavior of 12 insecticides in soil and aqueous suspensions of soil and sediment. Water Res. 14(8):1095-1100.

Sheath, R. G., J. M. Burkholder, J. A. Hambrook, A. M. Hogeland, E. Hoy, M. E. Kane, M. O. Morison, A. D. Steinman, and K. L. VanAlstyne. 1986. Characteristics of softwater streams in Rhode Island; III. Distribution of macrophytic vegetation in a small drainage basin. Hydrobiol. 140:183-191.

Shepard, F. P., and D. G. Moore. 1960. Bays of central Texas coast. Pages 117-152 in F. P. Shepard, F. B. Phleger, and T. H. Van Andel (eds.), Recent sediments, northwest Gulf of Mexico. Am. Assoc. Petrol. Geol., Tulsa, OK.

Sheridan, J. M., and R. K. Hubbard. 1987. Transport of solids in streamflow from coastal plain watersheds. J. Environ. Qual. 16(2):131-136.

Sheridan, P. F., and R. J. Livingston. 1979. Cyclic trophic relationships of fishes in an unpolluted, river-dominated estuary in north Florida. Pages 143-161 in R. J. Livingston (ed.), Ecological Processes in Coastal and Marine Systems. Plenum Press, New York.

Shirazi, M. A., and W. K. Seim. 1979. A stream systems evaluation—an emphasis on spawning habitat for salmonids. EPA-600/3-79-109. US Environmental Protection Agency, Washington, DC. 44 p.

Short, F. R., and C. A. Short. 1984. The seagrass filter: Purification of estuarine and coastal waters. In V. S. Kennedy (ed.), The Estuary as a Filter. Academic Press, Inc.

Sidle, R. C. 1986. Seasonal patterns of allochthonous debris in three riparian zones of a coastal Alaska drainage. Pages 283-304 in D. L. Correll (ed.), Watershed Research Perspectives. Smithsonian Institution Press, Washington, DC.

Siegel, D. I. 1981. Hydrogeologic setting of the glacial Lake Agassiz peatlands, North Minnesota. US Geol. Surv. Water Resour. Invest. 81-24. 10 p.

Siegel, D. I. 1988a. The recharge-discharge function of wetlands near Juneau, Alaska; Part I, Hydrogeological investigations. Ground Water 26(4):427-434.

Siegel, D. I. 1988b. The recharge discharge function of wetlands near Juneau, Alaska; Part II, Geochemical investigations. Ground Water 26(5):580-586.

Sigafoos, R. S. 1964. Botanical evidence of floods and floodplain deposition, vegetation and hydrologic phenomena. Prof. Paper 485-A, US Geol. Surv. Reston, VA.

Sigler, J. W., T. C. Bjornn, and F. H. Everest. 1984. Effects of chronic turbidity on density and growth of steelheads and coho salmon. Tran. Am. Fish. Soc. 113:142-150.

Silberhorn, G. M., G. M. Dawes, and T. A. Barnards, Jr. 1974. Coastal wetlands of Virginia: Guidelines for activities affecting Virginia wetlands. Va. Inst. Mar. Sci. Inter. Rep. No. 3., Gloucester Point. 52 p.

Simberloff, D., and L. G. Abele. 1982. Refuge design and island biogeographic theory: Effects of fragmentation. Am. Nat. 120:41-50.

Simenstad, C. A. 1983. The ecology of estuarine channels of the Pacific Northwest coast: A community profile. FWS/OBS-83/05. US Fish and Wildlife Service. 181 p.

Simmons, C. E. 1976. Sediment characteristics of streams in the Eastern Piedmont and Western Coastal Plain regions of North Carolina. Water Supply Paper 1798-0, US Geol. Surv., Reston, VA. 32 p.

Simpson, K., and J. P. Kelsall. 1979. Capture and banding of adult great blue herons at Pender Harbour, British Columbia. Proc. 1978 Conf. Colonial Waterbird Group 2:71-78.

Simpson, P. W., J. R. Newman, M. A. Keirn, R. M. Matter, and P. A. Guthrie. 1982. Manual of Stream Channelization Impacts on Fish and Wildlife. FWS/OBS-82/24. Office of Biological Services, US Fish and Wildlife Service. 155 p.

Simpson, R. L., D. F. Whigham, and R. Walker. 1978. Seasonal patterns of nutrient movement in a freshwater tidal marsh. Pages 243-257 in R. E. Good, D. Whigham, and R. Simpson (eds.), Freshwater Wetlands. Academic Press, New York.

Simpson, R. L., R. E. Good, R. Walker, and B. R. Frasco. 1983. The role of Delaware River freshwater tidal wetlands in the retention of nutrients and heavy metals. J. Environ. Qual. 12:41-48.

Sklar, F. H. 1985. Seasonality and community structure of the backswamp invertebrates in a Louisiana cypress-tupelo wetland. Wetlands 5:69-86.

Sklar, F. H., and W. H. Conner. 1979. Effects of altered hydrology on primary production and aquatic animal populations in a Louisiana swamp forest. Pages 191-208 in J. W. Day, Jr., D. D. Culley, Jr., R. E. Turner, and A. J. Mumphrey, Jr. (eds.), Proc. Third Coastal Marsh and Estuary Management Symposium. Louisiana State Univ. Division of Continuing Education, Baton Rouge, LA.

Sklash, M., and R. Farvolden. 1979. The role of groundwater in storm runoff. J. Hydrol. 43:45-65.

Sloan, C. E. 1972. Ground-water hydrology of prairie potholes. US Geol. Surv. Prof. Paper 585-C. 28 p.

Sloan, C. E. 1979. Prairie potholes and the water table. US Geol. Surv. Prof. Paper 700-B:227-231.

Sloane-Richey, J., M. A. Perkins, and K. W. Malueg. 1981. The effects of urbanization and stormwater runoff on the food quality in two salmonid streams. Verh. Internat. Verein. Limnol. 21:812-818.

Sloey, W. E., F. L. Spangler, and C. W. Fetter, Jr. 1978. Management of freshwater wetlands for nutrient assimilation. Pages 321-340 in R. E. Good, D. F. Whigham, and R. L. Simpson (eds.), Freshwater Wetlands. Academic Press, New York.

Small, E. 1972. Ecological significance of four critical elements in plants of raised sphagnum peat bogs. Ecology 53:498-503.

Smith, B. D., P. S. Maitland, M. R. Young, and M. J. Carr. 1981. The ecology of Scotland's largest lochs: Lomond, Awe, Ness, Morar and Shiel; 7. The littoral zoobenthos. Monogr. Biol. 44:155-204.

Smith, B. D., P. S. Maitland, and S. M. Pennock. 1987. A comparative study of water level regimes and littoral benthic communities in Scottish lochs. Biol. Conserv. 39:291-316.

Smith, C. J., and R. D. DeLaune. 1983. Nitrogen loss from freshwater and saline estuarine sediments. J. Environ. Qual. 12:514-518.

Smith, C. J., R. D. DeLaune, and W. H. Patrick, Jr. 1985. Fate of riverine nitrate entering an estuary; I. Dentrification and nitrogen burial. Estuaries 8:15-21.

Smith, D. W. 1980. An evaluation of marsh nitrogen fixation. Pages 135-142 in V. S. Kennedy (ed.), Estuarine Perspectives. Academic Press, New York.

Smith, D. W., and A. J. Horne. 1988. Experimental measurement of resource competition between planktonic microalgae and macroalgae (seaweeds) in mesocosms simulating the San Francisco Bay-Estuary, California. Hydrobiol. 159:259-268.

Smith, J. A., P. J. Witkowski, and T. V. Fusillo. 1988. Manmade organic compounds in the surface waters of the United States—A review of current understanding. US Geol. Surv. Circular 1007. 92 p.

Smith, S. M., J. G. Hoff, S. P. O'Neil, and M. P. Weinstein. 1984. Community and trophic organization of nekton utilizing shallow marsh habitats, York River, Virginia. Fish. Bull. 82:455-467.

Smith, T. J., III, and W. E. Odum. 1981. The effects of grazing by snow geese on coastal marshes. Ecology 61:98-106.

Smith, V. H., and M. Wallsten. 1986. Prediction of emergent and floating-leave macrophyte cover in central Swedish lakes. Can. J. Fish. Aquat. Sci. 43:2519-2523.

Smock, L. A., and K. L. Harlowe. 1983. Utilization and processing of freshwater wetland macrophytes by the detritivore *Asellus forbesi*. Ecology 64:1556-1565.

Smock, L. A., E. Gilinsky, and D. L. Stoneburner. 1985. Macroinvertebrate production in a southeastern United States blackwater stream. Ecology 66(5):1491-1503.

Sollins, P., Glassman, C. A., and C. N. Dahm. 1985. Composition and possible origin of detrital material in streams. Ecol. 66(1):297-299.

Soots, R. F., Jr., and M. C. Landin. 1978. Development and management of avian habitat on dredged material islands. Tech. Rep. DS-78-18. US Army Engineer Waterways Experiment Station, Vicksburg, MS. 134 p.

Sophocleous, M., and J. A. McAllister. 1987. Basinwide water-balance modeling with emphasis on spatial distribution of ground water recharge. Water Resour. Bull. 23(6):997-1010.

Spangler, F. L., W. E. Sloey, and C. W. Fetter, Jr. 1976. Wastewater treatment by natural and artificial marshes. EPA-600/2-76-207. US Environmental Protection Agency, Washington, DC. 171 p.

Spear, P. R., and C. R. Gamble. 1964. Magnitude and frequency of floods in the United States; Part 2-A, South Atlantic slope basins, James River to Savannah River. Water-Supply Paper 1673. US Geological Survey, Reston, VA. 329 p.

Spencer, W. F., M. M. Cliath, W. J. Farmer, and R. A. Shepard. 1974. Volatility of DDT residues in soil as affected by flooding and organic matter applications. J. Environ. Qual. 3:126-129.

Stabel, H., and M. Geiger. 1985. Phosphorus adsorption to river suspended matter: Implications for the phosphorus budget of Lake Constance. Water Resources 19(11):1347-1352.

Stall, J. B. 1972. Effects of sediment on water quality. J. Environ. Qual. 1:353-360.

Stall, J. B., and C. T. Yang. 1972. Hydraulic geometry and low stream flow regime. Water Resour. Cen. Res. Rep. 54, Urbana, IL.

Stalmaster, M. V., and J. R. Newman. 1978. Behavioral responses of wintering bald eagles to human activity. J. Wildl. Manage. 42:506-513.

Starret, W. C. 1951. Some factors affecting the abundance of minnows in the Des Moines River, Iowa. Ecology 32:13-27.

Stauffer, D. F., and L. B. Best. 1980. Habitat selection by birds of riparian communities: Evaluating effects of habitat alterations. J. Wildl. Manage. 44:1-15.

Steedman, R. J. 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in Southern Ontario. Can. J. Fish. Aquat. Sci. 45:492-501.

Stengel, E., W. Carduck, and C. Jebsen. 1987. Evidence for denitrification in artificial wetlands. Pages 543-550 in K. R. Reddy and W. H. Smith (eds.), Aquatic Plants for Water Treatment and Resource Recovery. Magnolia Publishing, Inc.

Stephenson, M., G. Turner, P. Pope, and A. Knight. 1981. Wetlands for wastewater treatment: Just another threat—or an opportunity? Wetlands 1:204-207.

Stern, D. H., and M. S. Stern. 1980. Effects of bank stabilization on the physical and chemical characteristics of streams and small rivers: A synthesis. FWS/OBS-80/11. US Fish Wildl. Serv. 43 p.

Stern, E. M., and W. B. Stickle. 1978. Effects of turbidity and suspended material in aquatic environments; Literature review. Tech. Rep. D-78-21. US Army Engineer Waterways Experiment Station, Vicksburg, MS. 116 p.

Stevens, L. E., B. T. Brown, J. M. Simpson, and R. R. Johnson. 1977. The importance of riparian habitat to migrating birds. Pages 156-164 in R. R. Johnson and D. A. Jones (tech. coord.), Importance, Preservation and Management of Riparian Habitat: A Symposium. USDA For. Serv. Gen. Tech. Rep. RM-42, Washington, DC.

Stevenson, J. C. 1988. Comparative ecology of submersed grass beds in freshwater, estuarine, and marine environments. Limnol. Oceanogr. 33(4):867-893.

Stevenson, J. C., D. Heinle, D. Flemer, R. Small, R. Rowland, and J. Ustach. 1977. Nutrient exchanges between brackish water marshes and the estuary. Pages 219-240 in M. Wiley (ed.), Estuarine Processes; Vol 2, Circulation, Sediments, Transfer of Material in the Estuary. Academic Press. New York.

Stevenson, J. C., L. G. Ward, and M. S. Kearney. 1988. Sediment transport and trapping in marsh systems: Implications of tidal flux studies. Mar. Geol. 80:37-59.

Steward, K. K., and W. H. Ornes. 1975. Assessing a marsh environment for wastewater renovation. J. Water Pollut. Control Fed. 47:1880-1891.

Stewart, E. M. 1983. Distribution and abundance of fishes in natural and modified bottomland wetlands. M.S. thesis, Univ. of Missouri, Columbia. 90 p.

Stewart, R. E., and H. A. Kantrud. 1972. Vegetation of prairie potholes in North Dakota in relation to quality of water and other environmental factors. US Geol. Surv. Prof. Pap. 585-D. 36 p.

Stewart, R. E., and H. A. Kantrud. 1973. Ecological distribution of breeding waterfowl populations in North Dakota. J. Wildl. Manage. 37:39-50.

Stewart, R. E., and H. A. Kantrud. 1974. Breeding waterfowl populations in the prairie pothole region of North Dakota. Condor 76:70-79.

Stoner, A. W. 1983. Distribution of fishes in seagrass meadows: Role of macrophyte biomass and species composition. Fish. Bull. 81:837-846.

Stow, C. A., R. D. DeLaune, and W. H. Patrick, Jr. 1985. Nutrient fluxes in a eutrophic coastal Louisiana freshwater lake. Environ. Manage. 9:243-252.

Strange, R. J., C. R. Berry, and C. B. Schreck. 1975. Aquatic plant control and reservoir fisheries. Pages 513-521 in R. H. Stroud and M. Clepper (eds.), Black bass biology and management. Sport Fish. Inst., Washington, DC.

Strawn, K. 1961. Factors influencing the zonation of submerged monocotyledons at Cedar Kay, Florida. J. Wildl. Manage. 25:178-189.

Striegl, R. G. 1987. Suspended sediment and metals removal from urban runoff by a small lake. Water Resour. Bull. 23(6):985-996.

Stuart, V., E. J. H. Head, and K. H. Mann. 1985. Seasonal changes in the digestive enzyme levels of the amphipod *Corophium volutator* (Pallas) in relation to diet. J. Exp. Mar. Biol. Ecol. 88:243-256.

Stumpf, R. P. 1983. The process of sedimentation on the surface of a salt marsh. Estuarine Coastal Shelf Science 17:495-508.

Sugden, L. G. 1978. Canvasback habitat use and production in Saskachewann parklands. Canada Wildl. Serv. Occas. Pap. No. 34. Canadian Wildlife Service, Ottawa. 32 p.

Sutcliffe, W. H., Jr. 1972. Some relations of land drainage, nutrients, particulate material, and fish catch in two eastern Canadian bays. J. Fish. Res. Bd. Can. 29:357-362.

Swanson, G. A. 1977. Diel food selection by Anatiinae on a waste stabilization system. J. Wildl. Manage 41: 226-231.

Swanson, G. A., G. L. Krapu, and J. R. Serie. 1979. Foods of laying female dabbling ducks on the breeding grounds. Pages 47-57 in T. A. Bookout (ed.), Waterfowl and wetlands—An integrated review. N. Central Sec. Wildl. Soc., Madison, WI.

Swanson, G. A., V. A. Adomeitis, F. B. Lee, J. R. Serie, and J. A. Shoesmith. 1983. Limnological conditions influencing duckling use of saline lakes in south-central North Dakota. J. Wildl. Manage. 48:340-349.

Swanson, G. A. 1988. Aquatic habitats of breeding waterfowl. Pages 195-202 in D. D. Hook, W. H. McKee, Jr., H. K. Smith, J. Gregory, V. G. Burrell, Jr., M. R. DeVoe, R. E. Sojka, S. Gilbert, R. Banks, L. H. Stolzy, D. Brooks, T. D. Matthews, and T. H. Shear (eds.). The Ecology and Management of Wetlands; Vol 1: Ecology of Wetlands. Croom Helm, London and Sydney.

Swanson, G. A., and M. I. Meyer. 1977. Impact of fluctuating water levels on feeding ecology of breeding blue-winged teal. J. Wildl. Manage. 41:426-433.

Swift, B. L., J. S. Larson, and R. M. DeGraff. 1984. Relationship of breeding bird density and diversity to habitat variables on forested wetlands. Wilson Bull. 96:48-59.

Syers, J. K., R. F. Harris, and D. E. Armstrong. 1973. Phosphate chemistry in lake sediments. J. Envir. Qual. 2:1-4.

Szijj, J. 1972. Some suggested criteria for determining the international importance of wetlands in the western Palearctic. Pages 111-119 in E. Carp (ed.), Proc. Int. Conf. on Conservation of Wetlands and Waterfowl, Ramsar, Iran. International Wildfowl Res. Bur., Slimbridge, Glasgow, England.

Tarplee, W. H., Jr., D. E. Louder, and A. J. Weber. 1971. Evaluation of the effects of channelization of fish populations in North Carolina's coastal plain streams. *North Carolina Wildlife Resources Commission*, Raleigh, NC.

Tassone, J. 1981. Utility of hardwood leave strips for breeding birds in Virginia's central Piedmont. M.S. thesis, Va. Polytech. Inst., Blacksburg. 83 p.

Tate, C. M., and J. L. Meyer. 1983. The influence of hydrologic conditions and successional state on dissolved organic carbon export from forested watersheds. Ecology 64:25-32.

Taylor, T. J., and J. S. Barclay. 1985. Renovation of a Plains State stream—physical problem solving. Pages 62-66 in R. R. Johnson et al. (eds.), Riparian Ecosystems and Their Management. Gen. Tech. Rep. RM-120. USDA Forest Service, Fort Collins, CO.

Taylor, F., and A. C. Hendricks. 1987. The influence of fish on leaf breakdown in a Virginia pond. Freshw. Biol. 18:45-51.

Taylor, W. D., V. W. Lambow, L. R. Williams, and S. C. Hern. 1980. Trophic state of lakes and reservoirs. Tech. Rep. E-80-3. US Army Engineer Waterways Experiment Station, Vicksburg, MS. 26 p.

Teal, J. M. 1962. Energy flow in the salt marsh ecosystem of Georgia. Ecology 43:614-624.

Teal, J. M., I. Valiela, and D. Berlo. 1979. Nitrogen fixation by rhizosphere for free-living bacteria in salt marsh sediments. Limnol. Oceanogr. 24:126-132.

Teal, J. M. 1986. The ecology of regularly flooded marshes of New England: A community profile. Biological Report 85(7.4). US Fish and Wildlife Service, Washington, DC.

Teal, J. M., A. Giblin, and I. Valiela. 1982. The fate of pollutants in American salt marshes. In B. Gopal, R. E. Turner, R. G. Wetzel, and D. F. Whigham (eds.), Wetlands Ecology and Management. National Institute of Ecology and International Scientific Publications, Jaipur, India. 514 p.

Teels, B. M., G. Anding, D. H. Arner, E. D. Norwood, and N. E. Wesley. 1978. Aquatic plant-invertebrate and waterfowl associations in Mississippi. Proc. Southeast. Assoc. Fish Wildl. Agenc. 30:610-616.

Tenore, K. R. 1977. Utilization of aged detritus derived from different sources by the polychaete *Capitella capitata*. Mar. Biol. 44:51-55.

Thayer, G. W. 1971. Phytoplankton production and the distribution of nutrients in a shallow unstratified estuarine system near Beaufort, North Carolina. Chesapeake Sci. 12:240-253.

Thayer, G. W., D. A. Wolfe, and R. B. Williams. 1975. The impact of man on seagrass systems. Am. Scientist 63:288-296.

Thayer, G. W., D. W. Engel, and M. W. LaCroix. 1977. Seasonal distributions and changes in the nutritive quality of living, dead and detrital fractions of *Zostera marina*. J. Exp. Mar. Biol. Ecol. 30:109-127.

Thayer, G. W., W. J. Kenworthy, and M. S. Fonseca. 1984. The ecology of eelgrass meadows of the Atlantic Coast: A community profile. FWS/OBS-84/02. US Fish Wildl. Serv., Washington, DC. 147 p.

Thayer, G.W., D. R. Colby, and W. F. Hettler, Jr. 1987. Utilization of the red mangrove prop root habitat by fishes in south Florida. Mar. Ecol. Prog Ser. 35:25-38.

Thayer, G. W., H. H. Stuart, W. J. Kenworthy, J. F. Ustach, and A. B. Hall. 1979. Habitat values of salt marshes, mangroves, and seagrasses for aquatic organisms. Pages 235-247 in P. Greeson et al. (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Thayer, G. W., P. L. Parker, M. W. LaCroix, and B. Fry. 1978. The stable carbon isotope ratio of some components of an eel-grass, *Zostera marina*. Oecologia 35:1-12.

Theurer, F. D., I. Lines, and T. Nelson. 1985. Interaction between riparian vegetation, water temperature, and salmonid habitat in the Tucannon River. Water Resour. Bull. 21(1):53-64.

Thibodeau, F. R., and N. H. Nickerson. 1985. Changes in a wetland plant association induced by impoundment and draining. Biol. Conserv. 33:269-279.

Thomas, D. M., and M. A. Benson. 1970. Generalization of streamflow characteristics from drainage basin characteristics. Water-Supply Paper 1975. US Geological Survey, Washington, DC.

Thomas, J. W., C. Maser, and J. E. Rodiek. 1979. Wildlife habitats in managed rangelands—the Great Basin of southeastern Oregon: Edges. US For. Serv. Gen. Tech. Rep. PNW-85. 17 p.

Thompson, D. 1973. Feeding ecology of diving ducks on Keokuk Pool, Mississippi River. J. Wildl. Manage. 37:367-381.

Thorne-Miller, B., M. M. Harlin, G. B. Thursby, M. M. Brady-Campbell, and B. A. Dworetzky. 1983. Variations in the distribution and biomass of submerged macrophytes in five coastal lagoons in Rhode Island, USA. Bot. Mar. 26:231-242.

Thorp, J. H., E. M. McEwan, M. F. Flynn, and F. R. Hauer. 1985. Invertebrate colonization of submerged wood in a cypress-tupelo swamp and blackwater stream. Am. Midl. Nat. 113:56-58.

Thyssen, N., M. Erlandsen, E. Jeppesen, and T. F. Holm. 1983. Modelling the reaeration capacity of low-land streams dominated by submerged macrophytes. Pages 861-867 in W. K. Lauenroth, G. V. Skogerboe, and M. Flug (eds.), Analysis of ecological systems: State-of-the-art in ecological modelling. Elsevier Scientific Publ., New York.

Tilley, L. J., and W. A. Dawson. Plant nutrients and the estuary mechanism in the Duwamish River Estuary, Seattle, Washington. Pages C185-C191 in Prof. Paper No. 750-C. US Geological Survey, Reston, VA.

Tilton, D. L., and B. R. Schwegler. 1978. The value of wetland habitat in the Great Lakes Basin. Pages 267-277 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Tilton, D. L., and R. A. Kadlec. 1979. The utilization of a freshwater wetland for nutrient removal from secondary treated waste water effluent. J. Environ. Qual. 8:328-334.

Tiner, R.C. 1981. Interrelationships between biological, chemical, and physical variables in Mount Hope Bay, Massachusetts. Estuarine, Coastal, and Shelf Science 12:701-712.

Todd, R., R. Lowrance, O. Hendrickson, L. Asmussen, R. Leonard, J. Fail, and B. Herrick. 1983. Riparian vegetation as filters of nutrients exported from a coastal plain agricultural watershed. Pages 485-493 in R. Lowrance, R. L. Todd, L. E. Assmussen, and R. A. Leonard (eds.), Nutrient Cycling in Agricultural Ecosystems. Univ. of Georgia, Coll. of Agric. Experiment Stations, Spec. Publ. No. 23.

Tonn, W. M., and J. J. Magnuson. 1982. Patterns in the species composition and richness of fish assemblages in northern Wisconsin lakes. Ecology 63:1149-1166.

Topinka, J., L. Tucker, and W. Korjeff. 1981. The distribution of fucoid macroalgal biomass along central coastal Maine. Bot. Mar. 24:311-319.

Toth, J. 1971. Groundwater discharge: A common generator of diverse geologic and morphologic phenomena. Bull. Int. Sci. Hydrol. 16(1.3):7-24.

Toth, L. 1972. Reeds control eutrophication of Balaton Lake. Water Research 6:1533-1539.

Tourbier, J., and R. Westmacott. 1974. Water resources protection measures in land development—A handbook. Final Report for Dept. of the Interior (Contract No. 14-31-0001-9036). Univ. of Pennsylvania, Philadelphia.

Tramer, F. J., and D. E. Suhrweir. 1975. Farm woodlots as biogeographic islands: Regulation of tree species richness. Bull. Ecol. Soc. Am. 56:53.

Tremblay, J., and L. N. Ellison. 1979. Effects of human disturbance on breeding of black-crowned night herons. Auk 96:364-369.

Trent, L., E. J. Pullen, and R. Proctor. 1976. Abundance of macrocrustaceans in a natural marsh and a marsh altered by dredging, bulkheading and filling. Fish. Bull. 74:195-200.

Triska, F. J., J. R. Sedell, K. Cromack, Jr., S. V. Gregory, and F. M. McCorison. 1984. Nitrogen budget for a small coniferous forest stream. Ecol. Monogr. 54:119-140.

Trotzky, H. M., and R. W. Gregory. 1974. The effects of water flow manipulation below a hydroelectric power dam on the bottom fauna of the upper Kennebec River, Maine. Tran. Am. Fish. Soc. 113:142-150.

Turk, T. R., M. J. Risk, R. W. M. Hirtle, and R. K. Yeo. 1980. Sedimentological and biological changes in the Windsor mudflat, an area of induced siltation. Can. J. Fish Aq. Sci. 37:1387-1397.

Turner, R. E. 1977. Intertidal vegetation and commercial yields of penaeid shrimp. Trans. Am. Fish. Soc. 106:411-416.

Turner, R. E. 1978. Community plankton respiration in a salt marsh estuary and the importance of macrophytic leachates. Limnol. Oceanog. 23:442-451.

Turner, R. E., W. W. Woo, and H. R. Jitts. 1979. Estuarine influences on a continental shelf plankton community. Science 206:218-220.

Turner, R. R. 1980. Impacts of water level fluctuation on physical and chemical characteristics of reservoirs. Pages 3-19 in S. G. Hildebrand (ed.), Analysis of environmental issues related to small scale hydroelectric development; III. Water level fluctuations. Oak Ridge Nat. Lab., Environ. Sci. Div., Publication No. 1591, Oak Ridge, TN.

Turner, R. R., E. A. Laws, and R. C. Harriss. 1983. Nutrient retention and transformation in relation to hydraulic flushing rate in a small impoundment. Freshw. Biol. 13:113-127.

Tyler, A. V. 1971. Surges of winter flounder, *Pseudopleuronectes americanus*, into the intertidal zone. J. Fish. Res. Bd. Can. 28:1727-1732.

US Army Corps of Engineers. 1977. Shore Protection Manual; Vol 1:3-49-3-53. Coastal Engineering Research Center, Fort Belvoir, VA.

US Army Corps of Engineers, Lower Mississippi Valley Division. 1980. A Habitat Evaluation System (HES) for Water Resources Planning. Vicksburg, MS.

US Fish and Wildlife Service. 1980. Habitat Evaluation Procedures (HEP). Ecol. Services Manual 101. US Fish and Wildl. Serv., Washington, DC. 84 p.

Valiela, I. 1984. Marine Ecological Processes. Springer-Verlag, New York.

Valiela, I., and J. M. Teal. 1979. The nitrogen budget of a salt marsh ecosystem. Nature 280:652-656.

Valiela, I., J. M. Teal, C. Cogswell, S. Allen, D. Goehringer, R. Van Etten, and J. Hartman. 1985. Some long-term consequences of sewage contamination in salt marsh ecosystems. Pages 301-316 in P. J. Godfrey, E. R. Kaynor, S. Pelczarski, and J. Benforado (eds.), Ecological Considerations in Wetland Treatment of Municipal Wastewater. Van Nostrand Reinhold, New York.

Valiela, I., J. M. Teal, and W. Sass. 1973. Nutrient retention in salt marsh plots experimentally fertilized with sewage sludge. Est. Coast. Mar. Sci. 1:261-269.

Valiela, I., J. M. Teal, and W. J. Sass. 1975. Production and dynamics of salt marsh vegetation and the effects of experimental treatment with sewage sludge: Biomass, production and species composition. J. Appl. Ecol. 12:973-981.

Valiela, I., J. M. Teal, and N. Y. Persson. 1976. Production and dynamics of experimentally enriched salt marsh vegetation: Below-ground biomass. Limnol. Oceanogr. 21:245-252.

Valiela, I., J. M. Teal, S. Volkmann, D. Shafer, and E. J. Carpenter. 1978. Nutrient and particulate fluxes in a salt marsh ecosystem: Tidal exchanges and inputs by precipitation and groundwater. Limnol. Oceanogr. 23:798-812.

Valiela, I., J. M. Teal, S. Volkmann, R. A. Van Etten, and S. Allen. 1984. Decomposition in salt marsh ecosystems: The phases and major factors affecting disappearance of aboveground organic matter. J. Exp. Mar. Biol. Ecol. 89:29-54.

Vallentyne, J.R. 1952. Insect removal of nitrogen and phosphorus compounds from lakes. Ecology 33:573-577.

Van der Valk, A. G., J. L. Baker, C. B. Davis, and C. E. Beer. 1979. Natural freshwater wetlands as nitrogen and phosphorus traps for land runoff. Pages 457-467 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Van der Valk, A. G., S. D. Swanson, and R. F. Nuss. 1983. The response of plant species to burial in three types of Alaskan wetlands. Can. J. Bot. 61:1150-1164.

Van Hassel, J. H., J. J. Ney, and D. L. Garling, Jr. 1980. Heavy metals in a stream ecosystem at sites near highways. Trans. Am. Fish. Soc. 109:636-643.

Van Kessel, J. F. 1978. Gas production in aquatic sediments in the presence and absence of nitrate. Water Res. 12:291-297.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Can. J. Fish Aq. Sci. 37:130-137.

Van Voast, W. A., and R. P. Novitzki. 1968. Ground-water flow related to stream-flow and water quality. Water Resour. Res. 4:769-775.

Varshney, C. K., and Mandhan. 1982. Estimation of nitrogen fixation associated with *Typha*. Aquat. Bot. 13:351-357.

Vaske, J. J., A. R. Graefe, and F. R. Kuss. 1982. Recreation impacts: A synthesis of ecological and social research. Trans. 48th North American Wildl. and Nat. Resour. Conf. 48:96-107.

Vecchioli, J., H. E. Gill, and S. M. Lang. 1962. Hydrologic role of the Great Swamp and other marshland in Upper Passaic River Basin. Am. Water Works Assoc. J. 54:697-701.

Verhoff, E. H., D. A. Melti, and S. M. Yaksich. 1982. An analysis of total phosphorus transport in river systems. Hydrobiol. 91:241-252.

Verry, E. S., and D. H. Boelter. 1979. Peatland hydrology. Pages 389-402 in P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. Am. Water Resour. Assoc., Minneapolis, MN.

Verry, E. S. 1986. Forest harvesting and water: The lake states experience. Water Resour. Bull. 22(6):1039-1047.

Voights, D. K. 1976. Aquatic invertebrate abundance in relation to changing marsh vegetation. Am. Midl. Nat. 95(2):313-322.

Vollenweider, R. A. 1976. Advances in defining critical loading levels of phosphorus in lake eutrophication. Mem. Ist. Ital. Idrobiol. 33:53-83.

Vorhees, L. D., and J. F. Cassel. 1980. Highway right-of-way: Mowing versus succession as related to duck nesting. J. Wildl. Manage. 44:155-163.

Voshell, J. R., and C. R. Parker. 1985. Quantity and quality of seston in an impounded and a free-flowing river in Virginia, USA. Hydrobiol. 122:271-280.

Walker, D. 1970. Direction and rate in some British post-glacial hydroseres. Pages 117-139 in D. Walker and R. G. West (eds.), Studies in the vegetational history of the British Isles. Cambridge University Press, Cambridge, England.

Walker, M. D., and R. Sniffen. 1985. Fish utilization of an inundated swamp-stream floodplain. EPA-600/3-85-046. US Environmental Protection Agency, Environmental Res. Lab, Corvallis, Oregon. 73 p.

Wallis, M., H. B. N. Hynes, and S. A. Telang. 1981. The importance of groundwater in the transportation of allochthonous dissolved organic matter to the streams draining a small mountain basin. Hydrobiol. 79:77-90.

Ward, L. G., W. M. Kemp, and W. R. Boyton. 1984. The influence of waves and seagrass communities on suspended particles in an estuarine embayment. Mar. Geol. 59(1-4):85-103.

Ware, F. J., and R. O. Gasaway. 1978. Effects of grass carp on native fish populations in two Florida lakes. Proc. Southeast. Assoc. Fish Wildl. Agenc. 30:324-335.

Warwick, R. M. 1988. Effects on community structure of a pollutant gradient—summary. Mar. Ecol. Prog. Ser. 46:207-211.

Watson, G. F., A. I. Robertson, and M. L. Littlejohn. 1984. Invertebrate macrobenthos of the seagrass community in Western Port, Victoria. Aquat. Bot. 18:175-197.

Wayne, C. J. 1976. The effect of sea and marsh grass on wave energy. Coastal Res. 4:6-8

Weiler, P. R. 1978. Littoral-pelagic water exchange in Lake Wingra, Wisconsin, as determined by a circulation model. Report 100. Inst. Environ. Studies, Univ. of Wisconsin, Madison. 34 p.

Weinstein, M. P. 1979. Shallow marsh habitats as primary nurseries for fishes and shellfish, Cape Fear River, North Carolina. Fish. Bull. 77:339-357.

Weinstein, M. P., and H. A. Brooks. 1983. Comparative ecology of nekton residing in a tidal creek and adjacent seagrass meadow: Community composition and structure. Mar. Ecol. Prog. Ser. 12:15-27.

Weinstein, M. P., S. L. Weiss, and M. F. Walters. 1980. Multiple determinants of community structure in shallow marsh habitats, Cape Fear River Estuary, North Carolina, USA. Mar. Biol. 58:227-243.

Welch, H. E., J. K. Jorgenson, and M. F. Curtis. 1988. Emergence of Chironomidae (Diptera) in fertilized and natural lakes at Saqvaqjuac, N.W.T. Can. J. Fish. Aquat. Sci. 45:731-736.

Welcomme, R. L. 1979. Fisheries Ecology of Floodplain Rivers. Longman, New York. 317 p.

Weller, M. W. 1975. Migratory waterfowl: A hemispheric perspective. Publicaciones Biologicas Instituto De Investigaciones Cientificas, UANL, Mexico 1(7):89-130.

Weller, M. W. 1978. Management of freshwater marshes for wildlife. Pages 267-284 in R. E. Good, D. F Whigham, and R. L. Simpson (eds.), Freshwater Wetlands - Ecological Processes and Management Potential. Academic Press, New York.

Weller, M. W. 1979. Density and habitat relationship of blue-winged teal nesting in northwestern Iowa. J. Wildl. Manage. 43:367-374.

Weller, M. W., and C. S. Spatcher. 1965. Role of habitat in the distribution and abundance of marsh birds. Iowa Agr. Home Econ. Exp. Stn. Spec. Rep. No. 43. 31 p.

Weller, M. W., and L. H. Fredrickson. 1974. Avian ecology of a managed glacial marsh. Liv. Bird 12:269-291.

Welsh, B. L. 1980. Comparative nutrient dynamics of a marsh-mudflat ecosystem. Est. Coast. Mar. Sci. 10:142-164.

Welsh, G. L., J. P. Herring, D. Bressette, and L. Read. 1976. The importance of an holistic approach to ecosystem management and community planning. Pages 16-33 in M. W. Lefor, W. C. Kennard, and T. B. Helfgott (eds.), Proc. 3rd Wetlands Conf. Univ. of Conn. Inst. Water Resour. Rep. No. 26, Storrs, CT.

Werme, C. 1981. Resource partitioning in a salt marsh fish community. Ph.D. thesis, Boston University, Boston, MA.

Werschkul, D. F., E. McMahon, and M. Leitschuh. 1976. Some effects of human activities on the great blue heron in Oregon. Murrelet 58:7-12.

Wesche, T. A., and P. A. Rechard. 1980. A summary of instream flow methods for fisheries and related research needs. Univ. Wyom. Water Resour. Res. Inst., Eisenhower Consort. Bull. No. 9, Laramie. 122 p.

Wesche, T. A., C. M. Goertler, and C. B. Frye. 1987. Contribution of riparian vegetation to trout cover in small streams. N. Am. J. Fish. Manage. 7:151-153.

Westerdahl, H. E., W. B. Ford III, J. Harris, and C. R. Lee. 1981. Evaluation of techniques to estimate annual water quality loadings to reservoirs. Tech. Rep. E-81-1. US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Westlake, D. 1963. Comparison of plant productivity. Biol. Rev. 38:385-425.

Wetzel, R. 1975. Limnology. W. B. Saunders Co., Philadelphia, PA. 743 p.

Wetzel, R. G. 1979. The role of the littoral zone and detritus in lake metabolism. Arch. Hydrobiol. Beih. Ergebn. Limnol. 13:145-161.

Wetzel, R. M. 1958. Mammalian succession on midwestern floodplains. Ecology 39:262-271.

Wetzel. R. G. 1983. Attached algal-substrate interactions: Fact or myth, and when and how? Pages 207-215 in R. G. Wetzel (ed.), Periphyton in Freshwater Ecosystems. Junk, The Hague, Netherlands.

Wharton, C. H. 1970. The Southern River Swamp: A multiple use environment. Ga. St. Univ. Bur. Bus. Econ. Res., Atlanta. 48 p.

Whetstone, B.H. 1982. Techniques for estimating magnitude and frequency of floods in South Carolina. Water Resour. Investig. Rep. 82-1. US Geological Survey, Reston, VA.

Whigham, D. F., and R. L. Simpson. 1976. The potential use of tidal marshes in the management of water quality in the Delaware River. Pages 173-186 in J. Tourbier and R. W. Pierson, Jr. (eds.), Biological control of water pollution. Univ. Penn. Press, Philadelphia.

Whigham, D., R. L. Simpson, and K. Lee. 1980. The effect of sewage effluent on the structure and function of a freshwater tidal marsh ecosystem. New Jersey Water Resources Research Institute, Rutgers University, State University of New Jersey, New Brunswick, NJ. 160 p.

Whigham, D. F., C. Chitterling, E. Palmer, and J. O'Neill. 1986. Modification of runoff from upland watersheds. Pages 305-332 in D. L. Correll (ed.), Watershed Research Perspectives. Smithsonian Institution Press, Washington, DC.

Whipple, W., Jr., J. M. DiLouie, and T. Pytlar, Jr. 1981. Erosional potential of streams in urbanizing areas. Water Resources Bulletin 17:36-45.

Whipple, W., Jr., and J. DiLouie. 1981. Coping with increased stream erosion in urbanizing areas. Water Resour. Bull. 17:1561-1564.

Whitaker, G. A., R. H. McCuen, and J. Brush. 1979. Channel modification and macroinvertebrate community diversity in small streams. Water Res. Bull. 15(3):874-879.

White, D. H., C. A. Mitchell, and E. Cromartie. 1982. Nesting ecology of roseate spoonbills at Nueces Bay, Texas. Auk 99:275-284.

White, D. S., C. D'Avanzo, I. Valiela, C. Lasta, and M. Pascual. 1986. The relationship of diet to growth and ammonium excretion in salt marsh fish. Envir. Biol. Fishes 16:105-111.

Whitehead, W. R., and E. J. Langhtee. 1978. Use of bounding wells to counteract the effects of preexisting groundwater movement. Water Resour. Res. 14:273-277.

Whiteside, M. C. 1970. Danish chydorid Cladocera: Modern ecology and core studies. Ecol. Monogr. 40:79-118.

Whitlatch, R. B. 1982. The ecology of New England tidal flats: A community profile. FWS/OBS-81/01. US Fish Wildl. Serv., Washington, DC. 125 p.

Whitlatch, R. B. 1980. Patterns of resource utilization and coexistence in marine intertidal deposit-feeding communities. J. Mar. Res. 38:743-765.

Whitman, W. R., and R. V. Cole. 1987. Ecological conditions and implications for waterfowl management in selected coastal impoundments of Delaware. Pages 99-119 in W. R. Whitman and W. H. Meredith (eds.), *Proceedings of a Symposium on Waterfowl and Wetlands Management in the Coastal Zone of the Atlantic Flyway*. Delaware Dept. of Natural Resources and Environmental Control, Dover, DE.

Whitmore, R. C. 1975. Habitat ordination of passerine birds of the Virgin River Valley, southwestern Utah. Wilson Bull. 87:65-74.

Wiebe, W. J., R. E. Johannes, and K. L. Webb. 1978. Nitrogen fixation in a coral reef community. Science 188:257-259.

Wieder, R. K., and G. E. Lang. 1982. Modification of acid mine drainage in a freshwater wetland. Pages 43-53 in B. R. McDonald (ed.), Proc. Symp. Wetlands of the Unglaciated Appalachian Region, W. Va. Univ., Morgantown.

Wiener, J.G. 1987. Metal contamination of fish in low-pH lakes and potential implications for piscivorous wildlife. Pages 645-657 in Trans. 52nd N. Am. Wildl. and Nat. Res Conf.

Wigington, P. J., Jr., C. W. Randall, and T. J. Gizzard. 1986. Accumulation of selected trace metals in soils of urban runoff swale drains. Water Resources Bull. 22(1):73-79.

Wilcove, D. S. 1985. Nest predation in forest tracts and the decline of migratory songbirds. Ecology 66:1211-1214.

Wilcox, D. A., R. J. Shedlock, and W. R. Henrickson. 1986. Hydrology, water chemistry and ecological relations in the raised mound of Cowles Bog. J. Ecol. 74:1103-1117.

Wiley, M. J., R. W. Gorden, S. W. Waite, and T. Powless. 1984. The relationship between aquatic macrophytes and sport fish production in Illinois ponds: A simple model. N. Am. J. Fish. Manage. 4:111-119.

Willard, D. E. 1977. The feeding ecology and behavior of five species of herons in southeastern New Jersey. Condor 79:462-470.

Willford, W. A., M. J. Mac, and R. J. Hesselberg. 1987. Assessing the bioaccumulation of contaminants from sediments by fish and other aquatic organisms. Hydrobiol. 149:107-111.

Williams, D. D., and H. B. N. Hynes. 1977. Benthic community development in a new stream. Can. J. Zool. 55:1071-1076.

Williams, D. D. 1980. Some relationships between stream benthos and substrate heterogeneity. Limnol. Oceanogr. 25:166-172.

Williams, G. P. 1986. River meanders and channel size. J. Hydrol. 88:147-164.

Williams, J. D., and C. K. Dodd, Jr. 1979. Importance of wetlands to endangered and threatened species. Pages 565-575 in P. E. Greeson et al. (eds.), Wetland functions and values: The state of our understanding. Tech Publ. 79-2. Am. Water Resour. Assoc., Minneapolis, MN.

Williams, L. E., Jr., and L. L. Martin. 1970. Nesting populations of brown pelicans in Florida. Proc. Southeast. Assoc. Game Fish Comm. 24:154-169.

Williams, R. E. 1968. Flow of groundwater adjacent to a small, closed basin in glacial till. Water Resour. Res. 4:777-783.

Willson, M. F. 1974. Avian community organization and habitat structure. Ecology 55:1017-1029.

Wilson, E. O., and E. O. Willis. 1975. Applied biogeography. Pages 522-534 in M. L. Cody and J. M. Diamond (eds.), Ecology and Evolution of Communities. Harvard Univ. Press, Cambridge, MA.

Wilson, J. O., I. Valiela, and T. Swain. 1987. Decomposition processes in a salt marsh ecosystem: Carbohydrate dynamics during decay of litter of *Spartina alterniflora*. Mar. Biol. 92:277-284.

Wilson, W. H., T. Brush, R. A. Lent, and B. A. Harrington. 1987. The effects of grid-ditching and open marsh water management on avian utilization of Massachusetts salt marshes. Pages 334-348 in W. R. Whitman and W. H. Meredith (eds.), Proceedings of a Symposium on Waterfowl and Wetlands Management in the Coastal Zone of the Atlantic Flyway. Delaware Dept. of Natural Resources and Environmental Control, Dover.

Windom, H. L. 1977. Ability of salt marshes to remove nutrients and heavy metals from dredged material disposal area effluents. Tech. Rep. D-77-37. US Army Engineer Waterways Experiment Station, Vicksburg, MS. 43 p.

Windom, H., G. Wallace, R. Smith, N. Dudek, M. Maeda, R. Dulmage, and F. Shorti. 1983. Behavior of copper in southeastern United States estuaries. Mar. Chem. 12:183-193.

Winter, T. C. 1977. Classification of the hydrologic settings of lakes in the north central United States. Water Resour. Res. 13:753-767.

Winter, T. C. 1981. Uncertainties in estimating the water balance of lakes. Water Resour. Bull. 17(1):82-115.

Winter, T. C. 1983. The interaction of lakes with variably saturated porous media. Water Resour. Res. 19:1203-1218.

Winter, T. C., and M. R. Carr. 1980. Hydrologic setting of wetlands in the Cottonwood Lake area, Stutsman County, North Dakota. Pages 1-42 in Water Resour. Invest. 80-99. US Geological Survey, Reston, VA.

Wolaver, T. G., R. L. Wetzel, J. C. Zieman, and K. L. Webb. 1980. Nutrient interactions between salt marsh, mudflats and estuarine water. Pages 123-134 in V. S. Kennedy (ed.), Estuarine Perspectives. Academic Press, New York.

Wolaver, T. G., W. Johnson, and M. Marozas. 1984. Nitrogen and phosphorus concentrations within North Inlet, South Carolina—Speculation as to sources and sinks. Est. Coast. Shelf Sci. 19:243-255.

Wolaver, T. G., and J. D. Spurrier. 1988. Carbon transport between a euhaline vegetated marsh in South Carolina and the adjacent tidal creek: Contributions via tidal inundation, runoff and seepage. Mar. Ecol. Prog. Ser. 42:53-62.

Wolaver, T. G., and J. D. Spurrier. 1988. The exchange of phosphorus between a euhaline vegetated marsh and the adjacent tidal creek. Estuarine Coastal Shelf Sci. 26:203-214.

Wolaver, T. G., R. F. Dame, J. D. Spurrier, and A. B. Miller. 1988. Sediment exchange between a euhaline salt marsh in South Carolina and the adjacent tidal creek. J. Coast. Res. 4(1):17-26.

Wolman, M. G., and A. P. Schick. 1967. Effects of construction on fluvial sediment, urban and suburban areas of Maryland. Water Resour. Res. 3:451-464.

Wolverton, B. C., and M. M. McKown. 1976. Water hyacinths for removal of phenols from polluted water. Aquat. Bot. 30:29-37.

Woo, M. K., and J. Valverde. 1981. Summer streamflow and water level in a mid-latitude forested swamp. For. Sci. 27(1):177-189.

Wood, C. C. 1987. Predation on juvenile Pacific salmon by the common merganser (*Mergus merganser*) on eastern Vancouver Island; II. Predation of stream resident juvenile salmon by merganser broods. Can. J. Fish. Aquat. Sci. 44:950-959.

Wood, W. W., and W. R. Osterkamp. 1984. Recharge to the Ogallala aquifer from playa lake basins in the Llano Estacado. Pages 337-349 in G. A. Whetstone (ed.), Proc. Ogallala Aquifer Sympos. II. Texas Tech University, Lubbock.

Wright, L. D., and A. T. Szluha. 1980. Impacts of water level fluctuations on biological characteristics of reservoirs. Pages 21-38 in S. G. Hildebrand (ed.), Analysis of environmental issues related to small scale hydroelectric development; III: Water level fluctuations. Oak Ridge Nat. Lab., Envir. Sci. Div. Publ. No. 1591. Oak Ridge, TN.

Wycoff, R. L., and D. G. Pyne. 1975. Urban water management and coastal wetland protection in Collier County, Florida. Water Resour. Bull. 2:455-469.

Yakupzack, P. M., W. H. Herke, and W. G. Perry. 1977. Emigration of juvenile Atlantic croakers, *Micropogon undulatus*, from a semi-impounded marsh in southwestern Louisiana. Tran. Am. Fish. Soc. 106:538-544.

Yarbro, L. A. 1979. Phosphorus cycling in the Creeping Swamp Floodplain ecosystem and exports from the Creeping Swamp watershed. Ph.D. dissertation, Univ. of North Carolina, Chapel Hill. 231 p.

Yarbro, L. A. 1983. The influence of hydrologic variations on phosphorus cycling and retention in a swamp stream ecosystem. Pages 223-245 in T. D. Fontaine and S. M. Bartell (eds.), Dynamics of Lotic Ecosystems. Ann Arbor Science, Ann Arbor, MI.

Yates, P., and J. M. Sheridan. 1983. Estimating the effectiveness of vegetated floodplains/wetlands as nitrate-nitrite and orthophosphorus filters. Agric. Ecosyst. and Environ. 9:303-314.

Yates, R. F. K., and F. P. Day. 1983. Decay rates and nutrient dynamics in confined and unconfined leaf litter in the Great Dismal Swamp. Am. Midl. Nat. 110:37-45.

Yeager, L. E. 1949. Effect of permanent flooding in a riverbottom timber area. Ill. Nat. Hist. Surv. Bull. 35:33-65.

Yoder, J. A., L. P. Atkinson, J. O. Blanton, D. R. Deibel, D. W. Menzel, and G. A. Paffenhofer. 1981. Plankton productivity and the distribution of fishes on the southeast continental shelf. Science 214:352-353.

Yorke, T. H., and W. J. Herb. 1976. Urban-area sediment yield—effects of construction-site-conditions and sediment-control methods. Pages 2-52 to 2-64 in Proc. of the Third Federal Interagency Sedimentation Conference, Denver, CO. US Water Resources Council, Washington, DC.

Young, C. E., and R. A. Klawitter. 1968. Hydrology of wetland forest watersheds. Pages 29-38 in Proceedings, Hydrology in Water Resources Management. Report No. 4. Water Resources Institute, Clemson University, Clemson, SC.

Zajac, R. M., and R. B. Whitlatch. 1982. Responses of estuarine infauna to disturbance; I. Spatial and temporal variation of initial recolonization. Mar. Ecol. Prog. Ser. 10:1-14.

Zedler, J. B. 1982. The ecology of Southern California coastal salt marshes: A community profile. FWS/OBS-81/54. US Fish Wildl. Serv., Washington, DC. 110 p.

Zedler, J. B., T. Winfield, and P. Williams. 1980. Salt marsh productivity with natural and altered tidal circulation. Oecologia 44:236-240.

Zieman, J. C. 1982. The ecology of the seagrasses of South Florida: a community profile. FWS/OBS-82/52. US Fish and Wildlife Service, Washington, DC. 123 p.

Zimmer, D. W., and R. W. Bachman. 1978. Channelization and invertebrate drift in some Iowa streams. Water Res. Bull. AWRA 14(4):868-883.

Zimmerman, R. J., T. J. Minello, and G. Zamora, Jr. 1984. Selection of vegetated habitat by brown shrimp, *Penaeus aztecus*, in a Galveston Bay salt marsh. Fish. Bull. 82:325-332.

Zimmerman, J. L. 1977. Virginia rail. Pages 46-56 in G. C. Sanderson (ed.), Management of migratory shore and upland game birds in North America. Inst. Assoc. Fish Wildl. Agen., Washington, DC.

Zimmerman, R. C., J. C. Goodlet, and G. H. Comer. 1967. The influence of vegetation on channel form of small streams. Pages 255-275 in Symp. on River Morphology; Reports and Discussions. Publ. No. 75. International Assoc. of Sci. Hydrol., General Assembly of Brn, 25 Sept.-7 Oct. 1967. L'Association Internationale d'-Hydrologie Asientifique, Gentbrugge, Belguque (Belgium).

Zimmerman, R. J., and T. J. Minello. 1984. Densities of *Penaeus aztecus*, and other natant macrofauna in a Texas salt marsh. Estuaries 7:421-433.

Ziser, S. W. 1978. Seasonal variations in water chemistry and diversity of the phytophilic macroinvertebrates of three swamp communities in southeastern Louisiana. Southwest. Nat. 23(4):545-562.

Zoltec, J., Jr., S. E. Bayley, A. J. Herman, C. R. Tortora, and T. J. Dolan. 1979. Removal of nutrients from treated municipal wastewater by freshwater marshes. Final report to the City of Clermont, Florida. Center for Wetlands, Univ. of Florida, Gainesville, FL. 325 p.

Appendix A: Scientific Names of Animals and Plants Mentioned in Text

Common Name	Scientific Name
Birds	
black duck	*Anas rubripes*
American coot	*Fulica americana*
bald eagle	*Haliaeetus leucocephalus*
bank swallow	*Riparia riparia*
black-bellied tree duck	*Dendrocygna autumnalis*
black skimmer	*Rynchops niger*
brant	*Branta bernicla*
bufflehead	*Bucephala albeola*
Canada goose	*Branta canadensis*
canvasback	*Aythya valisineria*
cliff swallow	*Petrochelidon pyrrhonota*
common goldeneye	*Bucephala clangula*
common merganser	*Mergus merganser*
fulvous tree duck	*Dendrocygna bicolor*
gadwall	*Anas strepera*
greater scaup	*Aythya marila*
white-fronted goose	*Anser albifrons*
harlequin duck	*Histrionicus histrionicus*
hooded merganser	*Lophodytes cucullatus*
least bittern	*Ixobrychus exilis*
lesser scaup	*Aythya affinis*
palm warbler	*Dendroica palmarum*
brown pelican	*Pelecanus occidentalis*
prothonotary warbler	*Protonotaria citrea*
red-breasted merganser	*Mergus serrator*
redhead	*Aythya americana*
ring-necked duck	*Aythya collaris*
ring-necked pheasant	*Phasianus colchicus*
roseate spoonbill	*Ajaia ajaja*
Ross' goose	*Chen rossii*
ruddy duck	*Oxyura jamaicensis*
seaside sparrow	*Ammospiza maritimus*
snow goose	*Chen caerulescens*
sora rail	*Porzana carolina*
whistling swan	*Olar columbianus*
wood duck	*Aix sponsa*
yellow-headed blackbird	*Xanthocephalus xanthocephalus*
Mammals	
beaver	*Castor canadensis*
manatee	*Trichechus manatus*
mink	*Mustela vison*
muskrat	*Ondatra zibethicus*
nutria	*Myocastor coypus*
raccoon	*Procyon lotor*
river otter	*Lutra canadensis*
sea otter	*Enhydra lutris*
swamp rabbit	*Sylvilagus aquaticus*
water shrew	*Sorex palustris*
white-tailed deer	*Odocoileus viginianus*
Virginia opossum	*Didelphis virginiana*
(Continued)	

Common Name	Scientific Name
Fish	
common carp	*Cyprinus carpio*
grass carp	*Ctenopharyngodon idella*
largemouth bass	*Micropterus salmoides*
mummichog	*Fundulus heteroclitus*
muskellunge	*Esox masquinongy*
northern pike	*Esox lucius*
sauger	*Stizostedion canadense*
walleye	*Stizostedion vitreum*
winter flounder	*Pseudopleuronectes americanus*
yellow perch	*Perca flavescens*
Herpetofauna	
American alligator	*Alligator mississippiensis*
bog turtle	*Clemmys muhlenbergii*
loggerhead sea turtle	*Caretta caretta*
Crustaceans	
blue crab	*Callinectes sapidus*
brown shrimp	*Penaeus aztecus*
Plants	
bayberry	*Myrica pensylvanica*
black spruce	*Picea mariana*
bulrush	*Scirpus* spp.
cattail	*Typha* spp.
cypress	*Taxodium distichum*
duckweed	*Lemna* spp.
freshwater cordgrass	*Spartina pectinata*
pondweed	*Potamogeton* spp.
reed	*Phragmites australis*
salt marsh cordgrass	*Spartina alterniflora*
speckled alder	*Alnus rugosa*
waterhyacinth	*Eichornia crassipes*
water lettuce	*Pistia stratiotes*