Evaluating Cumulative Effects on Wetland Functions: A Conceptual Overview and Generic Framework

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ABSTRACT / This article outlines conceptual and methodological issues that must be confronted in developing a sound scientific basis for investigating cumulative effects on freshwater wetlands. We are particularly concerned with: (1) effects expressed at temporal and spatial scales beyond those of the individual disturbance, specific project, or single wetland, that is, effects occurring at the watershed or regional landscape level; and (2) the scientific (technical) component of the overall assessment process. Our aim is to lay the foundation for a research program to develop methods to quantify cumulative effects of wetland loss or degradation on the functioning of interacting systems of wetlands. Toward that goal we: (1) define the concept of cumulative effects in terms that permit scientific investigation of effects; (2) distinguish the scientific component of cumulative impact analysis from other aspects of the assessment process; (3) define critical scientific issues in assessing cumulative effects on wetlands; and (4) set up a hypothetical and generic structure for measuring cumulative effects on the functioning of wetlands as landscape systems.

We provide a generic framework for evaluating cumulative effects on three basic wetland landscape functions: flood storage, water quality, and life support. Critical scientific issues include appropriate delineations of scales, identification of threshold responses, and the influence on different functions of wetland size, shape, and position in the landscape.

The contribution of a particular wetland to landscape function within watersheds or regions will be determined by its intrinsic characteristics, e.g., size, morphometry, type, percent organic matter in the sediments, and hydrologic regime, and by extrinsic factors, i.e., the wetland's context in the landscape mosaic. Any cumulative effects evaluation must take into account the relationship between these intrinsic and extrinsic attributes and overall landscape function. We use the magnitude of exchanges among component wetlands in a watershed or larger landscape as the basis for defining the geographic boundaries of the assessment. The time scales of recovery for processes controlling particular wetland functions determine temporal boundaries. Landscape-level measures are proposed for each function.

Ideas can have strong intuitive appeal, yet not affect decisionmaking because they lack any explicit operational formulation. *Cumulative impact* is such an idea. Cumulative effects on freshwater wetland ecosystems are more tangible than any scientific basis for measuring these effects. The notion that individually insignificant actions can produce major change through the accumulation of effects is compelling enough to have influenced federal legislation (most notably, the National Environmental Policy Act), initiated court action, and produced international meetings (Beanlands and others 1986). Yet constraints remain more obvious than any specific approach or method for implementing this idea in natural resource regulation and management.

As one of several federal agencies responsible for regulating the nation's wetland resources, the U.S. Environmental Protection Agency (EPA) has undertaken a research program to overcome some of the technical constraints on cumulative impact evaluation. The program is intended to support both the agency's oversight and permit-review function under Section 404 of the Clean Water Act, and recent initiatives to develop more holistic and anticipatory approaches to wetland protection (see Hirsch 1988, Lee and Gosselink 1988). The latter initiatives, according to EPA Administrator, Lee Thomas, include "identifying geographic areas, wetland types, and wetland impacts that merit a special measure of attention" (Thomas 1986, press release) in advance of development and permit decisions. Such approaches present the opportunity for a comprehensive, top-down approach to regulating cumulative impact. The former bottom-up approach operates on a project-by-project basis and constitutes the major challenge to efforts to evaluate cumulative impact. This and the following articles form part of EPA's research

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program to develop technical information relevant to both regulatory approaches.

In this article we outline the conceptual and methodological issues that must be confronted in developing a sound scientific basis for investigating cumulative effects on freshwater wetlands. We are particularly concerned with: (1) effects expressed at temporal and spatial scales beyond those of the individual disturbance, specific project, or single wetland, i.e., effects occurring at the watershed or regional landscape level; and (2) the scientific, (technical) component of the overall assessment process.

Our approach draws heavily on papers in Beanlands and others (1986) and on recent changes in the way ecosystems are perceived (Risser and others 1984, Levin 1987), including the delineation of appropriate scales and variables for ecosystem study, identification of the key linkages and interactions within ecosystems, and recognition of the predominance in natural systems of multiple stresses.

Our aim is to lay the foundation for a research program to develop methods to quantify cumulative effects of wetland loss or degradation on the functioning of interacting systems of wetlands. Toward that goal we: (1) define the concept of cumulative effects in terms that permit scientific investigation of effects, (2) distinguish the scientific component of cumulative impact analysis from other aspects of the assessment process, and (3) define critical scientific issues in assessing cumulative effects on wetlands. We conclude by setting up a hypothetical and generic structure for measuring cumulative effects on the functioning of wetlands as landscape systems (Forman and Godron 1986). In so doing we hope to define the problem more clearly, but not necessarily to create an operational assessment strategy. Some of the specific ideas and methods for that structure, as well as the intended challenges to it, are provided in the following articles.

Concepts of Cumulative Impact

Cumulative impact has been defined in many ways. The definition we use here is regulatory, stemming from the National Environmental Policy Act of 1969 (NEPA). Regulations published by the Council on Environmental Quality (CEQ) to implement NEPA require that environmental impact statements prepared to comply with NEPA "anticipate a cumulatively significant impact on the environment" (38 CFR 1500.6). Cumulative impacts are defined as:

> the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-Federal) or person undertakes such actions. Cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time (40 CFR 1508.7).

This definition recognizes generally that the effects of one project may contribute to and interact with effects of other projects. However, as part of a legislative document intended to give broad policy guidance, it does so without specifying how those effects might interact or how they are to be measured.

Thus, discussion frequently arises over what is meant by the term cumulative impacts. The discussion is important because, as Baskerville (1986) and Clark (1986) have pointed out, "impacts in biological systems accumulate in different ways," and recognition of how they accumulate is central to developing a scientific approach to evaluating cumulative impact. In the following sections, we discuss various concepts related to the scientific framework we seek to develop. We begin by differentiating cumulative impact evaluation from conventional impact evaluation.

Cumulative Impact Evaluation vs Conventional Impact Evaluation

An essential difference between conventional impact assessment and cumulative impact assessment lies in the manner in which spatial and temporal boundaries of the evaluation are established (Baskerville 1986). Cumulative impact assessment takes a broader view; the boundaries it draws in regard to the number of disturbances, the geographic area, and the time frame considered are larger. Conventional impact assessments are typically bounded by the expected zone of influence of a single disturbance or proposed project. The effects on environmental resources falling within that zone are then estimated. While such a bounding process allows evaluation of the local impacts on resources, it does not allow evaluation of impacts of the project on these resources as a whole, of the total impact on these resources from all anthropogenic disturbances, or of secondary impacts resulting from the interaction of impacts from the project with other anthropogenic disturbances. This is true because the spatial and temporal boundaries of the analysis have not fully enclosed spatial and temporal dynamics of the environmental resources of concern and the anthropogenic activities influencing them.

To evaluate cumulative effects, the evaluation must be bounded by the spatial and temporal boundaries of the environmental resources of concern. The aggregate influence of multiple disturbances on the total resource must be estimated. That is, the evaluation is done from the perspective of the "valued environmental components" (Beanlands and Duinker 1983), not from the perspective of a particular project or disturbance.

The distinction between conventional and cumulative impact evaluation may best be captured through an analogy (Gosselink and Lee 1988). Imagine a Renaissance mosaic of a mother and child, composed of tiles of various shapes and colors. With age, the mosaic has begun to lose tiles and we must decide which tiles to reinforce to best preserve its value. If conventional environmental assessment strategies were used, the tiles would be evaluated in terms of their individual intrinsic value. Those of highest intrinsic value would be selectively preserved. This strategy would not preserve the image of mother and child. Yet the *image* is the feature making the mosaic more valuable than the sum of the values of its component tiles; the image itself is the resource of concern. If the image in the mosaic is to be preserved, the value of each tile must be determined by its importance in conveying the central image of the mosaic within the spatial boundaries of the mosaic as a whole.

To evaluate the cumulative effects of wetland loss or degradation, the value of component wetlands in a watershed or regional wetland system must be determined, based on their relative contribution to the functioning of the entire landscape system (Forman and Godron 1986). The role of wetlands in the landscape may depend not just on their acreage but on the mosaic of wetland types and specific physical and chemical conditions (Whigham and others 1988). For example, water quality in a watershed may improve as runoff proceeds downstream through a series of different wetland types. Because of the potential for interactions between nutrient loading and mobilization of heavy metals through plant growth effects on sediment redox reactions (Giblin 1982), the elimination of one wetland that is efficient at nutrient retention could reduce the ability of a downstream wetland to remove heavy metals. The overall effect on the quality of water exported from the watershed cannot be predicted solely from knowledge of the effects occurring in any one wetland within the watershed.

Establishing the appropriate spatial boundaries for an evaluation of cumulative effects on wetland system functions is a crucial first step. Without these boundaries, there is no basis for determining which anthropogenic disturbances to include in the cumulative impact analysis and which to exclude. Once the boundaries have been established, all anthropogenic disturbances within them should be included in the cumulative impact analysis.

Establishing boundaries is itself a complicated process. The boundaries must fully enclose the system functions to be evaluated. Boundaries for different functions may differ, and some functions may depend on others. We will discuss procedures for bounding functions of wetland systems in later sections. Forman and Godron (1986) and Urban and others (1987) provide useful discussions of spatial and temporal scales of landscape processes. The ideas they develop are applicable to the problem of bounding cumulative impact evaluations.

The circumstances under which cumulative effects on wetlands are likely to occur must be identified, and methods specific to these circumstances must be developed. Standard environmental assessment methods were designed to evaluate effects of particular projects on individual wetlands (Lonard and Clairain 1986). They can be used directly to estimate cumulative effects only if these effects are related linearly and noninteractively to impacts on the function of individual wetlands. In such cases, individual wetland impacts can be summed to estimate cumulative impact, and the much more difficult task of estimating cumulative impacts resulting from *functional interaction* among wetlands and multiple anthropogenic disturbances can be avoided. Clark (1986) has made a strong case for identifying those cases in which cumulative effects need not be invoked. As he so aptly puts it, "One of the most useful roles for science in environmental impact assessment is . . . to reduce as many apparently cumulative problems as possible to simple cases of single cause and effect."

A Typology of Cumulative Effects

In a significant number of cases, however, nonadditive cumulative effects will occur. How are these effects different from the single cause-and-effect cases? What are the implications for developing a scientific approach to their evaluation?

Beanlands and others (1986) summarized five ways in which effects accumulate and differ from the simple case:

1) Time-crowded perturbations. Disturbances occur sufficiently close in time that the system does not recover in the time between.
2) Space-crowded perturbations. Disturbances overlap in space or occur so close together that their effects are not dissipated in the distance between.
3) Synergisms. The interactions of different types of disturbances produce effects qualitatively and quantitatively different from the individual disturbances.
4) Indirect effects. Disturbances initiate a chain of events that produce effects delayed in time or space from the original disturbance.
5) Nibbling. Disturbances produce effects by small changes, i.e., incremental or decremental effects.

Baskerville (1986) and Clark (1986) elaborate these concepts and explore their implications for designing scientific studies of cumulative effects. Baskerville describes two particular types of indirect effects that illustrate potential problems in research design. First, he lists situations "where a single action or limited intervention results in alteration of the system structure, or system dynamics, such that the system itself accumulates the cause of the impact over time." As an example he cites is the forest management practices in Nova Scotia, which have altered the age structure of the forest by allowing spruce budworms to "harvest" 80% of the forest. Thus, the forest is set up for an insect outbreak many years later when a forest dominated by one age class and one species will come to maturity. He likens this form of accumulation to "the delay between exposure to a carcinogenic agent and the onset of cancer."

This form of accumulation is certainly an indirect effect, but Baskerville's emphasis on the importance of change in system structure or dynamics is worth noting. We can expect analogous cases in wetlands. For example, leaks from the cooling lake of a power plant into a marsh initiated changes in plant community structure, which later left the marsh sediments (and associated nutrients) subject to erosion during annual flooding of an adjacent river (Bedford 1980). The focus of the impact study on the marsh had been on the plant community itself. This and Baskerville's example underscore the fact that without knowledge of system structure and dynamics, an impact study may not be bounded appropriately in terms of time and space, or the variables and indicators chosen for measurement.

Baskerville's second case of indirect effects also pertains to correct choice of temporal and spatial boundaries. He points out that some types of disturbances "accumulate by cycling over geographic time and space." His example is clear-cutting in forests. A certain number of patches are cut each year; these begin the successional process of regrowth, and additional patches are cut in each succeeding year. The total effect on the forest migrates across the landscape with time. Only a study that focused at the regional forest level rather than the individual patch level would capture the accumulating effect of altering the structure and functioning of an entire region's forests. An analogous example for wetlands would be the pattern of timber harvesting in the bottomland hardwoods of the Mississippi-Atchafalaya delta, or peat harvesting from different parts of a wetland complex over a number of years. The conventional focus on effects that are local in time and space would miss this accumulation of effects.

Clark (1986) elaborates several concepts of cumulative effects that address the issue of research design. First, he emphasizes the importance of identifying the characteristics that distinguish cumulative effects from single cause-and-effect cases. He does this by utilizing the null hypothesis; that is, he seeks to "define and assess the conditions necessary for rejecting the null hypothesis" of no cumulative impact. Second, he states the concepts of temporal and spatial distance in terms from which quantitative measures can be derived. Effects are time-crowded, or cumulative in time, and the null hypothesis of no cumulative impact can be questioned if "the time required for the natural system to remove or dissipate a unit of disturbance is of the same order or greater than the time between such disturbances." He gives an analogous definition for effects that accumulate in space. Third, he develops the concept of synoptic assessment, which considers not just a single impact but all sources of disturbance to the valued components of an environmental resource.

This approach sets boundaries for the assessment from the perspective of the resource rather than from a single project or disturbance. Concepts may aid or constrain scientific endeavors (Cook 1981). Elaborations such as those made by Clark (1986) and Baskerville (1986) are central to transforming concepts of cumulative impact into a scientific approach to its assessment.

Impacts vs Effects

Although the standard terminology is *cumulative impacts*, a distinction should be drawn between effects and impacts. The use of the term *impacts* connotes a value judgement. As Erckmann (1986) points out, ". . . in any kind of impact assessment we are trying to predict and detect changes that can be causally linked to some source(s); when we can make the connection, we call the changes effects. When the effects are valued as negative, we call them impacts, because what we deem now as a positive effect may in the future be seen as an impact." In focusing on the scientific and technical component of cumulative impact assessment rather than the social or political, we will, therefore, use the term *effects*.

The Scientific Component of Assessing Cumulative Effects

Our aim in this article is limited relative to the overall process of cumulative impact assessment. In its broadest context, cumulative impact assessment includes an almost overwhelming number of components (Cline and others 1983, Vlachos 1985, Beanlands and others 1986, Lee and Gosselink 1988, Stakhiv 1988). Many of them are not scientific questions; they have to do with socio-economic, legal, jurisdictional, administrative, or policy issues, such as goal-setting. Although they have been discussed extensively (see Cline and others 1983 and references therein), they still present some of the most serious impediments to controlling cumulative impacts (Cline and others 1983, Robilliard 1986, Lee and Gosselink 1988, Stakhiv 1988).

But there are other constraints as well. Robilliard (1986) summarized these noninstitutional constraints as: "procedural, from a legal viewpoint; methodological, from a 'how do we conduct the analysis?' perspective; and technical, from the standpoint of what data/problems/analyses/etc. are available and do we understand how the system(s) work?"

It is only the last of these three categories that we explore here. We will concentrate on identifying issues, data, problems, and analyses that help us understand how systems of interacting wetlands function and respond to multiple disturbances. We recognize that parts of the technical process—identifying valued environmental components, establishing regulatory standards or reference states and baseline conditions, and delineating both geographically and temporally appropriate boundaries—have social and political dimensions, but they also have significant scientific dimensions.

Two related problems are associated with overcoming the difficulties of scientific assessment. One has to do with asking the right questions, and the other with getting the right answers. That is, what are the scientific questions or issues relevant to evaluating cumulative effects on wetland ecosystems? And how do we surmount the difficulties of doing research on cumulative effects? We discuss both problems in the next section.

The Toy/Real Paradigm in Cumulative Effects Research

No clearer statement has been made of the problems associated with research on cumulative effects than Baskerville's (1986) adaptation of the toy/real paradigm from Sprague and Sprague (1976):

> Problems take one of two forms, real and toy. Real problems are those that exist in their real-world context, and their principal characteristics are large size, high spatial and temporal variability, and general uncontrollability with respect to experimentation. The second group of problems are referred to as toy. Toy problems are the caricatures or models that we make of part or all of a real problem. Toy problems are characterized by being simple, small, clearly structured, and well controlled with respect to experimentation. Similarly, they divide research approaches into real and toy categories. Real research is characterized by scientific rigor. This means a high level of control in well bounded situations with explicit measures and test protocols. There is a well-defined experimental approach and application of treatments to controlled subjects under controlled conditions with precise measurements and rigorous statistical analysis. Real research usually addresses cause/effect connections in a simple and direct manner. Toy research is characterized by observation of system states, which are the outcome of cause/effect connections, perhaps not even including measurement. From this, superficial analysis of system function is attempted from the outcomes, and in the absence of experimental control.
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> There is a potent message in the toy/real paradigm with respect to the use of science in environmental impact assessment in general, and in cumulative impact assessment in particular. Clearly, real research on toy problems is absolutely essential. These can be important building blocks for tackling the real problem if the toy problems are well constructed and bounded in the context of the real problem. Toy research on real problems is absolutely essential. This work can constitute integration of scientific understanding over the temporal and spatial bounds of the real problem if the scientific approach is rigorous. The key is to avoid toy research on toy problems, and to be honest about the extremely limited extent to which we can carry out real research on real problems. (Baskerville 1986)

Cumulative effects on wetlands are a real problem. To surmount the difficulties of doing research on cumulative effects, we will need to do both real research on toy problems and toy research on the real problem. The real research on toy problems will provide the pieces or building blocks of the puzzle. The toy research on the real problem is essential to understand system structure and provide an integrated picture of the system at appropriate temporal and spatial scales; it will tell us where and how the pieces fit. If both types of research are not designed with scientific rigor, we will have pieces that do not fit the whole, or no understanding of what pieces are needed and how they fit together in the system. We would have the right answers to the wrong questions, or worse, the wrong questions and the wrong answers.

Baskerville's adaptation of the toy/real paradigm explains that research relevant to cumulative effects need not necessarily look directly at those effects. Research designed to elucidate wetland system structure, and with explicitly stated system relationships, is "crucial in bridging from the toy problems used for real research, to rigorous scientific analysis at real problem level" (Baskerville 1986). The real challenge lies in correctly identifying the pieces, the structure into which they fit, and measures appropriate to the temporal and spatial scales of concern in evaluating cumulative effects on specific wetland functions.

Scientific Issues in Evaluating Cumulative Effects on Wetland Systems

The scientific issues relevant to evaluating cumulative effects on an interacting complex of wetland ecosystems have both generic and specific forms. The latter apply to: (1) particular types of landscape units,

(2) particular functions that wetlands perform within these units, and (3) particular anthropogenic disturbances that alter the level of performance of function. In this section we identify the generic issues and offer some specific examples, in the hope that others will provide more.

Definition of Wetland Functional Values

For illustration, we will focus on a small subset of the ecosystem functions that wetlands perform in a landscape, representing the functions most readily connected to some human value. Our use of the term *function* does not imply evolved or highly integrated purpose, but rather the ecosystem properties that derive from the spatially structured interactions among many processes, and the biological and physical–chemical components within that system. We provide the following definition of these wetland functions; however, a sound research program must, as a matter of critical importance, imbue these definitions with more scientific precision.

Hydrologic function: the capacity of wetlands to reduce and desynchronize peak flood discharge, influence base flow, and modify groundwater interactions with surface water.

Water quality function: the capacity of wetlands to remove or transform excess nutrients, organic compounds, trace metals, sediment, and refractory chemicals from water as it moves downstream.

Life support function: the capacity of wetlands to supply the requirements (qualitatively and quantitatively) of the biota normally using a wetland system. These requirements include food, shelter, the appropriate hydrologic regime, and other system attributes critical to maintaining populations and communities. Measures of life support function must integrate many wetland attributes, including water quality, hydroperiod and hydrodynamics, habitat structure, biogeographical setting, nutrient status, corridors for migration, and primary production. As defined here, the life support function includes both *habitat* and *food chain support* as defined by Adamus and Stockwell (1983). (See also Sather and Stuber (1984) for discussions that highlight limitations of these terms as currently used in assessment.)

These wetland functions can be viewed as a hierarchy of complexity and interdependency (Gosselink and Turner 1978). Understanding the water budget of wetland systems is fundamental to understanding other functions (Winter 1988). The water budget controls wetland formation and maintenance. Water is the primary medium that carries chemicals into wetland systems, where their reaction and sedimentation influence water quality, both within the system and in discharges. Water budget, water quality, and other properties of the hydrologic regime determine in part the suitability of the environment for wetland biota. Properties of the biota, such as primary production, transpiration, and nutrient uptake, in turn influence hydrologic regime and water quality.

For the ecosystem functions that we have just defined, the following generic scientific issues must be addressed: (1) scale, (2) thresholds, and (3) size, shape, and position in landscape.

Scale

Attention to cumulative effects is essentially recognition of processes operating at different, but related, spatial and temporal scales in the landscape. We must shift our perception to the appropriate scales at which to view disturbance to ecosystems. Local disturbances to wetlands are now seen to be related to changes in the distributions and exchanges of major elements and pollutants, and to loss of biotic and habitat diversity at the scale of regional landscapes. In fact, disturbances, both natural and anthropogenic, operate at many scales (Urban and others 1987), as do the ecological processes controlling wetland functions and wetland responses to disturbance. Our ability to predict system response to cumulative disturbance is inextricably tied to our ability to understand how processes and system attributes differ according to scale, and how they relate across scales.

Defining a reference state for system properties and setting boundaries are two issues central to evaluating cumulative effects that require attention to scale. Variability in system properties to be managed or regulated depends upon the scale at which we view or measure these properties. In general, the finer the spatial scale, the more the property may be seen to vary with time. The number of waterfowl breeding in a single prairie pothole may vary more from year to year than the number breeding in a regional system of potholes. Daily nutrient concentrations of water discharged from a wetland will show more variability than yearly averages. Over short periods of time, wetlands may sequester certain compounds, but over longer periods of time these might show a net release to downstream waters because of seasonal or annual fluctuations in sediment redox status. We need to choose the temporal and spatial scales at which we wish to set standards.

Only after the temporal and spatial scale for assessment is determined can the choice of internal variables to measure or monitor be made. The types of variables or indicators will change, but not necessarily increase, as scale increases (Baskerville 1986). The use of distributions (Levin and Paine 1974, Paine and Levin 1981) or ratios (White and Pickett 1985) considered at the watershed or regional scale, rather than absolute measures considered on an individual wetland basis, may offer a means of dealing with local variability vs broader scale equilibrium. For example, we could use size class distributions of wetlands in a region, or distributions of type, or distributions of functional classes of wetlands as a measure of the *reference state* of the watershed or regional system. Or we might use ratios of the size of proposed disturbance to total wetland acreage, or to particular type of wetland, or to territory size of important species.

The variation with scale for many wetland processes is unknown. Vitousek (1985), noting that patches the size of a single tree gap alter stand dynamics in temperate and tropical forests, has asked how such small disturbances affect nutrient dynamics. Do we know the critical size below which a patch disturbance such as filling does not alter wetland nutrient dynamics? What are the appropriate temporal scales for evaluating the sequestration of heavy metals by wetland sediments?

As a further question of scale, the spatial and temporal boundaries for different wetland functions are not likely to be congruous with one another. They also may differ within functions. That is, the boundaries of study for water quality will differ from those for life support function, and the boundaries for different species of waterfowl or for different pollutants will not be the same. Not all waterfowl have the same migration patterns or feeding behaviors, and not all elements behave the same way. In fact, as O'Neill and others (1986) have pointed out, "The ecosystem appears to have distinctly different structural properties when examined for different elements." Ecologists no longer view ecosystems as a series of Chinese boxes (Reiners 1987), with all their properties defined by a common boundary set by obvious physiognomic properties, such as the edge of a lake or stand of trees. Root (1973) clearly stated this in his distinction between compound and component communities. Nonetheless, the significance in defining appropriate boundaries to study functional relationships and tight interactions among elements as well as species has only recently received sufficient emphasis (Bolin and Cook 1983, O'Neill and others 1986). Failure to work on appropriately delimited systems continues to be a frequent problem.

Thresholds

Understanding threshold responses offers the potential to manage some types of cumulative effects. Unfortunately, little is known about either quantitative or qualitative thresholds. A quantitative threshold exists where further addition of a substance or more of the same kind of disturbance eliminates the valued wetland function. Hemond and Benoit (1988) offer the example of some pollutants, such as phosphorus and heavy metals, which accumulate in wetland sediments and ultimately saturate the capacity of the sediments to operate as effective net sinks. A qualitative threshold exists where the nature of the response to more of the same source of disturbance shifts after some critical loading limit. Continued sediment loading eventually will initiate changes in other water quality functions of a wetland by altering the physical, chemical, and biological characteristics of the wetland sediments (Hemond and Benoit 1988). Other examples of threshold issues include the number of individuals that can be removed from a population without initiating population decline, and the proportion of total wetland acreage that can be lost without diminishing flood control in the watershed (Novitski 1978).

Size, Shape, and Position in the Landscape

The size, shape, and position of wetlands in the landscape are infrequently measured, yet are often altered radically through the cumulative consequences of human activities. Sufficient evidence exists from studies of other ecosystems, and from some wetland studies, to establish that these properties strongly influence the functioning of wetlands. Knowledge of how such landscape-level properties relate to specific wetland functions could guide decisions regarding permits to fill wetlands, mitigation requirements, and the designation of "advanced identification" areas (see Hirsch 1988, Stakhiv 1988).

Size, shape, and configuration of habitats or ecosystem types within a landscape or region have been considered explicitly and most frequently in regard to supporting populations of particular species or maximizing the total number of different species in a given geographic area. Usually they are considered under the topics of habitat fragmentation and patch dynamics (Curtis 1956, Burgess and Sharpe 1981, Simberloff and Abele 1982, Harris 1984, 1988, Soulé and Simberloff 1986, Weller 1988, Lee and Gosselink 1988; see also articles in Soulé and Wilcox 1980, Pickett and White 1985, Soulé 1986, and Verner and others 1986). Although some aspects of the issue are still debated, there is little doubt that, as with other types of ecosystems and landscapes, such spatial properties strongly affect the floral and faunal diversity of wetland landscapes. (See Bedford and Preston 1988, for further discussion.)

Size, shape, and landscape position also exert strong effects on hydrologic relationships and water quality in wetland landscapes (Winter 1988, Siegel

1988, O'Brien 1988, Brinson 1988, Whigham and others 1988). At the most fundamental level, it is the spatial distribution of landforms and geological materials that, along with climate, determine where and what types of wetlands develop in a given region (Moore and Bellamy 1974, Damman 1979). One can expect then that wetland functional characteristics will change at some level as the spatial characteristics of the landscape are rearranged.

Considerations particularly relevant to applying the concepts of landscape spatial structure to wetland cumulative impact assessment include:

1) Regional differences. The undisturbed and current spatial distribution of wetlands in a landscape, and the factors controlling those distributions, differ significantly among regions (O'Brien 1988). Areas such as the Mississippi-Atchafalaya delta formerly had large areas of contiguous bottomland hardwood forest, whereas wetlands of the intermountain West and prairie pothole region of the Upper Midwest and adjacent Canadian provinces have always contained smaller wetlands more widely dispersed among upland habitats. Changes in a region need to be evaluated against that region's historical patterns.
2) Size vs volume. Area alone is an inadequate measure for wetland evaluations. Wetlands are three-dimensional systems. Some are shallow with thin organic layers; others occur in relatively deep basins, or contain deep layers of peat. Many processes affecting water quality, such as phosphate absorption, will be a function of volume, not area. A wetland's capacity to store flood waters is obviously a three-dimensional property.
3) Input/output relations. Changes in the spatial configuration of wetlands and adjacent uplands will alter input/output relations of water and other materials, including pollutants, within the wetlands. Resultant changes could include alterations in element cycling or pollutant transformations.
4) Hydrologic pathways. Changes in the spatial configuration of wetlands and adjacent uplands will alter existing patterns of water movement within the landscape. Concommitant changes can be expected in the movement of sediments, nutrients, and pollutants.
5) Dispersal. Water is one of the primary means of dispersal for species, nutrients, and pollutants. Changes in the size, shape, and spatial configuration of wetlands in a landscape necessarily will alter patterns of dispersal.
6) Annual and seasonal fluctuations. Most wetlands experience both annual and seasonal changes in water level, and, hence, in the availability of habitat. The total area of wetland within a landscape or region must be large enough to provide habitat under the extremes of these fluctuations.

A Generic Framework for Evaluating Cumulative Effects on Watershed and Regional Wetland Function

To stimulate ideas and to expand thinking about environmental effects into larger temporal and spatial scales, we have constructed a hypothetical and generic framework for evaluating cumulative effects on three wetland ecosystem functions: flood storage, water quality, and life support. The spatial scales of concern are the watershed and regional landscape. Our reasoning and the conceptual basis for the framework are as follows:

The contribution of a particular wetland to landscape function within watersheds or regions will be determined by its intrinsic characteristics, including size, morphometry, type, percent organic matter in the sediments, and hydrologic regime, and by extrinsic factors, that is, the wetland's context in the landscape mosaic. Any cumulative effects evaluation must take into account the relationship between intrinsic and extrinsic attributes and overall landscape function. Each component wetland can then be assigned a relative value that reflects its contribution to maintaining each system function. Anticipated effects on a particular wetland, as for a permit decision, could then be evaluated in light of that wetland's total contribution.

Furthermore, the relationship between intrinsic and extrinsic wetland attributes and landscape function could serve as the basis for a functional classification of wetlands. Wetlands with similar kinds and levels of landscape function, and those that respond similarly to disturbance, could be grouped into classes. Such functional classes could provide a basis for predesignating certain wetland classes for protection because of their importance to landscape function or their sensitivity to disturbance.

The framework must be based on explicitly defined spatial and temporal boundaries, measurable variables, and specified relationships. These must be developed systematically for each function to be included in a cumulative effects evaluation. Figure 1 illustrates the sequence of activities necessary to structure and then conduct a cumulative effects evaluation. This sequence is described in detail in the next two sections.

Structuring the Evaluation

Establishing spatial and temporal boundaries. The crucial first step is to develop criteria and procedures for

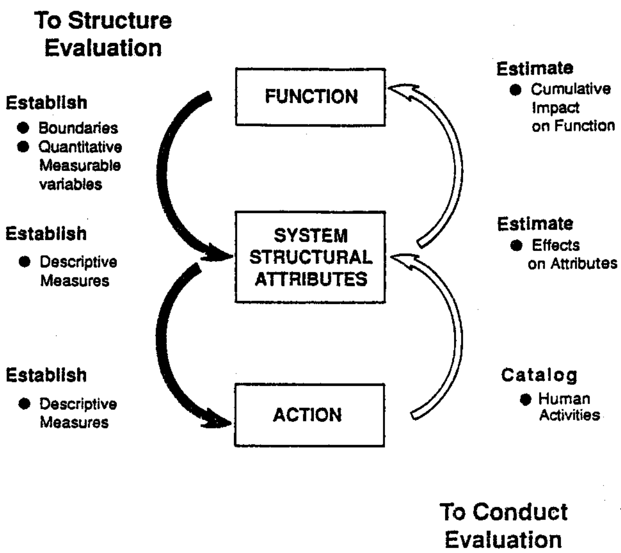

Figure 1. The logic sequence used in structuring and then conducting a cumulative effects evaluation. The arrows represent the relationships to be defined (between functions and attributes and between human activities and attributes). In all cases the head of the arrow points to dependent variables, and the arrow comes from the independent variables. The sequence should be read counterclockwise, beginning from top left.

bounding a particular cumulative effects evaluation in space (the impact area) and in time (the impact period of duration). The resulting boundaries have a profound influence on all aspects of a cumulative effects evaluation:

1) They form the basis for deciding which anthropogenic disturbances to include in the evaluation.
2) The spatial and temporal scales chosen influence the choice of measures of system function and effects (Urban and others 1987).
3) The credibility of the results depends upon assurance that the system processes most important in determining level of system function are subsumed by the boundaries.

Measurable variables. Once spatial and temporal scales have been chosen, appropriate variables must be selected for the evaluation. These should include:

1) Measures of function. Specific measures to quantify level of function for each function of concern. Change in level of function is the measure of cumulative effect in the analysis, the dependent variable.
2) Descriptive measures of the wetland structural attributes that determine the level of system function.
3) Descriptive measures of the proposed human activities.

Relationships between variables. Next, the following relationships between these variables must be defined:

1) Function–attribute relationship. The relationship between wetland functioning and both intrinsic and extrinsic wetland attributes. This may be qualitative or quantitative, and relates change in function of the system of wetlands to change in landscape-level characteristics of the system.
2) Attribute–action relationship. The relationship between change in specific human activities (e.g., dredging) and change in the system attributes previously defined.

Conducting the Evaluation

If the function–attribute and attribute–action relationships are known within the spatial/temporal boundaries of the analysis, the cumulative effect of changes in the level of human activities on system function can be evaluated as follows (Figure 1):

1) Catalog the relevant measures of human activities within the evaluation area.
2) Estimate their effects on system attributes with the attribute–action relationship.
3) Estimate the change in system function from changes in system attributes using the function–attribute relationship.

Bounding the Evaluation in Space

We use the magnitude of exchanges among component wetlands in a watershed or ecoregion as the basis for defining the geographic boundaries of the study. If the total accumulation of effects is different from the sum of effects in individual wetlands, interactions of some kind must occur between wetlands, the result of exchanges of energy or materials. The magnitude of these exchanges characterizes the degree of functional interdependence between wetlands, as well as the potential for cumulative effects. In defining boundaries for an impact area, therefore, the magnitude of exchanges among component wetlands should be a primary consideration. Exchanges of energy or materials within the impact area should be greater, by some degree, than such interactions with wetlands outside the impact area.

The choice of medium of exchange to be measured will depend on the particular functional value being considered. Individuals and propagules of plant and animal species are the relevant medium for evaluating life support functions. In evaluating flood storage and other aspects of hydrologic function, we need to measure water exchanges. Pollutants are the relevant substances of exchange for water quality evaluations.

We define the impact area for each functional value as the area within which the relevant exchange of energy or materials among wetlands and associated uplands takes place. Individual wetlands are the basic geographic building blocks, the subdivisions within the impact area whose contributions to disturbance effects we seek to interpret. The logical aggregation units for evaluation differ among functions. Because hydrologic processes are watershed-level phenomena (Winter 1988), the watershed is the appropriate aggregation unit for both flood storage and water quality functions. Life support functions, however, operate on scales defined by intersection of the ranges of wetland species. We suggest that ranges of valued species and ecoregion boundaries (Omernik 1987) may serve as logical geographic units for evaluating impacts at the landscape level.

Bounding the Evaluation in Time

We base our determination of temporal boundaries on the time scales of recovery for processes controlling particular wetland functions. If disturbances are infrequent enough so that the system has time to recover between disturbances, no accumulation of effects can occur. When the proposed activity will occur before the system has recovered from a previous disturbance, or if another disturbance can be expected to occur at some time in the future before the system recovers from the proposed activity, then impact period or duration must include the previous and/or future disturbances.

Although the general concept of using time scales of recovery to delineate temporal boundaries is fairly straightforward, the specific requirements for each function are extensive and challenge current understanding. In order to establish system recovery rates for each function, the following factors must be known or determined:

1) The key physical, chemical, and biological processes that regulate the relevant function.
2) The influence of wetland attributes (both intrinsic and extrinsic factors) on process rates.
3) Recovery rates of the different relevant processes for wetlands with given attributes.
4) The types of interactions, especially key process interactions and tight connections, e.g., *component communities* (Root 1973), that affect the function and its rate of recovery.

These may differ in the face of different disturbances.

Although our knowledge is far from complete, a good deal is known about which processes control system function. The processes themselves tend not to be location-specific, and knowledge gained in a few systems can thus be extrapolated to others. Unfortunately, this is not true for rates. Location-specific factors govern process rates. The impact period or duration, therefore, must be bounded by recovery time scales for the relevant function within the particular wetland system. Location-specific factors that are most likely to influence rates must be identified through additional research and critical synthesis of existing data before assessments can be made with only minimal site-specific data.

Evaluating Cumulative Effects on Flood Storage Function of Wetlands

The purpose of the evaluation is to estimate the effects of cumulative loss of wetland flood storage capacity on peak water depth at a point or series of points during flood events. In this case, functional value can be quantified by peak water depth. The relevant wetland attributes are those affecting storage capacity in the impact area. Impact can be quantified by the effect of disturbance on these attributes. A mass balance analysis of water within the impact area provides the conceptual framework for the evaluation. The evaluation could be accomplished by:

1) First, defining the points where peak water depth is to be measured; that is, the flood damage evaluation points.
2) Next, defining the impact area to include all significant water inputs affecting peak water depth at these points that are subject to modification by wetlands; that is, the hydrologic unit of interest.
3) Finally, evaluating the effects of modification of wetland characteristics (size, location in the impact area, location relative to the downstream flood damage evaluation points) on peak flows for floods of different frequencies.

Each of these steps is elaborated below.

Defining the Flood Damage Evaluation Points

Evaluation points may be defined by either the location of a valued resource (such as a city) on the floodplain or as the point of discharge from one hydrologic unit to another. The choice must be made with the ultimate goal of the evaluation in mind. Evaluation of peak flood to a valued resource is appropriate if the resource is the sole target of the evaluation. However, if one wishes to develop a hierarchical evaluation system in which discharges from small hydrologic units can be aggregated to estimate larger scale effects, then natural hydrologic unit discharge points are the appropriate choice.

Defining the Hydrologic Unit of Interest

The spatial boundaries of the evaluation should be chosen so that the evaluation area includes: (1) all significant sources of water contributing to flood events at the chosen flood damage evaluation point, and (2) all wetlands in the hydrologic path connecting the sources to the evaluation point. To do this, the term *significant source* must be operationally defined and an analysis of the water budget for the evaluation point must be conducted. The choice of evaluation point has profound effects on the spatial scale of the evaluation (Figure 2). Relevant spatial scales include watershed, regional drainage, and continental drainage.

Bounding the Evaluation in Time

Since peak water depth during flood events is of primary interest, the temporal scale chosen must be appropriate to capture changes in important variables affecting flood peak. Daily or shorter intervals are appropriate. To evaluate events of different frequencies and magnitudes, the temporal patterns of important variables on longer time scales (seasonal, annual, 10 years, 20 years, 100 years) should be evaluated.

Describing the Temporal Dynamics of the Water Budget

Most floods are generated by snow melt or precipitation. Their size depends on the drainage basin size, antecedent moisture conditions, the quantity of water in the snow pack, the rates of rainfall and snowmelt, the infiltration characteristics of basin soils, and the storage capacity of wetlands and lakes in the drainage basin (Carter and others 1978). The volume (V) of water in a hydrologic unit is the product of its surface area (A) and depth (D). Volume can also be quantified as input volume (I) plus storage volume (S) minus output volume (O). Therefore

$$V = A \times D = I + S - O$$
$$D = \frac{I + S - O}{A}$$

The change in water depth with time (dD/dt) equals dV/dt divided by dA/dt, so

$$\frac{dD}{dt} = \frac{\frac{dI}{dt} + \frac{dS}{dt} - \frac{dO}{dt}}{\frac{dA}{dT}}$$

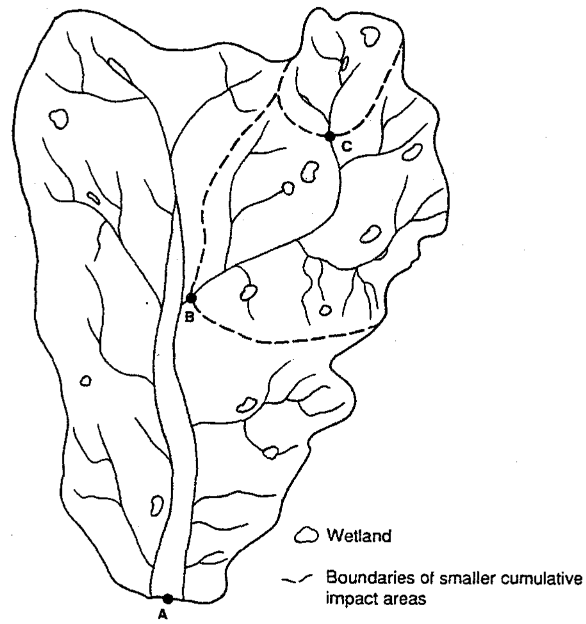

Figure 2. A hypothetical watershed showing possible impact areas for evaluating cumulative effects on the flood storage function of wetlands. Points A, B, and C represent possible flood damage evaluation points. Note the consequences for spatial scale depending upon which point is chosen for evaluation.

It follows that peak water depths during flood events are the maxima of this function. This simple model illustrates that a quantitative evaluation will require information on water input, output, and storage rates.

Evaluating the Effects of Wetland Modification on Flood Peak

Wetlands in a drainage basin attenuate flood peaks and storm flows primarily by temporarily storing surface water. Many factors contribute to this function (Ogawa and Male 1986): (1) size of the wetland, (2) its location relative to the drainage as a whole, (3) its location relative to the downstream flood damage evaluation point, and (4) its hydrologic class (Novitski 1978). The evaluation should define the relationship between these wetland characteristics and reduced flood peak at the evaluation point.

Ogawa and Male (1986) have used watershed hydrologic simulation experiments to approximate such an evaluation for three watersheds in eastern Massachusetts. They included the above factors 1, 2, and 3, but did not evaluate importance of wetland hydrologic class. The value of a wetland for reducing downstream flooding was measured by

$$\%dQ_p = \frac{Q_e - Q_p}{Q_p} \times 100$$

where Q_e and Q_p are the peak flows at a downstream location, generated with and without encroachment on upstream wetland storage, respectively. Q_e changed with the degree of wetland encroachment.

Simulation experiments were run for three rainfall intensities (10 year, 100 year, and 500 year maxima), three antecedent moisture conditions, and four wetland encroachment scenarios. The results suggested that within these watersheds, wetland encroachment of up to 25% would not affect peak flows significantly except for encroachment on mainstem wetlands. Encroachment on upstream tributary wetlands generally alters the peak flows significantly only for a few miles downstream. Thus, unless the area to be protected from flooding is located closer than several miles downstream of a wetland, the functional value of the wetland for flood mitigation is negligible. Encroachment on downstream mainstem wetlands, in contrast, has a great influence further downstream (Ogawa and Male 1986). Further research should examine the influence of incorporating hydrologic class in the model.

Ranking Wetland Value

Simulation experiments similar to those conducted by Ogawa and Male (1986) could be used to evaluate the effects of wetland loss on a case-by-case basis. However, the results would be site- and disturbance-specific. In addition, the effort and resources required to do this are likely to preclude routine use in a regulatory context.

A more generic approach might involve creating a wetland classification scheme relating wetland attributes to peak flow characteristics for different regions. Two philosophically different approaches are possible.

Bottom-up. This approach uses a series of representative case studies similar to that of Ogawa and Male (1986). Simulation experiments executed for a representative sample of watersheds within a hydrologic unit could be used to estimate the value of different wetland classes (based on location, size, and hydrologic class) in reducing peak flood flows. For each region, the typical value (or range of values) of wetlands in each class could be determined, and the results extrapolated to larger spatial scales.

Top-down. This approach involves evaluating spatial overlays of hydrologic unit maps, ecoregion maps, and wetland inventory maps. Hydrologic unit maps are used to delimit the cumulative effects evaluation area. Ecoregion intersections within the hydrologic unit are used to delimit wetlands within relatively homogeneous landscape units. Landscape-level measures of wetland attributes and wetland classifications on the wetland inventory maps are used to assign wetlands to functional classes. A few case studies in hydrologic units in contrasting ecoregions would be used to determine the best landscape-level measures of wetland attributes to use in different regions. This approach is suitable for relatively rapid synoptic evaluations of potential cumulative impacts of wetland loss within hydrologic units. The ease of analysis bears a cost, however. On the whole, cumulative effects evaluations within a region would be generally accurate, but estimates of effects at specific sites would be imprecise. If greater precision were required, a quantitative case-specific analysis could be conducted.

Evaluating Cumulative Effects on the Water Quality Function of Wetlands

Wetlands affect water quality through element cycling, sediment deposition, ion and molecule adsorption, and temperature modification. The quality of water leaving the wetland may differ substantially from that which enters. Dissolved materials (nutrients, trace metals) may be retained or transformed and sediments may settle out (Carter 1986, Nixon and Lee 1986, Hemond and Benoit 1988).

The loading capacity of a wetland or system of wetlands is the quantity of a pollutant (e.g., nutrient, sediment, trace metal, synthetic organic) that the system can retain or transform per unit time. It is markedly influenced by hydrologic regime. The source, velocity, and seasonal distribution of water in the system directly controls the spatial heterogeneity and development of wetlands and the nutrient, oxygen, and pollutant load of the sediments. As demonstrated by Richardson (1986) and discussed by Hemond and Benoit (1988) and Brinson (1988), loading capacity is a temporally dynamic characteristic. It is likely to change seasonally and during the system's successional evolution. In addition, soils, biota, and system size are important determinants of loading capacity.

The biogeochemistry of wetlands can be divided into (1) intrasystem cycling through various transformation processes, and (2) the exchange of chemicals between the system and its surroundings. Intrasystem cycling, along with hydrologic conditions, influences the degree to which chemicals are transported to or from wetlands. Wetlands can be relatively open or relatively closed, depending on the degree of exchange (Mitsch and Gosselink 1986, Brinson 1988). For the present discussion we are concerned with biogeochemically open systems that interact through exchange of water.

Overview of the Problem

The purpose of the water quality evaluation is to estimate the effects of cumulative loss of wetland pollutant loading capacity on pollutant concentration at a particular point or series of points in the system. This evaluation is inherently more complex than the flood storage evaluation. In addition to hydrology, multiple independent variables (target pollutants), multiple dependent variables (pollutant loading capacities), complex chemistry, and pollutant toxicity must be considered. The time scale for the evaluation must be continuous rather than event-based.

Wetlands within an evaluation area can be viewed as a series of flow-through reaction vessels that receive reactants from hydrologic and other sources and export transformed reaction products downstream at specific concentrations and rates. Each wetland in the system provides a catalytic surface for physical and chemical reactions and a settling surface for sediments. Export of pollutants from the wetland is a function of the concentration of reactants entering, residence time (flow rate/volume), and loading capacity of the wetland. Export to the evaluation point depends on characteristics of wetlands in the flow sequence and their degree of excess loading capacity.

Up to this point, the discussion has ignored the potential effect on loading capacity of pollutants entering a wetland. When pollutants produce a toxic effect on the biota, the biota responds both physiologically and evolutionarily (through natural selection). This response may well affect the loading capacity of the system, by increasing or decreasing it, and the critical load limit of the system must be considered. The critical load limit is reached when the pollutant load (all pollutant inputs) initiates a response in the biota that reduces loading capacity, or causes a change in community structure greater than normal temporal variations. A desirable goal might be to manage wetland systems so that the loading capacity of the system, and the critical load limit for any component wetland, is not exceeded.

For a given component wetland and pollutant, both the loading capacity and the critical load limit may vary seasonally, from year to year, and/or with pollutant mixture. To preserve the biological integrity of the wetland, pollutant concentrations should be kept below the critical load limit for the wetland. Any pollutant excess above the wetland's loading capacity will be passed to the next wetland in the system. To prevent degradation of water quality at the evaluation point, the loading capacity of the system must not be exceeded.

Bounding the Evaluation in Space

Spatial scale should be chosen to include the major reactant inputs and hydrologic flows to the evaluation point. This will probably result in essentially the same boundaries previously described under the heading: "Defining the Hydrologic Unit of Interest."

Bounding the Evaluation in Time

Time scale may need to be no longer than residence times of water in component wetlands within the wetland system. However, the evaluation must be able to account for seasonal and year-to-year variation in important variables in the budget of the pollutant under study. Thus, the evaluation as a whole should be bounded by the residence time of pollutants within the wetland system in the evaluation area. Pollutant residence time will vary as a function of type(s) of pollutant(s), and of differences among wetlands in rates of relevant biogeochemical processes and physical mass transport (Nixon and Lee 1986, Hemond and others 1987).

Cumulative Effects Evaluation

The evaluation must estimate the effect of wetland loss or degradation on function as a result of pollutant contamination. Effect is measured as change in excess system loading capacity from point of pollutant input to evaluation point. Level of function of the wetland system is measured as pollutant concentration at the evaluation point (peak or average).

The pollutant budget of the wetland system provides the conceptual framework for the evaluation. To the extent possible, fundamental understanding of the kinetics of chemical, biological, and physical processes that govern the transport and transformation of materials in the system should be synthesized into a description of the pollutant budget. These kinetics are functions of environmental driving forces, including both internally controlled system resources (e.g., dissolved oxygen and pH) and the externally imposed system environment (e.g., site geology, hydrology, and regional climate).

There is significant literature on modeling wetlands (Mitsch and others 1982, Mitsch 1983). The Hydrologic Simulation Program (Johanson and others 1984) is specifically designed to assess water quality over an entire watershed. This model may provide a suitable framework for incorporating wetland modules, allowing evaluation of the water quality consequences of progressive degradation of basin wetlands (Burns 1986).

The usefulness of these models for prediction is under debate. Wetland systems exhibit great variability in their ability to process pollutants, and current knowledge cannot adequately account for this variability (Nixon and Lee 1986, Hemond and Benoit 1988). Most available models do not specifically account for changes in the biota stimulated by pollutant toxicity; limitations of current knowledge in this area may be a significant obstacle to incorporation of such effects.

The evaluation, however, is conceptually tractable. Either the bottom-up or the top-down approaches could be applied, depending on the degree of quantification required (see the section titled: "Ranking Wetland Value") and the availability of models and appropriate data.

A model could be structured to estimate loading capacity for pollutants given a set of values for driving variables (determined by geographic setting) and state variables (determined by wetland type, size, etc.). Simulation experiments could be run to determine the range in estimated loading capacities for pollutants of concern for wetlands of various types and sizes in different regions. For each pollutant, wetland response would be classified based on location in the flow sequence, wetland type, size, and excess loading capacity. Experiments such as these executed for a representative sample of watersheds within a hydrologic unit could be used to estimate the influence of wetland classes on changing pollutant concentrations at the evaluation point.

Conceptual models such as the one herein described can be developed on the basis of present understanding of which wetland processes affect water quality (Nixon and Lee 1986, Hemond and Benoit 1988). These models will help to structure qualitative assessments and direct future research by identifying major hypotheses and data requirements. Development of quantitative assessment methods, however, awaits improved understanding of factors that control the rates of these processes and their interactions and thereby account for differences among wetlands. Whigham and others (1988) and Brinson (1988) present cogent arguments that in the interim the top-down, landscape-level approach is likely to prove more tractable.

Evaluating Cumulative Effects on the Life Support Function of Wetlands

Life support is the most complex wetland function to be evaluated. The regulatory goal is preservation of native biotic diversity at all levels in the hierarchy of ecological structure. The Clean Water Act refers to this diversity as maintenance of "balanced indigenous populations" and "balanced biological communities."

Though considerable progress has been made in recent years in understanding the factors determining biotic diversity, the quantitative relationships between these factors and the biotic diversity of wetland landscapes have not been established. At present, such relationships must be inferred from knowledge and principles derived from other ecological systems (Soulé and Wilcox 1980, Rudis and Ek 1981, Simberloff and Abele 1982, Soulé and Simberloff 1986, Wilson 1988). The following discussion draws heavily on those principles, as well as principles from landscape ecology (Forman and Godron 1986).

The life support functioning of a wetland system is dependent not only on hydrology and water quality, but on biogeographical setting. The regional fauna and flora, climate, and the landscape mosaic determine the potential biotic diversity of the system (Ricklefs 1987). Within this framework, management choices will determine the degree to which the potential is realized. Society has expressed at least two goals in this regard: maintenance of populations of particular valued species, and of biological integrity.

Certain species are regarded by society as particularly valuable and worthy of preservation. Some are of direct commercial value, others are chosen because they are rare or endangered, still others are valued because of their contribution to human recreation or aesthetics, and some species are recognized for their critical roles in regulating the structure or function of ecological communities (keystone species). Determining effects on the biotic diversity within populations of these valued species must be one target of cumulative impact evaluations when the objective is simply preservation of a viable natural population. If the objective is preserving a sustainable yield of a resource species, environmental carrying capacity or habitat-related measures may be the appropriate dependent variable.

Another goal of these evaluations should be determining effects on the biological integrity of wetland systems. Biological integrity refers to the plant and animal species that are characteristic of a region and their relative abundances in the absence of human intervention (Karr and others 1986).

Life support functions operate on scales defined by the complex temporal and spatial intersection of ranges of wetland species. Bounding biotic communities is, therefore, particularly problematic. Ecoregion maps (Bailey 1976, Omernik 1987) attempt to bound ecologically homogeneous regions. Relatively distinct ecological systems tend to occur in spatial units with relatively homogeneous landscape characteristics (i.e., climate, landform, soils, potential natural vegetation, land use). Studies by Inkley and Anderson (1982) and Larsen and others (1986) demonstrate correspondence between wildlife communities and fish assemblages, respectively, and ecoregions. Species functional interactions are likely to be more intense within these spatial units than between them. We propose, therefore, to use watersheds aggregated within ecoregions to delimit the impact area for an evaluation of life support function on aquatic communities.

Migratory and wide-ranging species present special problems (Myers and others 1987), since they may be seasonally important in different ecoregions, or may not be greatly influenced by the landscape characteristics used to delimit ecoregions. For these species, species' range may provide the primary boundary, while the intersection of species range with ecoregion boundaries may provide appropriate spatial subunits for analysis.

Biotic Diversity as a Dependent Variable

If biotic diversity is to be the dependent variable in a cumulative effects evaluation, appropriate biotic diversity measurement scales must be developed for both valued species and biological integrity. Weller (1988) and Harris (1984, 1988) have discussed factors influencing this diversity and suggested a number of indicators.

Valued species. Two primary factors contribute to within-species diversity. Within a freely interbreeding population, the degree of genetic variability is determined by (1) the number of polymorphic loci, (2) the number and type of alleles at these loci, and (3) the average level of heterozygosity. This genetic variability allows the population to respond to changing environmental selective pressures and is important for continued evolutionary adaptation to a changing environment. A second source of biotic diversity derives from separation of the population into local subpopulations that do not interbreed freely. Local selective pressures result in different gene frequencies among subpopulations and may favor different gene mutations, leading to broader genetic variability in the population as a whole.

One endpoint of a biotic diversity scale might be the minimum heterozygosity that will conserve sufficient genetic variability to permit continued evolutionary adaption to changing selective pressures. Harris (1984) suggests approaches for estimating this. The other endpoint might be the characteristic diversity of the population in the absence of human disturbance. This endpoint could be determined by sampling representative wild populations. Neither of these measures, however, is without problems, and the topic of appropriate genetic and demographic measures for conservation of species has engendered considerable discussion (Vrijenhoek 1985; see also Schonewald-Cox and others 1983, Soulé 1986, articles and references therein).

Biological integrity. Several changes occur in biota in response to stress (Mooney and Godron 1983, Levin and Kimball 1984, Karr and Freemark 1985, Harris and Gosselink 1986). Karr and Freemark (1985) provide a clear discussion of the complexity of responses to disturbance in vertebrate communities, including the Everglades and marshes of the prairie pothole region as examples. Stress-induced changes may include loss of higher trophic levels, leading to shortened foodchains and loss of the habitat specialists that create faunal and floral identity for an ecosystem or landscape. These changes result in a truncated biotic assemblage heavy with generalists. Any measure or scale of ecological integrity thus should be sensitive to changes in both the composition and structure of ecological communities. Karr and others (1986) have developed a multiparameter Index of Biological Integrity (IBI) for fish communities that embodies these principles. It uses 12 different ecological attributes to summarize the status of a community in terms of species richness, trophic composition, and species abundance and condition. Perhaps a similar procedure can be used to develop an appropriate index of ecological integrity for wetland species assemblages for different wetland ecoregions (Karr 1987). It's application, however, must account for the seasonal, year-to-year, and longer-term cycles characteristic of different wetlands.

Impact as the Independent Variable

The impact of wetland loss or degradation must be quantified in relation to the factors in the landscape mosaic that influence biotic diversity. Harris (1984) suggests that the primary factors are total habitat area, the size–frequency distribution and quality of habitat patches, and the distribution of these patches in relation to each other and to drainage patterns in the landscape. Valued species are dependent on the wetland component of the landscape to varying degrees (Harris 1988). To the degree that other habitat types are also important, the distributional characteristics of wetland habitat must be considered in relation to the distribution of patches of other important habitat types.

A measure of impact must reflect the change in habitat distribution resulting from human disturbance with respect to a reference standard. A standard could be developed empirically from current and historical data on the size and distributional characteristics of habitat within the species ranges and ecoregions subject to cumulative effects evaluation. Development of the standard would need to take into account the relatively short timespan of historical data and natural fluctuations in wetland size and distribution.

Evaluation of Cumulative Effects on Life Support

Evaluations of cumulative effects on the life support functioning of wetlands are likely to be qualitative for the foreseeable future and based largely on autoecological knowledge of particular species, their habitat requirements, and their interactions with other species. If it is possible to develop a reference standard for the distributional characteristics of wetlands within the ranges of valued species and many ecoregions, then this landscape-level standard could be used to estimate the magnitude of wetland loss due to cumulative human disturbance. The growing body of literature on the consequences of habitat loss and habitat fragmentation can be invoked to estimate the direction and magnitude of changes in biotic diversity to be expected from such disturbance. The relative functional value of individual wetlands (based on their type, size, and location) in maintaining biotic diversity at the landscape level could then be qualitatively estimated.

Conclusions

Evaluations of cumulative effects must estimate the aggregate influence of all anthropogenic disturbances on environmental resources of concern. Therefore, such evaluations should be bounded by the spatial and temporal distribution of these resources and the disturbances to them. Wetland resources of concern have herein been defined in terms of the hydrologic, water quality, and life support functions that wetlands provide. For each function, procedures for establishing boundaries and a generic framework for evaluation have been proposed.

The flood storage function of wetlands operates on the spatial scales of watersheds and river drainages. The cumulative effect of wetland loss on this function should be evaluated relative to the storage roles of affected wetlands in the larger spatial context of their watersheds and river drainages. An appropriately scaled model of the dynamics of the water budget provides an analytical framework to determine the importance of wetland attributes in modifying peak flood depth. This approach has been applied successfully (Ogawa and Male 1986). A more generic approach involves creating a wetland classification scheme relating intrinsic and landscape-level wetland attributes to peak flow characteristics for different regions.

The water quality function of wetlands also operates on the spatial scale of watersheds and river drainages. The cumulative effect of wetland loss or degradation can be evaluated relative to the loss of pollutant loading capacity to the larger hydrologic system. The analytical framework for this evaluation is a model of the dynamics of the pollutant budget of the hydrologic system. Though the technology of water quality modeling is probably sufficient to construct such a model, there are significant limitations in current understanding of the relationship between wetland attributes and processes affecting water quality. These limitations will necessitate the use of simplifying assumptions that may undermine the credibility of such a model. A qualitative synoptic approach is likely to be more tractable in the near term.

The life support function of wetlands operates on the spatial scales of the ranges of valued species and biotic ecoregions. Cumulative effects are best evaluated by estimating the effect of wetland loss or degradation on biotic diversity and/or sustainable yield. A general analytical framework for such an evaluation does not exist but could be derived from principles emerging from the fields of community ecology, conservation biology, landscape ecology, and habitat management. For the foreseeable future, however, evaluations of cumulative effects on the life support function of wetlands will remain qualitative.

Literature Cited

Adamus, P., and L. R. Stockwell. 1983. A method for wetland functional assessment, vol. 1. Critical review and evaluation concepts. U.S. Department of Transportation, Federal Highway Administration Report FHWA-IP-82-83. National Technical Information Service, Springfield, Virginia, 176 pp.

Bailey, R. G. 1976. Ecoregions of the United States. U.S. Forest Service, Intermountain Region, Ogden, Utah, map.

Baskerville, G. 1986. Some scientific issues in cumulative environmental impact assessment. Pages 9–14 *in* G. E. Beanlands, W. J. Erckmann, G. H. Orians, J. O'Riordan, D. Policansky, M. H. Sadar, and B. Sadler (eds.), Cumulative environmental effects: a binational perspective. Canadian Environmental Assessment Research Council/U.S. National Research Council, Ottawa, Ontario and Washington, D.C.

Beanlands, G. E., and P. N. Duinker. 1983. An ecological framework for environmental impact assessment in Canada. Institute for Resource and Environmental Studies, Dalhousie University, Halifax, Nova Scotia and Federal Environmental Assessment Review Office, Hull, Quebec.

Beanlands, G. E., W. J. Erckmann, G. H. Orians, J. O'Riordan, D. Policansky, M. H. Sadar, and B. Sadler (eds.). 1986. Cumulative environmental effects: a binational perspective. Canadian Environmental Assessment Research Council/U.S. National Research Council, Ottawa, Ontario, and Washington, D.C., 166 pp.

Bedford, B. L. 1980. Cooling water discharge into floodplain meadows and marshes: effects, mechanisms, and implications for environmental analysis. Dissertation, University of Wisconsin, Madison, Wisconsin.

Bedford, B. L., and E. M. Preston. 1988. Developing the scientific basis to assess cumulative effects of wetland loss and degradation on landscape functions: status, perspectives, and prospects. *Environmental Management* 12(5):751–771.

Bolin, B., and R. B. Cook (eds.). 1983. The major biogeochemical cycles and their interactions. SCOPE 21. John Wiley and Sons, New York, 532 pp.

Brinson, M. M. 1988. Strategies for assessing the cumulative effects of wetland alteration on water quality. *Environmental Management* 12(5):655–662.

Burgess, R. L., and D. M. Sharpe (eds.). 1981. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, 310 pp.

Burns, L. A. 1986. Validation methods for chemical exposure and hazard assessment models. Pages 148–172 *in* Gesellschaft für Strahlen- und Umwelt- forschung mbH München, Projektgruppe "Umweltgefahrdungspotentiale von Chemikalien" (eds.), Environmental modelling for priority-setting among existing chemicals. Ecomed-Verlagsgesellschaft, München-Landsberg/Lech, West Germany.

Carter, V. 1986. Overview of the hydrologic concerns related to wetlands in the United States. *Canadian Journal of Botany* 64:364–374.

Carter, V., M. S. Bedinger, R. P. Novitzki, and W. O. Wilen. 1978. 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. American Water Resources Association, Minneapolis, Minnesota.

Clark, W. C. 1986. The cumulative impacts of human activities on the atmosphere. Pages 113–123 *in* G. E. Beanlands, W. J. Erckmann, G. H. Orians, J. O'Riordan, D. Policansky, M. H. Sadar, and B. Sadler (eds.), Cumulative environmental effects: a binational perspective. Canadian Environmental Assessment Research Council/U.S. National Research Council, Ottawa, Ontario and Washington, D.C.

Cline, E. W., E. C. Vlachos, and G. C. Horak. 1983. State-of-the-art and theoretical basis of assessing cumulative impacts on fish and wildlife. U.S. Fish and Wildlife Service, Office of Biological Services, Kearneysville, West Virginia, 69 pp.

Cook, R. E. 1981. Pin cherry perceptions. *Natural History* 90:97–102.

Curtis, J. T. 1956. The modification of mid-latitude grasslands and forests by man. Pages 721–736 *in* W. L. Thomas (ed.), Man's role in changing the face of the earth. University of Chicago Press, Chicago, Illinois.

Damman, A. W. H. 1979. Geographic patterns in peatland development in eastern North America. Pages 42–57 *in* Proceedings of the international symposium on classification of peat and peatland. International Peat Society, Hyytisis, Finland.

Erckmann, J. W. 1986. Commentary II, on some scientific issues in cumulative environmental impact assessment. Pages 19–21 *in* G. E. Beanlands, W. J. Erckmann, G. H. Orians, J. O'Riordan, D. Policansky, M. H. Sadar, and B. Sadler (eds.), Cumulative environmental effects: a binational perspective. Canadian Environmental Assessment Research Council/U.S. National Research Council, Ottawa, Ontario and Washington, D.C.

Forman, R. T. T., and M. Godron. 1986. Landscape ecology. John Wiley and Sons, New York, 619 pp.

Giblin, A. E. 1982. Comparisons of the processing of elements by ecosystems. Part 2. Metals. ERC Report 21, Ecosystems Research Center, Cornell University, Ithaca, New York.

Gosselink, J. G., and L. C. Lee. 1988. Cumulative impact assessment principles. Pages 196–203 *in* J. A. Kusler, M. L. Quammen, and G. Brooks (eds.), Proc. Nat. Wetland Symp.: Mitigation of impacts and losses. ASWM Tech Rept 3. Association of State Wetland Managers, Berne, NY.

Gosselink, J. G., and R. E. Turner. 1978. The role of hydrology in freshwater wetland ecosystems. Pages 63–78 *in* R. E. Good, D. F. Whigham, and R. L. Simpson (eds.), Freshwater wetlands, ecological processes and management potential. Academic Press, New York.

Harris, L. D. 1984. The fragmented forest. University of Chicago Press, Chicago, Illinois, 211 pp.

Harris, L. D. 1988. The nature of cumulative impacts on biotic diversity of wetland vertebrates. *Environmental Management* 12(5):675–693.

Harris, L. D., and J. G. Gosselink. Cumulative impacts of bottomland hardwood conversion on hydrology, water quality, and terrestrial wildlife. *In* J. G. Gosselink, L. C. Lee, and T. Muir (eds.), Bottomland hardwood wetlands: ecological and regulatory perspectives, Lewis Publishers, Chelsea, MI, in press.

Hemond, H. F., T. P. Army, W. K. Nuttle, and D. G. Chen. 1987. Pages 519–537 *in* R. A. Hites and S. J. Eisenreich (eds.), Sources and fates of aquatic pollutants. American Chemical Society, Washington, D.C.

Hemond, H. F., and J. Benoit. 1988. Cumulative impacts on water quality functions of wetlands. *Environmental Management* 12(5):639–653.

Hirsch, A. 1988. The regulatory context for cumulative impact research. *Environmental Management* 12(5):715–723.

Inkley, D. B., and S. H. Anderson. 1982. Wildlife communities and land classification systems. Pages 73–81 *in* Transactions of the 47th North American wildlife and natural resources conference. Wildlife Management Institute, Washington, D.C.

Johanson, R. C., J. C. Imhoff, J. L. Kittle, Jr., and A. S. Donigian, Jr. 1984. Hydrological simulation program—Fortran (HSPF): user's manual for release 8.0. EPA 600/3-84-066. U.S. Environmental Protection Agency, Athens, Georgia, 767 pp.

Karr, J. R. 1987. Biological monitoring and environmental assessment: a conceptual framework. *Environmental Management* 11(2):249–256.

Karr, J. R., and K. E. Freemark. 1985. Disturbance and vertebrates: an integrative perspective. Pages 153–168 *in* S. T. A. Pickett and P. S. White (eds.), The ecology of natural disturbance and patch dynamics. Academic Press, New York.

Karr, J. R., K. D. Fausch, P. L. Angermeier, P. R. Yant, and I. J. Schlosser. 1986. Assessing biological integrity in running waters: a method and its rationale. Illinois Natural History Survey special publication 5. Champaign, Illinois, 28 pp.

Larsen, D. P., J. M. Omernik, R. M. Hughes, C. M. Rohm, T. R. Whittier, A. J. Kinney, A. L. Gallant, and D. R. Dudley. 1986. Correspondence between spatial patterns in fish assemblages in Ohio streams and aquatic ecoregions. *Environmental Management* 10: 815–828.

Lee, L. C., and J. G. Gosselink. 1988. Cumulative impacts on wetlands: linking scientific assessments and regulatory alternatives. *Environmental Management* 12(5):591–602.

Levin, S. A. 1987. Scale and predictability in ecological modeling. Pages 2–8 *in* T. L. Vincent, Y. Cohen, W. J. Granthum, G. P. Kirkwood, and J. M. Skowronski (eds.), Modeling and management of resources under uncertainty. Lecture Notes in Biomathematics 72. Springer-Verlag, Berlin.

Levin, S. A., and K. D. Kimball (eds.). 1984. New perspectives in ecotoxicology. *Environmental Management* 8(5):375–442.

Levin, S. A., and R. T. Paine. 1974. Disturbance patch formation, and community structure. *Proceedings of the National Academy of Sciences* (USA) 71:2744–2747.

Lonard, R. I., and E. J. Clairain, Jr. 1986. Identification of methodologies for assessment of wetland functions and values. Pages 66–72 *in* J. A. Kusler and P. Riexinger (eds.), Proceedings of the national wetland assessment symposium. ASWM technical report 1. Association of State Wetland Managers, Chester, Vermont.

Mitsch, W. J. 1983. Ecological models for management of freshwater wetlands. Pages 283–310 *in* S. E. Jorgensen and W. J. Mitsch (eds.), Application of ecological modelling in environmental management, Part B. Elsevier, Amsterdam, The Netherlands.

Mitsch, W. J., J. W. Day, Jr., J. R. Taylor, and C. Madden. 1982. Models of North American freshwater wetlands. *International Journal of Ecology and Environmental Sciences* 8:109–140.

Mitsch, W. J., and J. G. Gosselink. 1986. Wetlands. Van Nostrand Reinhold, New York, 539 pp.

Mooney, H. A., and M. Godron (eds.). 1983. Disturbance and ecosystems: components of response. Springer-Verlag, New York, 292 pp.

Moore, P. D., and D. J. Bellamy. 1974. Peatlands. Springer-Verlag, New York, 221 pp.

Myers, J. P., R. I. G. Morrison, P. Z. Antas, B. A. Harrington, T. E. Lovejoy, M. Sallaberry, S. E. Senner, and A. Tarak. 1987. Conservation strategy for migratory species. *American Scientist* 75:19–26.

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 saltwater wetlands as sources, sinks, and transformers of nitrogen, phosphorus, and various heavy metals. Technical report Y-86-2. U.S. Army Corps of Engineers, Vicksburg, Mississippi, 229 pp.

Novitzki, R. P. 1978. Hydrologic characteristics of Wisconsin's wetlands and their influence on floods, stream flow, 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. American Water Resources Association, Minneapolis, Minnesota.

O'Brien, A. L. 1988. Evaluating the cumulative effects of alteration on New England wetlands. *Environmental Management* 12(5):627–636.

Ogawa, H., and J. W. Male. 1986. Simulating the flood mitigation role of wetlands. *Journal of Water Resources Planning and Management* 112:114–128.

Omernik, J. M. 1987. Ecoregions of the conterminous United States. *Annals of the Association of American Geographers* 77(1):118–125.

O'Neill, R. V., D. L. DeAngelis, J. B. Waide, and T. F. H. Allen. 1986. A hierarchical concept of ecosystems. Monographs in population biology 23. Princeton University Press, Princeton, New Jersey, 253 pp.

Paine, R. T., and S. A. Levin. 1981. Intertidal landscapes: disturbance and the dynamics of pattern. *Ecological Monographs* 51:145–178.

Pickett, S. T. A., and P. S. White. 1985. The ecology of natural disturbance and patch dynamics. Academic Press, New York, 472 pp.

Reiners, W. A. 1987. Escaping paradigms: multiple models for ecosystems. Page 5 *in* G. E. Likens, J. J. Cole, J. Kolasa, J. B. McAninch, M. J. McDonnell, G. G. Parker, D. L. Strayer (eds.), Status and future of ecosystem science 3, The Institute of Ecosystem Studies, The New York Botanical Garden, Millbrook, New York, 23 pp.

Richardson, C. J. 1986. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. *Science* 228:1424–1427.

Ricklefs, R. E. 1987. Community diversity: relative roles of local and regional processes. *Science* 235:161–171.

Risser, P. G., J. R. Karr, and R. T. T. Forman. 1984. Landscape ecology: directions and approaches. Illinois Natural History Survey special publication 2. Champaign, Illinois, 16 pp.

Robilliard, G. A. 1986. Commentary 1, on management of the estuarine ecosystem against cumulative effects of pollution and development. Pages 107–108 *in* G. E. Beanlands W. J. Erckmann, G. H. Orians, J. O'Riordan, D. Policansky, M. H. Sadar, and B. Sadler (eds.), Cumulative environmental effects: a binational perspective. Canadian Environmental Assessment Research Council/U.S. National Research Council, Ottawa, Ontario, and Washington, D.C., 166 pp.

Root, R. B. 1973. Organization of a plant-arthropod association in simple and diverse habitats: the fauna of collards (*Brassica oleracea*). *Ecological Monographs* 43:95–124.

Rudis, V. A., and A. R. Ek. 1981. Optimization of forest island spatial patterns: methodology for analysis of landscape pattern. Pages 241–256 *in* R. L. Burgess and D. M. Sharpe (eds.), Forest island dynamics in man-dominated landscapes. Ecological Studies 41, Springer-Verlag, New York.

Sather, J. H., and P. J. R. Stuber (technical coordinators). 1984. Proceedings of the national wetland values assessment workshop. U.S. Fish and Wildlife Service, Western Energy and Land Use Team, FWS/OBS-84/12, Fort Collins, Colorado, 100 pp.

Schonewald-Cox, C. M., S. M. Chambers, B. MacBryde, and W. L. Thomas (eds.). 1983. Genetics and conservation: a reference for managing wild animal and plant populations. Benjamin/Cummings, Menlo Park, California, 722 pp.

Siegel, D. I. 1988. Evaluating cumulative effects of disturbance on the hydrologic function of bogs, fens, and mires. *Environmental Management* 12(5):621–626.

Simberloff, D., and L. G. Abele. 1982. Refuge design and island biogeographic theory: effects of fragmentation. *The American Naturalist* 120(1):41–50.

Soulé, M. E. 1986. Conservation biology—the science of scarcity and diversity. Sinauer Associates, Sunderland, Massachusetts, 584 pp.

Soulé, M. E., and D. Simberloff. 1986. What do genetics and ecology tell us about the design of nature reserves? *Biological Conservation* 35:19–40.

Soulé, M. E., and B. A. Wilcox (eds.). 1980. Conservation biology: an evolutionary-ecological perspective. Sinauer Associates, Sunderland, Massachusetts, 382 pp.

Sprague, L. G., and C. R. Sprague. 1976. Management science? *Interfaces* 7:57–62.

Stakhiv, E. Z. 1988. An evaluation paradigm for cumulative impact analysis. *Environmental Management* 12(5):725–748.

Urban, D. L., R. V. O'Neill, and H. H. Shugart, Jr. 1987. Landscape ecology. *BioScience* 37:119–127.

Verner, J., M. Morrison, and C. J. Ralph (eds.). 1986. Modeling habitat relationships of terrestrial vertebrates. Wildlife 2000. University of Wisconsin Press, Madison, Wisconsin, 470 pp.

Vitousek, P. M. 1985. Community turnover and ecosystem nutrient dynamics. Pages 325–334 *in* S. T. A. Pickett and P. S. White (eds.), The ecology of natural disturbance and patch dynamics. Academic Press, New York.

Vlachos, E. 1985. Assessing long-range cumulative impacts. Pages 49–80 *in* V. T. Covello, J. L. Mumpower, P. J. M. Stallen, and V. R. R. Uppuluri (eds.), Environmental impact assessment, technology assessment, and risk analysis. Springer-Verlag, New York.

Vrijenhoek, R. C. 1985. Animal population genetics and disturbance: the effects of local extinctions and recolonizations on heterozygosity and fitness. Pages 266–285 *in* S. T. A. Pickett and P. S. White (eds.), The ecology of natural disturbance and patch dynamics. Academic Press, New York.

Weller, M. W. 1988. Issues and approaches in assessing cumulative impacts on waterbird habitat in wetlands. *Environmental Management* 12(5):695–701.

Whigham, D. F., C. Chitterling, and B. Palmer. 1988. Impacts of freshwater wetlands on water quality: a landscape perspective. *Environmental Management* 12(5):663–671.

White, P. S., and S. T. A. Pickett. 1985. Natural disturbance and patch dynamics: an introduction. Pages 3–16 *in* S. T. A. Pickett and P. S. White (eds.), The ecology of natural disturbance and patch dynamics. Academic Press, New York.

Wilson, E. O. 1988. Biodiversity. National Academy Press, Washington, D.C., 521 pp.

Winter, T. C. 1988. A conceptual framework for assessing cumulative impacts on the hydrology of nontidal wetlands. *Environmental Management* 12(5):605–620.