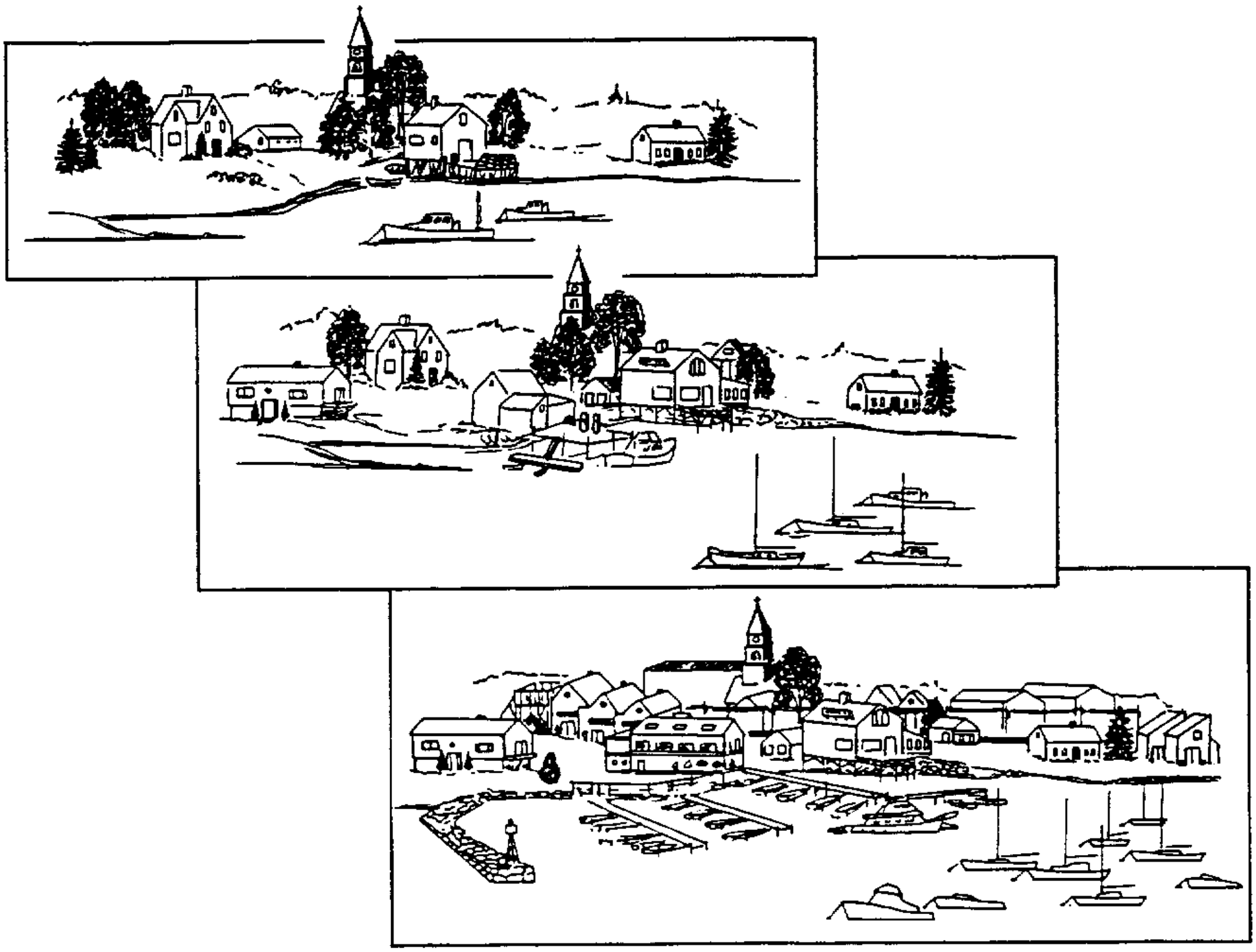
1626

METHODOLOGY AND MECHANISMS FOR MANAGEMENT OF CUMULATIVE COASTAL ENVIRONMENTAL IMPACTS

WORKSHOP MATERIALS



MARINE LAW INSTITUTE
University of Maine School of Law
246 Deering Avenue
Portland, ME 04102

April 1993

Prepared under NOAA Coastal Ocean Program Award #NA26RG0247-01

Preface

The following materials were compiled as background information for individuals who will be participating in a workshop entitled "Methodology and Mechanisms for Management of Cumulative Coastal Environmental Impacts," to be held in Narragansett, Rhode Island on May 6th and 7th, sponsored by NOAA's Coastal Ocean Program. They consist of selected articles and excerpts from reports authored by individuals who will be making presentations on specific methodologies (S. Leibowitz, C. Hunsaker, S. Williamson, G. Shaffer and W. Eichbaum); an article by a respondent to the methodologies panel (C. Contant); and materials authored by two individuals who were unable to participate in the workshop (B. Bedford and J. Gosselink). Also enclosed with these materials, but sent as a separate document, is "Making Decisions on Cumulative Environmental Impacts: A Conceptual Framework," World Wildlife Fund, 1992, co-authored by workshop presenter Frances Irwin.

It is the hope of workshop organizers that participants familiarize themselves with these materials prior to the workshop so that discussions can start from a shared understanding of the basic concepts and approaches presented by the authors. These materials may also serve as a valuable source of information for participants who wish to pursue particular approaches in more detail after the workshop.

In organizing the workshop and selecting these materials, an attempt has been made to identify state-of-the-art methodologies for assessing and managing cumulative environmental impacts. While the focus of the workshop is application in a coastal environment, methodologies developed in other contexts are included as well.

During the last three years there has once again been a resurgence of interest in research and writing about assessment and management of cumulative environmental impacts. However, due to our particular focus, these workshop materials only include a fraction of the valuable material in the literature. These materials are intended for the limited use of workshop participants only, and are not to be reproduced for wider distribution.

Alison Rieser
Associate Professor and
Director, Marine Law Institute

Barbara A. Vestal, Project Coordinator
Associate Director
Marine Law Institute

Increasing the Scale of Analysis: The Challenge of Cumulative Impact Assessment for Great Lakes Wetlands

Barbara L. Bedford
Cornell University
Ithaca, New York

INTRODUCTION

"A fundamental incongruity confronts those who regulate or study wetland ecosystems. The scale at which they observe human impacts on wetland resources to be accumulating is far greater than the scale at which they ask questions or make decisions. Entire wetland landscapes have been altered inadvertently through the cumulative effects of numerous localized individual actions. Insights gained through research conducted at one site and on one process cannot provide straightforward answers about the consequences of multiple interacting processes operating at the scale of watersheds and landscapes." (Preface to Bedford and Preston, 1988b)

I thus described in an earlier work what I here refer to as an incongruity of scale in both the way we conduct scientific inquiry and the way we regulate and manage the environment. Within the regulatory world, it is referred to as the problem of cumulative impact assessment.

This paper presents the conceptual framework for resolving that incongruity for wetlands of the Great Lakes. I argue that the primary need is for a shift upward in the level of analysis at which we conduct assessments. Rather than the individual project, discharge, or site, the minimum level of analysis needs to be that of the landscape -- watersheds, individual lake basins, and the entire Great Lakes Basin. I then discuss various elements necessary to making that shift: guidelines for establishing temporal and spatial boundaries, a functional approach to classifying wetlands and wetland types within the Great Lakes Basin, the information required to provide a context for decision-making at larger temporal and spatial scales, and a provisional set of goals for the entire wetland resource base of the Great Lakes.

In developing the framework, I have drawn heavily on my own previous work (Bedford and Preston, 1988a; Preston and Bedford, 1988) and several of the articles in Bedford and Preston (1988b), especially Lee and Gosselink (1988) and Brinson (1988).

DEFINITIONS, SCALES, AND BOUNDARIES

Conventional vs. Cumulative Impact Scales

Regulations published by the Council on Environmental Quality to implement the 1969 U.S. National Environmental Policy Act define cumulative impact 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 ... 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)." (Bolding mine.)

The fundamental aspects of this definition (see bolding) dictate a change in perspective -- an increase in the level of analysis both temporally and spatially (Figure 1). The scale for conventional impact assessment has been that of a particular development, project or discharge. For conventional cumulative assessments, the scale increases to that of an individual wetland, within which several projects or activities are considered. I have argued that the appropriate scale for wetlands is even larger -- "that of interacting systems of wetlands located within watersheds, landscapes, and regions. The assessment then becomes bounded by the distribution (spatial and temporal) of the resources of concern and considers the total effect of all human activities and their interrelationships on all wetland functions within these landscape systems" (Bedford and Preston, 1988b). If the resource of concern is the Great Lakes,

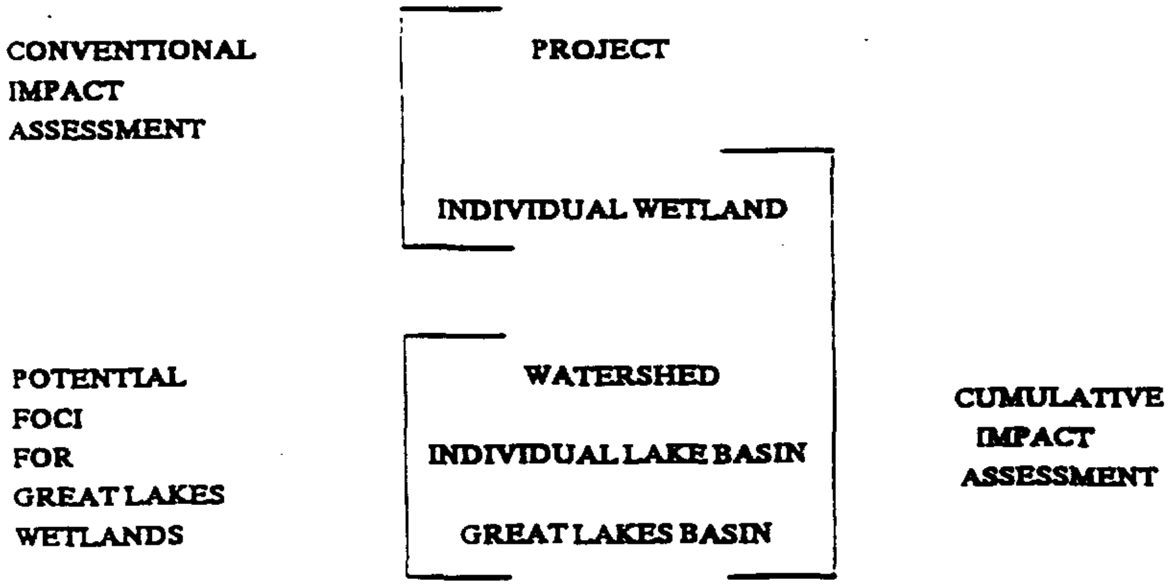


Figure 1. SPATIAL SCALES OF CONVENTIONAL AND CUMULATIVE IMPACT ASSESSMENT

then wetlands of the entire basin need to be considered in assessments.

Why Use Larger Scales

The reasons for increasing the scale of analysis are both political and scientific. First, the shift in scale is consistent with the expressed policy of the International Joint Commission to adopt an ecosystems approach to the Great Lakes (National Research Council and The Royal Society, 1985), with the U.S. National Environmental Policy Act (see above), and with various pieces of state and federal wetland legislation. The wetland functions valued by society and frequently invoked in legislation -- hydrologic, water quality, and life support functions -- are not the product of a single wetland but of a wetland's relationship to other wetlands, other ecosystems, and other land use types. They derive from the role wetlands play as components of larger landscape units. Consider, for example, wetland water quality functions. Inputs from terrestrial systems, as well as wetland characteristics, influence the effect a wetland may have on downstream waters. And because outputs from one system are inputs to another, spatial relationships of wetlands to other ecosystem types within a landscape influence the movement of pollutants within that landscape (Whigham et al., 1988). Migratory waterfowl, that use both uplands and wetlands for breeding and feeding (Cowardin et a 1985), and shorebirds, that rely on many wetlan along continental-scale flyways for feeding, restin and breeding (Myers et al., 1987), further em the need for larger scale analyses. Analysis, plan and regulatory decisions should be made, therefo with regard to these relationships of one system another.

Second, both the development and maintenan of wetlands within a landscape reflect large-sca factors and long-term processes, as well as more lo processes (Gorham, 1957; Damman, 1979; Ge 1985; Winter, 1988). Landscape and regional va ables such as climate, topography, geology, soi vegetation, and land use patterns determine hydr logic variables which in turn determine if wetlan will form, where they will form, and what their bi geochemical properties will be. Wetlands within t Great Lakes Basin formed over thousands of ye following the retreat of the Wisconsin glaciati some 10,000 years ago (Heinselman, 1963; 19 Friedman and DeWitt, 1978). Geis (1985) identifi variables correlated with shoreline morphology a hydrologic regime as the primary features defini the environmental gradients along which Gr Lakes wetlands developed.

Third, the scale at which degradation and los have occurred and are occurring is far larger than individual site, or even sub-watersheds. The pattern has reversed from one of scattered local impacts and losses to loss or degradation of most of the wetlands within the entire Great Lakes Basin. No single inventory for the basin as a whole exists, nor a definitive baseline against which to judge the precise extent of losses (Weller, 1988). Yet a clear picture of the magnitude of wetland losses emerges when assessments of individual portions of the basin are considered together (Table 1). The western end of Lake Erie, once an extensive marsh of approximately 4000 km^2, now contains only about 150km^2 of wetlands, most of which are diked (Herdendorf, 1987). The Green Bay portion of Lake Michigan has lost approximately 60% of the coastal marshland that existed in the 1840s at comparable water levels (Harris et al., 1977). Other portions of Lake Michigan, as well as other lakes, similarly have experienced extensive losses.

Of the wetlands that remain, the general picture is one of degradation, with multiple impacts of human activities accumulating over time. Historic trends of increasing human impact on Great Lakes wetlands have been well documented (Trautman, 1977; Mudroch, 1980; Herdendorf et al., 1986; Herdendorf, 1987; Stuckey, 1989; Colburn, this volume). The combined effects of agriculture (draining, diking, pesticide and fertilizer runoff), lake commerce (dredging and disposal), water level regulation (Bedford et al., 1976), construction of roads and railways, residential development, wetland diking, and point and nonpoint discharges of pollutants, including toxic substances (see Evans, 1988), have resulted in eutrophication, alteration in historic patterns of water level fluctuation, elevated concentrations of heavy metals and toxic substances (e.g., PCB's) in sediments, fragmentation of habitat, loss of diversity, and invasion of exotic species in most remaining wetlands.

RUDIMENTS OF A BASIN-WIDE ASSESSMENT

Although additional work specific to the Great Lakes would be required for implementation, the rudiments of a basin-wide assessment of the cumulative effects of wetland loss and degradation can be outlined at this stage on the basis of previous work for other systems (Preston and Bedford, 1988; Bedford and Preston, 1988a; Lee and Gosselink, 1988; Gosselink and Lee, 1989). The basic elements for such an assessment are: (1) guidelines for establishing the spatial and temporal boundaries of the assessment; (2) a functional classification of wetlands in the basin; (3) providing context for decision-making; and (4) establishing goals. Major progress in each of these areas can be made for the Great Lakes Basin with systematic effort.

Because the political decision already has been made to view the entire set of lakes as an ecosystem (National Research Council and the Royal Society, 1985), and because watershed boundaries for the lakes are well-known, guidelines for setting spatial boundaries do not need to be established. For the Great Lakes, the appropriate scales for assessing cumulative effects are (Figure 1): (1) the sub-watershed — the watersheds of major streams and rivers feeding into individual lakes; (2) the drainage basin of each individual lake, which would subsume all the sub-watersheds; and (3) the drainage basin for the entire Great Lakes, which subsumes all individual drainage basins. These scales then establish the spatial boundaries for assessment in a straightforward and obvious way at the boundaries of the individual and aggregated watersheds (see Botts and Krushelnicki, 1987).

Where temporal boundaries for the assessment should be set are less obvious. The CEQ definition refers to "past" and "reasonably foreseeable future actions" but doesn't say how far into the past nor what is the reasonably foreseeable future. Clark (1986) correctly identified a cumulative effect as occurring if the system in question had not recovered from a previous disturbance, no matter how long ago the disturbance occurred. Preston and Bedford (1988) adopted this basis for setting time scales and recognized the several time scales over which processes controlling different wetland functions operate and recover. Lee and Gosselink (1988) and Gosselink and Lee (1989) emphasized the long time scales of some ecosystem processes (e.g., development time for a bottomland forest) and the permanence of many types of wetland alterations. They also noted the practical reality that historic data for wetland functions seldom go back more than 20-50 years. Given the lack of historic data for Great Lakes wetland functions, I suggest that we use structure for defining the boundaries of "past" -- i.e., the pre-settlement area of wetlands. A rational basis for setting the boundary into the future is less easily defined but certainly should include at least one to two human generations (i.e., 20-40 years).

Functional Classification

If all the wetlands of the Great Lakes Basin are to be considered, then some basis for simplifying the diversity of types must be identified in order to develop a picture of the resource as a whole without overwhelming data bases and the decision-making process. In Bedford and Preston (1988a), we developed the rationale for such a simplification. We

Table 1. SOME ESTIMATES OF CUMULATIVE WETLAND LOSSES FOR VARIOUS SECTIONS OF THE GREAT LAKES

GREAT LAKE & SECTION	% LOST	SOURCE
Lake Superior	?	
Lake Michigan		
Green Bay	~ 60	Harris et al. 1977
Northern Indiana	≥ 71	IDNR 1987 [in Weller 1988]
Michigan - various sections	50 - 72	Jaworski & Raphael 1978 [in Weller 1988]
Lake Huron	?	
Lake St. Clair	> 41	Herdendorf et al. 1986
Michigan	~ 72	Jaworski & Raphael 1976 [in Edsall et al. 1988]
Lake Erie		
Ohio since ~ 1950	> 56	Weeks 1975
since ~ 1850	~ 93	
Michigan	~ 62	Jaworski & Raphael 1978 [in Herdendorf 1987]
Lake Ontario		
Western end - Niagara River to Oshawa	83	McCullough 1977
Canadian shore west of Bay of Quinte - various sections	8 - 100	Whillans 1982
North shore - Oshawa through Prince Edward County	11	McCullough 1977

referred to it as a functional grouping or classification because it was based not on species composition or community properties but rather on characteristics of wetlands that determine their functioning. The scheme we proposed was based on three synthetic variables which strongly influence functioning: landscape variables controlling hydrology, geomorphology and position in the landscape, and soil properties. Hydrology directly or indirectly determines all structural and functional characteristics of wetlands. How long, how often, and when wetlands are flooded determines rates of biogeochemical processes and vegetation patterns. Paths and rates of water movement control the transport of pollutants and sediments. Classifying wetlands on the basis of geomorphology and landscape position stratifies them according to their landform and surrounding landforms. These patterns influence water movement, vectors and rates of water transport and nutrient regimes within wetlands, and the openness of the wetland to exchanges with adjacent systems. For example, wetlands fringing lake shores are far more open to exchanges with the lakes than wetlands behind barrier beaches that are infrequently breached. Soil properties strongly influence vegetation dynamics and biogeochemical cycling, including phosphorus retention (Richardson, 1985), and the accumulation of metals and toxic substances in sediments (Feijtel et al., 1988; Gambrell and Patrick, 1978; 1988). Refined soil measurements are not necessary for classification purposes but broad-scale differences in percent organic matter (high/low), mineral content (high/low), soil depth (shallow/deep), and particle size (muck/peat, sand/clay, etc.) can differentiate wetlands on the basis of variables strongly correlated with function.

A functional classification for Great Lakes wetlands could be developed by modifying Brinson's (1988) scheme to make it more specific to the Great Lakes. Brinson (1988) offered an initial basis for classifying wetlands on the basis of geomorphology and landscape position. He distinguished riverine, basin, and fringing wetlands because of fundamental differences in characteristics of these types of systems that control element cycles affecting water quality. A provisional classification for the Great Lakes (Table 2) would differentiate four major classes of wetlands: (1) lakeshore, (2) estuarine, (3) riverine, and (4) wetlands occurring within watershed basins draining into the lakes but not themselves in surface water contact with the lakes or their tributaries. The influence of this last type of wetland on the lakes might be exerted through effects on groundwater moving to the lakes or through their role as habitat for species using the lakes. Conversely, creating a surface water connection by draining these wetlands into tributaries would be expected to influence downstream water quality. Wetlands fringing the lakeshores would be divided further into those on shorelines and directly in contact with the lake (lacustrine), those behind barrier beaches or sand spits but with some surface water contact with the lake (barrier), those separated from the lake by dikes (diked), and those situated behind dunes that may not have surface water contact but are likely in subsurface contact through porous sands with the lake-level water table. Estuaries, riverine wetlands, and watershed basin wetlands, likewise, may need to be further differentiated where essential differences among them indicate major effects on functioning. Whether or not differences are likely to produce major effects can be evaluated by examining existing knowledge of variables determining hydrologic relationships (Bedford et al., 1976; Winter, 1988; Novitzki, 1982) and soil properties.

To this provisional functional classification must be added some basis for differentiating and evaluating biotic diversity. While many Great Lakes wetlands are seriously degraded, other wetlands support large populations of waterfowl (Bookhout et al., 1989), hundreds of species of vascular plants (Keddy and Reznicek, 1985; Stuckey, 1989), as well as uncommon plant communities and plant species (Carol Reschke, New York State Natural Heritage Program, pers. comm.; Crispin, this volume). Efforts already underway to identify and protect areas of high biotic diversity or unique natural heritage value (Report of the Great Lakes Science Advisory Board, 1989; Crispin, this volume), along with elements of the North American Waterfowl Management Plan (U.S. Fish and Wildlife Service and Canadian Wildlife Service, 1986; U.S. Fish and Wildlife Service, 1988), could provide the basis for integrating habitat and diversity functions into a functional classification for the Great Lakes.

Providing Context for Decision-Making

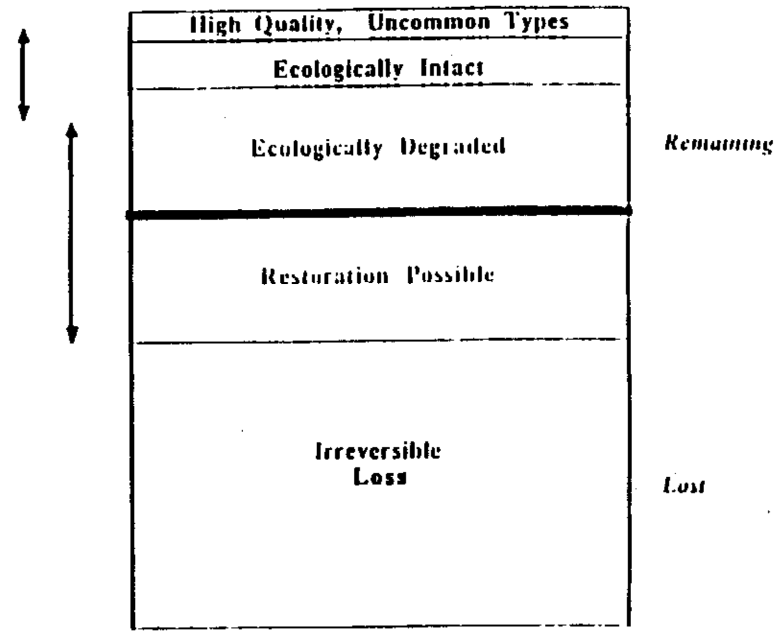
The essence of cumulative impact assessment requires that decisions be put in context. Cumulative effects, by definition, are landscape level, long-term phenomena. They occur as the consequence of numerous human activities in wetlands and wetland landscapes over time. Focusing on individual sites, projects, or species necessarily misses these larger-scale and longer-term patterns. Decision-making that is to be effective at the scales relevant to the Great Lakes as an ecosystem, therefore, must be put in the context of past and future actions affecting wetlands, as well as current patterns of impacts.

That context can be provided by developing a common geographic information system (GIS) for

Table 2. PROVISIONAL FUNCTIONAL CLASSIFICATION FOR GREAT LAKES WETLANDS

- **LAKESHORE/FRINGE WETLANDS**
  - **lacustrine**
  - **barrier**
  - **diked**
  - **interdunal**
- **ESTUARINE**
- **RIVERINE**
- **WATERSHED BASIN WETLANDS**

Figure 2. SCHEMATIC VIEW OF THE GREAT LAKES TOTAL WETLAND RESOURCE



Great Lakes wetlands. Such a system could be used to: (a) document historical patterns of wetland loss, (b) describe historical patterns of wetland degradation, (c) develop a geographic analysis of present trends in loss and degradation, and (d) establish a dynamic data base that locates and summarizes information on wetlands by functional class and quality, as well as by individual lake. With such a system available to them, decision-makers would have the tools for truly increasing the scale at which information on wetland processes and impacts is gathered, tracked through time, and analyzed. A context would be provided for decisions at several scales, from local permits to the setting of basin-wide priorities for protection, regulation, or special management. Principles for the use of a GIS in land resources assessment are well established (Burrough, 1986) and its many advantages for cumulative impact assessment already have been described by Johnston et al. (1988). Bailey (1988) has identified some of the pitfalls associated with creating map overlays -- either manually or with a GIS -- for assessment or planning purposes.

Establishing Goals

The most difficult but essential component of cumulative impact assessment is establishing goals for the resource under consideration (Lee and Gosselink, 1988; Gosselink and Lee, 1989; Bedford and Preston, 1988b). Goals may be based on scientific information but they are values, not facts, and as such cannot be objectively defined. Some goals may conflict with others. Nonetheless, as Lee and Gosselink (1988, p. 600) point out:

> "Goal setting determines the levels of cumulative effects that are to be interpreted as impacts. Goals drive the interpretation of the direction a proposed activity will have on maintaining the integrity of the landscape unit. Because the impact of most single permit requests in not detectable at the landscape level, direction of the impact with respect to the goal should be the regulatory concern, rather than just absolute magnitude of the individual impact and its significance in contributing to degradation of flood storage, water quality, and life support functions."

A general goal for Great Lakes wetlands is implicit in agreements already reached between the United States and Canada to restore the integrity of the Great Lakes ecosystem. As Steedman and Regier (1987) have argued, wetlands will play a central role in rehabilitating the Great Lakes. Based on my own qualitative assessment of the current status of Great Lakes wetlands, I suggest below a set of provisional goals. These goals will need to be refined and modified on the basis of discussion among relevant parties. They are offered here as the departure point for those discussions.

My formulation of these goals is based on a qualitative picture I have acquired in reviewing existing knowledge about Great Lakes wetlands. That picture is best conveyed in a schematic that depicts in a conceptual way the current state of affairs (Figure 2). If the size of the entire box represents the total wetland resource as it existed at the time of European settlement of the Great Lakes Basin, then the current picture is represented by the relative sizes of the internal boxes. The area below the thick horizontal line represents a minimal estimate of what has been lost since settlement time. No exact estimates for the entire Great Lakes Basin exists, but there is little doubt that more than half of the original wetland base has been lost. The actual figure is probably a good deal higher, given that extensive areas had been drained for agriculture well before we began making inventories of wetlands (Weeks, 1975). Much of that loss is irreversible, either literally or practically; cities now occupy many former wetland sites (e.g., Monroe, Michigan (Herdendorf, 1987)) and millions of acres have been drained so long for agricultural purposes that soils and hydrology have been altered severely. Restoration would be prohibitively expensive and/or would require an unrealistically-long time frame. Some areas remain where hydrologic patterns supportive of wetland development could be restored.

The largest fraction of wetlands remaining, represented as the area above the thick horizontal line in Figure 2, are ecologically degraded, usually for several reasons (see discussion above). Only a small fraction of the original wetland resource base is ecologically intact with their structure and functioning still largely undisturbed. High quality sites and sites supporting uncommon wetland communities, or uncommon plant or animal species, are but a tiny fraction of what once was a rather large wetland resource base.

The arrows to the left of the box (Figure 2) indicate the areas where we still have choices. I would argue that for those wetlands in the two boxes indicated by the upper arrow our choice should be preservation with no loss or degradation allowed. The cumulative loss and degradation to date are so great, and ecologically intact and high quality sites are such a small fraction of the original resource, that we have no latitude left if we are to retain even remnants of our wetland heritage and examples of systems from which we might learn to restore other systems. Only in the area indicated by the lower arrow (Figure 2) is there room for decision-making. It is for these types of wetland that goal setting will be critical for guiding those decisions.

Given this qualitative picture, I offer the following goals as a provisional basis for cumulative impact assessment of Great Lakes wetlands:

a. Manage to increase total area in wetland. Wetland area has been reduced significantly within the basin. If any further loss is permitted, it should be only from already degraded sites and should be compensated with restoration of other wetlands.

b. Manage to sustain or increase the diversity of wetland "types." The Great Lakes at one time undoubtedly supported a rich diversity, not only of species, but of different types of wetlands. Loss and degradation have reduced that original diversity. For example, diked marshes and cattail marshes are now among the more common types, while fens and sedge meadows are uncommon. An effort should be made to describe and restore some of the original richness of wetland species and communities.

c. Manage to preserve uncommon and high quality sites. In no case should a high quality site or an uncommon type be destroyed. There are already too few of them. While some people might disagree on what constitutes a high quality site, an uncommon type is an objective assessment that can be made on the basis of an inventory of all types and analysis of their relative frequency within individual lake basins and the entire Great Lakes Basin. Ontario already has made significant progress in identifying "provincially significant wetlands" (Glooschenko, 1985). The work of The Nature Conservancy in the U.S. and The Nature Conservancy of Canada on Great Lake wetlands (Crispin, this volume) can help define what constitutes a high quality site. In general, high quality sites are sites little disturbed by human activity.

d. Manage to sustain or improve function. While structure generally reflects function, wetlands may retain some semblance of wetland structure and yet show impaired functioning (e.g., Irondequoit Bay, Green Bay). These functions include the capacity to support a diverse assemblage of plants and animals and to regulate the flow of nutrients, sediments, and other pollutants moving through them. Assessing effects on functions is difficult but any activity that has major effects on the magnitude and the spatial and temporal patterns of water flow is likely to influence functioning and should be examined intensively.

e. Manage to sustain dynamic structure. Great Lake wetlands were formed and are maintained because of natural historic patterns in the flows of water, nutrients and other elements, and sediments into them. These forces are dynamic and their natural patterns must be maintained if the wetlands are to be maintained. In large part, maintaining dynamic structure means maintaining historical spatial relationships in the landscape, i.e., not cutting wetlands off from natural water flows, and regulating water levels to closely follow historic patterns (Bedford et al., 1976; Keddy and Reznicek, 1985). Because water levels have fluctuated historically, wetlands must have the spatial flexibility to absorb short and long-term fluctuations on their landward and lake sides. Dikes and other structures that constrain the spatial extent of flooding, or impede water flow, destroy this flexibility. Stabilizing water levels artificially or regulating outside the range of historic fluctuations eliminates the dynamic patterns that allows a diversity of wetland species and communities to exist.

f. Manage to minimize fragmentation. Large blocks of wetland that have not been cut up or encroached upon by human activity are uncommon elements of the Great Lakes landscape. Such fragmentation not only influences the flows of water and elements, it also influences the movement of animals and th capacity of the wetland to support large populations of wildlife or species that require large territories or distance from human activi (Harris, 1988). Every effort should be made to retain such large tracts intact.

SUSTAINABLE REDEVELOPMEN

Sustainable development of the biosphere h become a common theme of those seeking to i tegrate environmental concerns with developme (World Commission on Environment and Develo ment, 1987). Those working in the Great Lal region, which contains two of the most develop countries in the world, now recognize that sustaini the Great Lakes region means "redevelopment" o regional ecosystem degraded by exploitative devel ment in the past (Regier and Baskerville, 1986; Ste man and Regier, 1987; Harris et al., 1990). The Gr Lakes wetlands have been part of that pattern exploitation. At this stage in history, we still have t potential to reverse that pattern. If we are to do th decision-making cannot be limited to narrow poi in space and time, but must take the broader view. this paper I have outlined the rudiments for tak that broader view and proposed some regional goals as points for discussion in hopes of moving the people of the Great Lakes region toward sustainable redevelopment of their wetland heritage.

REFERENCES

Bailey, R. G. 1988. Problems with using overlay mapping for planning and their implications for geographic information systems. Environmental Management 12:11-17.

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

Bedford, B. L. and E. M. Preston (eds). 1988b. Cumulative effects on landscape systems of wetlands: scientific status, prospects, and regulatory perspectives. Environmental Management 12(5):561-776.

Bedford, B. L., R. Emanuel, J. Erickson, S. Rettig, R. Richards, S. Skavroneck, M. Veprauskas, R. Walters, and D. Willard. 1976. An Analysis of the International Great Lakes Levels Board Report on Regulation of Great Lakes Water Levels: Wetlands, Fisheries, and Water Quality. Rockefeller Foundation Monograph 76-04. IES Working Paper 30. University of Wisconsin, Madison, Wisconsin.

Bookhout, T. A., K. E. Bednarik, and R. W. Kroll. 1989. The Great Lakes marshes. *In* L. M. Smith, R. L. Pederson, and R. M. Kaminski (eds.), Habitat Management for Migrating and Wintering Waterfowl in North America. Texas Tech University Press, Lubbock, Texas.

Botts, L. and B. Krushelnicki. 1987. The Great Lakes, An Environmental Atlas and Resource Book. United States Environmental Protection Agency and Environment Canada, Chicago, Illinois.

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

Burrough, P. A. 1986. Principles of Geographical Information Systems for Land Resources Assessment. Clarendon Press, Oxford.

Clark, W. C. 1986. The cumulative impacts of human activities on the atmosphere. *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.

Colburn, T. (This volume).

Cowardin, L. M., D. S. Gilmer, and C. W. Shaiffer. 1985. Mallard recruitment in the agricultural environment of North Dakota. Wildlife Monographs 92:1-37.

Crispin, S. (This volume.)

Damman, A. W. H. 1979. Geographic patterns in peatland development in eastern North America. Pp. 42-57 *in* Proceedings of the International Symposium on Classification of Peat and Peatlands. International Peat Society, Hyytiälä, Finland.

Edsall, T. A., B. A. Manny, and C. N. Raphael. 1988. The St. Clair River and Lake St. Clair, Michigan: An Ecological Profile. U. S. Fish and Wildlife Service Biological Report 85(7.3). Washington, D.C.

Evans, M. S., ed. 1988. Toxic Contaminants and Ecosystem Health: A Great Lakes Focus. John Wiley and Sons, New York.

Feijtel, T. C., R. D. DeLaune, and W. H. Patrick, Jr. 1988. Biogeochemical control on metal distribution and accumulation in Louisiana sediments. Journal of Environmental Quality 17(1):88-94.

Friedman, R. M. and C. B. DeWitt. 1978. Wetlands as carbon and nutrient reservoirs: a spatial, historical, and societal perspective. *In* P. E. Greeson, et al. (eds.), Wetland Functions and Values: The State of Our Understanding. Proceedings of the National Symposium on Wetlands. American Water Resources Association, Minneapolis, Minnesota.

Gambrell, R. P. and W. H. Patrick, Jr. 1978. Chemical and microbiological properties of anaerobic soils and sediments. *In* D. D. Hook and R.M.M. Crawford (eds.), Plant Life in Anaerobic Environments. Ann Arbor Science, Ann Arbor, Michigan.

Gambrell, R. P. and W. H. Patrick, Jr. 1988. The influence of redox potential on the environmental chemistry of contaminants in soils and sediments. *In* D. D. Hook, et al. (eds.), The Ecology and Management of Wetlands. Volume 1: Ecology of Wetlands. Timber Press, Portland, Oregon.

Geis, J. W. 1985. Environmental influences on the distribution and composition of wetlands in the Great Lakes Basin. *In* H. H. Prince and F. M. D'Itri (eds.), Coastal Wetlands. Lewis Publishers, Inc., Chelsea, Michigan.

Glooschenko, V. 1985. Characteristics of provincially significant wetlands as assessed by the Ontario wetland evaluation system. *In* H. H. Prince and F. M. D'Itri (eds.), Coastal Wetlands. Lewis Publishers, Inc., Chelsea, Michigan.

Gorham, E. 1957. The development of peat lands. Quarterly Review of Biology 32:145-166.

Gosselink, J. G. and L. C. Lee. 1989. Cumulative impact assessment in bottomland hardwood forests. Wetlands 9:89-174.

Great Lakes Science Advisory Board. 1989. Report of the Great Lakes Science Advisory Board to the International Joint Commission, Hamilton, Ontario.

Harris, H. J., P. E. Sager, H. A. Regier, and G. R. Francis. 1990. Ecotoxicology and ecosystem integrity: the Great Lakes examined. Environmental Science and Technology 24(5):598-603.

Harris, H. J., T. R. Bosley, and F. D. Roznik. 1977. Green Bay's coastal wetlands: a picture of dynamic change. *In* C. DeWitt and E. Soloway (eds.), Wetlands Ecology, Values and Impacts. Proceedings of the Waubesa Conference on Wetlands. Institute for Environmental Studies, University of Wisconsin, Madison, Wisconsin.

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

Heinselman, M. L. 1963. Forest sites, bog processes, and peatland types in the glacial Lake Agassiz region, Minnesota. Ecological Monographs 33:327-374.

Heinselman, M. L. 1970. Landscape evolution, peatland types, and the environment in the Lake Agassiz Peatlands Natural Area, Minnesota. Ecological Monographs 40:235-261.

Herdendorf, C. E. 1987. The Ecology of the Coastal Marshes of Western Lake Erie: A Community Profile. U. S. Fish and Wildlife Service Biological Report 85(7.9), Washington, D.C.

Herdendorf, C. E., C. N. Raphael, and E. Jaworski. 1986. The Ecology of Lake St. Clair Wetlands: A Community Profile. U. S. Fish and Wildlife Service Biological Report 85(7.7), Washington, D.C.

Johnston, C. A., N. E. Detenbeck, J. P. Bonde, and G. J. Niemi. 1988. Geographic information systems for cumulative impact assessment. Photogrammetric Engineering and Remote Sensing 54(11):1609-1615.

Keddy, P. A. and A. A. Reznicek. 1985. Vegetation dynamics, buried seeds, and water level fluctuations on the shorelines of the Great Lakes. *In* H. H. Prince and F. M. D'Itri (eds.), Coastal Wetlands. Lewis Publishers, Inc., Chelsea, Michigan.

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

McCullough, G. B. 1981. Wetland losses in Lake St. Clair and Lake Ontario. *In* A. Champagne (ed.), Proceedings of the Ontario Wetlands Conference. September 18/19, 1981. Federation of Ontario Naturalists. Don Mills, Ontario.

Mudroch, A. 1980. Biogeochemical investigation of Big Creek Marsh, Lake Erie, Ontario. International Association Great Lakes Research 6(4):338-347.

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.

National Research Council of the United States and The Royal Society of Canada. 1985. The Great Lakes Water Quality Agreement: An Evolving Instrument for Ecosystem Management. National Academy Press, Washington, D.C.

Novitzki, R. P. 1982. Hydrology of Wisconsin Wetlands. Information Circular 40. U.S. Department of the Interior, Geological Survey and Wisconsin Geological and Natural History Survey, University of Wisconsin-Extension, Madison, Wisconsin.

Preston, E. M. and B. L. Bedford. 1988. Evaluating cumulative effects on wetland functions: a conceptual overview and generic framework. Environmental Management 12(5):565-585.

Regier, H. A. and G. L. Baskerville. 1986. Sustainable redevelopment of regional ecosystems degraded by exploitive development. *In* W. C. Clark and R. E. Munn (eds.), Sustainable Development of the Biosphere. Cambridge University Press, New York.

Richardson, C. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science 228:1424-1426.

Steedman, R. J. and H. A. Regier. 1987. Ecosystem science for the Great Lakes: perspectives on degradative and rehabilitative transformations. Canadian Journal of Fisheries and Aquatic Sciences 44:95-103.

Stuckey, R. L. 1989. Western Lake Erie aquatic and wetland vascular-plant flora: its origin and change. *In* K. A. Krieger (ed.), Lake Erie and its Estuarine Systems: Issues, Reso[...]es, Status, and Management. NOAA Estuary-of-the-M[...] Seminar Series No. 14. Washington, D.C.

Trautman, M. B. 1977. The Ohio Country From 1750 to 19[...] A Naturalist's View. Ohio Biological Survey, Biolo[...] Notes No. 10. Ohio State University, Columbus, Ohio. 25[...]

U. S. Fish and Wildlife Service and Canadian Wildlife Ser[...] 1986. North American Waterfowl Management Plan. Un[...] States Department of the Interior and Environment Can[...]

U. S. Fish and Wildlife Service. 1988. North American Water[...] Management Plan: The Lower Great Lakes/St. Law[...] Basin Joint Venture. U. S. Fish and Wildlife Service, [...] Gateway Center, Newton Corner, MA 02158.

Weeks, J. L. 1975. Ohio Wetlands Inventory, 1972-74. [...] Department of Natural Resources, Division of Wildlif[...] service Document 17 (678). Oak Harbor, Ohio.

Weller, P. 1988. State of the Great Lakes Region Environ[...] Wetlands. Prepared for Environment Canada, Environ[...]tal Interpretation Division, Conservation and Prote[...] Ottawa, Ontario, Canada.

Whigham, D. F., C. Chittering, and B. Palmer. 1988. Impa[...] freshwater wetlands on water quality: a landscape per[...]tive. Environmental Management 12(5):663-671.

Whillans, T. H. 1982. Changes in marsh area along the Can[...] shore of Lake Ontario. Journal of Great Lakes Res[...] 8(3):570-577.

Winter, T. C. 1988. A conceptual framework for assessing c[...]lative impacts on the hydrology of nontidal wet[...] Environmental Management 12(5):605-620.

World Commission on Environment and Development. [...] Our common future. Oxford University Press, Oxford, [...] York.

DEFINING AND ANALYZING CUMULATIVE ENVIRONMENTAL IMPACTS

CHERYL K. CONTANT AND LYNA L. WIGGINS

Increasing recognition of the accumulative nature of many environmental problems has led the courts, regulators, and practicing analysts to seek a broader, clearer, and more comprehensive definition of cumulative impacts. By examining previous discussions of these impacts, we suggest that a cumulative impact analysis of individual projects must consider two categories of contextual issues: the relationship between a proposed project and other development activities, and the complex and often non-incremental effects of a development activity on many natural systems. A new, comprehensive analysis approach is proposed to reflect these categories and contexts. Critical elements of this approach include: an increased emphasis on improved monitoring of both environmental conditions and past development activities, and enhanced modeling of both development patterns and natural systems' responses. Finally, techniques to accomplish these tasks are discussed.

Introduction

Several recent environmental problems illustrate the cumulative nature of the impacts of human development activities. The *Global 2000 Report to the President* (U. S. Council on Environmental Quality 1980) cites global problems of diminished biodiversity, build-up of carbon dioxide, depletion of the stratospheric ozone layer, and acid rain as significant cumulative impact problems. Closer to home, many urban communities have become aware of the cumulative effect of increasing development on transportation congestion, air pollution, and the avail-

Address correspondence to: Prof. Cheryl K. Contant, Graduate Program in Urban and Regional Planning, The University of Iowa, Iowa City, IA 52242.

ability of infrastructure (such as water, sewerage systems, and landfill capacity). Even ex-urban and rural areas have seen cumulative consequences resulting from farmland destruction, soil erosion, and agricultural chemical contamination of groundwater supplies. Present assessment and management approaches apparently have failed to predict and control the cumulative nature of human development actions and their impacts (Beanlands et al. 1986; Hirsch 1988).

The significance and currency of these environmental problems is undeniable. In each case, individual human activities produce impacts that may be insignificantly small. When combined with other past, present, and future activity, however, these small impacts become cumulatively significant. Resolving many such problems, therefore, may be closely tied to an improved understanding, analysis, and control of the cumulative environmental impacts of single-project level activities (Sonntag et al. 1987). Yet, definitional, scientific, and administrative limitations have severely constrained the development of effective cumulative impact analyses at the project-level (Dickert and Tuttle 1985; Roots 1986). These concerns over defining, predicting, and controlling the cumulative consequences of development activities remain the focus of continuing investigation, debate, and analysis.

In this paper, we define a framework for cumulative impact analysis at the individual project level. In developing this framework, we begin with a careful study of the evolving, complex, and often muddy definition of cumulative impacts. We consider the role of regulatory language, interpretations by the courts, and studies by academics and practicing environmental analysts in adding further depth and complexity to the topic. Two categories emerge in our formulation of cumulative impacts: effects resulting from a project's relationship to other development activities, and effects produced by an activity's presence within a set of many natural systems. We conclude by proposing an approach for cumulative impact analyses of individual projects. Our approach emphasizes the importance of monitoring and modeling efforts for the two categories of cumulative impact issues.

Defining Cumulative Impacts

Over their relatively brief history of use, environmental analyses of proposed development activities have examined primarily the direct impacts of a single proposed action on a particular set of critical environmental components. Depending on the severity of the anticipated impacts, project modifications or other impact management techniques were then prescribed to lessen or eliminate negative environmental effects. This process of predicting and minimizing the consequences of a single action has not adequately considered the accumulative nature of some effects, the nonlinear responses of some natural systems, nor the linkages between a single action and other related activities (Vlachos 1985; Roots

cumulative impact analysis evaluates the consequences of multiple activities and sources of impact on a larger set of environmental components (Clark 1986).

Typically, these additional analysis requirements exceed the theoretical bases and practical capabilities of most individual project-level environmental assessments (Hirsch 1988, Stakhiv 1988). Along with providing accurate predictions of the direct consequences of an action on an environmental parameter, cumulative impact analyses require placing a proposed action and its impacts in the context of other existing or expected actions and existing or expected environmental conditions (Cowart 1986).

A recognition of these context-based considerations forms the foundation for defining cumulative impacts and their analysis. The courts and governmental regulators have been actively involved in defining the appropriate scale and scope of cumulative impact considerations. Similarly, practitioners and analysts have attempted to further develop the concept of cumulative impacts. Both sets of activities lead to an enriched definition of cumulative impacts and an improved understanding of the tasks needed for an effective and complete analysis.

Litigative and Regulatory History

Much of the early concern over cumulative impacts of development projects came from attempts by developers to segment large projects into smaller units (Merson and Eastman 1980). Project segmentation was designed to avoid the detailed assessment requirements imposed on these collective activities that were expected to produce more than minor impacts. This method of avoiding costly and time-consuming environmental analyses was challenged in the courts in the early 1970s. At issue in these cases was the appropriate timing for a detailed analysis and the scope of activities to be included in that analysis.

In *Scientists' Institute for Public Information, Inc. v. Atomic Energy Commission* (ELR 1973a), the court defined an "irretrievable commitment of resources" to an overall planned project as the critical test in determining whether a given action was sufficiently connected to subsequent contemplated actions to warrant a full, detailed analysis. In another case that same year, *Indian Lookout Alliance v. Volpe* (ELR 1973b), a different court concluded that if a proposed segment of a project had "independent significance or utility," then an analysis was required only on the single segment, not the entire project. The seeming contradiction between these two court-defined provisions led to a continuing struggle in the courts to identify the appropriate timing and scope of assessments for the interrelated, but potentially separable, parts of related development actions (for example, *Minnesota Public Interest Research Group v. Butz* and *Trout Unlimited v. Morton*).

In 1975, the Supreme Court clarified earlier rulings on the timing and scope of the impact analyses of segmented projects in the landmark case, *Kleppe v.*

projects need to be included in cumulative analyses, and that contemplated or anticipated projects need not be addressed. This finding suggested that similar development activities in a region need not be considered together in an impact analysis unless they had been formally proposed as a program or set of proposals. When a series of intimately connected *proposed* actions were identified, however, the court suggested (quite boldly) that an environmental region may be a more appropriate scale for a cumulative impact analysis than the smaller project area.

Strong reaction to this narrowed scope of analysis ensued from both the environmental community and the President's Council on Environmental Quality (CEQ). In response, CEQ promulgated more inclusive language in its 1978 NEPA regulations (U.S. Council on Environmental Quality 1978). Specifically, the regulations require that connected, similar, and cumulative actions be grouped together in an analysis. Included in the grouping of actions which would warrant a common analysis are those activities that: (1) are interdependent parts of a larger action, (2) automatically trigger other actions, (3) have cumulatively significant impacts when viewed with other proposed actions, or (4) are similar enough in time or geography to other reasonably foreseeable or proposed actions.

In addition to defining the appropriate scope of activities to be included in impact analyses, timing considerations are addressed, in part, through the "tiering" process. In most situations, an individual project proposal is part of a larger program of activities (Hapke 1985, p. 10289), that is, a "branch in a tree of planned actions." By tiering, analyses are performed early in the planning process on a set of related or connected actions. Subsequent analyses of specific project proposals are then reviewed in the context of these detailed environmental analyses.

In the context of cumulative impact analysis, this tiering mechanism establishes the legitimacy of grouping together geographically proximate or similar types of activities to examine their overall impact.[1] Through tiering, cumulative impacts are considered early in a program planning process and within the context of the larger branch of related activities (Barney 1981).

Analytic and Conceptual Development

Several researchers and practitioners in the past decade have suggested comprehensive approaches to considering cumulative impacts in project review. By defining typologies for cumulative impacts, these analysts highlight the key elements to be considered in their assessments. Most typologies focus on the processes by which human activities produce cumulative effects within a natural system. Included in many of these typologies are the natural systems processes of accumulation (Clark 1986), delayed response (Baskerville 1986), triggers and thresholds (Preston and Bedford 1988), nonlinear functional relationships (Bedford and Preston 1988), and synergism (Vlachos 1985).

Baskerville (1986) characterizes cumulative impacts into three groups that reflect his concern for the way natural systems respond to external pressures. His first group results from repeated "incremental insults to the system." Each increment adds to the previous increments over time. His second group is the situation where a single action or some limited set of actions results in a system change in structure or dynamic. Exposure to a cancer causing agent exemplifies this type of impact. The initial exposure incident might have appeared benign, but its introduction into the system created changes in its structure that produce significant effects much later in time.

The third group of cumulative impacts noted by Baskerville corresponds to the accumulation of impact by cycling over space and time. He cites the clear-cutting of forests as an example. Although the impact of one clear-cut area may be partially mitigated by natural processes of recovery, these new species may not be the proper ones. As the natural balance is shifted by clear-cutting and recovery over time throughout a forest, the impact actually moves through various cycles, migrates around the forest, and changes the overall nature of the forest in a cumulative manner.

Beanlands and others (1986) developed another typology of cumulative effects that focuses on the sources of such natural system impacts. In particular, they include: time-crowded perturbations, space-crowded perturbations, synergisms, indirect effects, and nibbling. They use the concept of crowding to refer to the effect resulting from the inability of a system to recover from an earlier or closer perturbation before a new one is present. Total effects that are qualitatively or quantitatively different from the sum of the effects of the individual disturbances are synergistic. Indirect effects are those impacts produced at some time or distance away from the initial perturbation, or by some complex pathway; nibbling is the impact resulting from the incremental insult of repeated actions on an area over time.

These earlier efforts are summarized by Sonntag et al. (1987) into a fourpart typology of cumulative impacts: (1) linear additive effects, the impact of incremental small additions along a linear cause-effect response relationship; (2) amplifying or exponential effects, the result of an incremental addition that produces a larger effect than earlier additions; (3) discontinuous effects, the impact resulting from exceeding a threshold or the crossing of a stability boundary; and (4) structural surprises, multimedia and multisystem impacts that may produce long-term changes in natural systems' responses to further perturbations.

Issues concerning the relationship between development actions and their cumulative impacts on the natural environment were raised in a series of research

[1]To facilitate the analysis of a particular action's effects in the context of other actions, new tools have been developed. These techniques include: programmatic assessments (such as the analysis of future offshore gas development or river-basin hydropower development), and regional development assessments (such as analyses performed for Lake Tahoe, for the Chesapeake Bay, or through Florida's development of regional impact process) In these analyses, a large number of projects over a defined time frame and spatial area is examined in total to determine their total impacts. Individual projects that are a part of the grouped projects, therefore, require limited cumulative analyses since the programmatic/regional assessment have already considered these impacts. These larger analyses, however, tend to be weak in predicting specific subsequent development actions and changes in rates of growth resulting from precedent-setting or growth-inducing activities.

projects investigating cumulative impact assessment in the Corps of Engineers regulatory program (Contant 1984; Stakhiv 1988). Stakhiv (1988) views cumulative effects as the result of two sets of processes: summation of significant effects and the integration of systemic effects and consequences. He, therefore, recommends assessment procedures that document impacts associated with changes in the impetus for growth (economic and social) and impacts that accumulate within and/or modify the structure of a natural system.

In the typology developed by Contant (1984), cumulative impacts are the result of additive and aggregative actions producing impacts that accumulate incrementally or synergistically over time and space. Additive actions refer to repeated similar activities, while aggregative actions correspond to groupings of dissimilar activities. Impacts of both sets of actions accumulate in an incremental (nibbling) fashion and/or combine synergistically (interactively) to produce effects other than those occurring directly or indirectly from the actions. In later work, Contant and Wiggins (1989) add "growth inducement" to reflect their observation that certain activities can alter the rate of development of new activities in an area. This growth-induced impact recognizes the precedent-setting effect of some activities in stimulating even greater development than previously anticipated.

Components in a Cumulative Impact Definition

Developing a common set of concepts to describe cumulative impacts has seemingly alluded the courts, regulators, and researchers, alike. Many analysts have focused on the forms of development activities that produce cumulative impacts; others have concentrated on the functions of natural systems and the ways that impacts accumulate, interact, or produce systemic changes. These two categories of issues, which reflect the two contexts into which a proposed action must be placed, are particularly useful in identifying the key components in a general definition of cumulative impacts.

The first set of issues documents the presence and influence of past and current development activities, as well as the expectation of future development. These other activities may be similar or different, connected or unconnected, to the proposed action. The interrelationships between and accumulation of development activities establishes the context for an individual proposed project, and defines the cumulative nature of development actions. Effects resulting from this cumulative nature of development actions may be largely ignored or underestimated in most impact assessments of individual projects.

The second broad category of issues included within our definition of cumulative impacts describes the cumulative nature of the effects of development actions. In particular, these components refer primarily to the natural system context of development activities. Cumulative changes to a natural system include the nibbling away of a resource base by repeated actions; the crowding of a

TABLE 1. Components in Cumulative Impact Considerations

Context of other development activities	Context of many natural systems
Incremental effects (resulting from unmonitored past and current development)	Unanticipated responses
	Crowding (time or space)
Future development actions	Systemic changes:
	Synergism
	Structural changes
Growth inducement	Cycling
Regional structural development changes	Interactions across systems

resource base (in time or space); unanticipated (or nonaccumulative) responses of a natural system; systemic changes including synergistic responses, structural changes, or cycling; and interactions across natural systems.

Taken together, these two general categories yield a much broader and comprehensive definition of cumulative impacts. Our typology suggests that cumulative impacts include two categories of effects that result from the contexts within which a proposed project is placed. Table 1 summarizes the components of cumulative impacts that result from the two contexts: other development activities and many natural systems.

By noting that a particular project is part of a larger set of anticipated or unanticipated development activities, our typology adds the effects of other activities to the definition of cumulative impacts. Simple, incremental impacts are considered cumulatively significant when placed in the context of other past and current development activities. Further, cumulative impacts result also from failing to include the effects of future development actions, unrecognized growth-altering changes, or shifts in regional development structures.

A recognition of the complexity of impact processes in natural systems adds many of the concerns noted by scientists who have identified that the simple addition of impacts over time and space does not account for all of a project's impacts. Many cumulative impacts result from the project's perturbation to a variety of natural systems. Effects that exceed a system's ability to recover, are unanticipated (that is, non-linear or discontinuous), cause structural changes within the system, or occur interactively across several systems are included as part of this category of impacts.

Elements in a Cumulative Impact Analysis

To be comprehensive, therefore, an approach for considering cumulative environmental impacts of individual project proposals must include mechanisms that capture the interrelationships of development activities and the complexities of natural systems' responses to perturbations. In this section, we propose a general approach to cumulative impact analysis that responds to these contextual issues and is built upon the tasks of monitoring and modeling. Although numerous approaches have been used or proposed for use,[2] none has explicitly recognized the effect of a project's contexts (other development actions and natural systems) on the analysis of cumulative impacts. This new approach, depicted in Figure 1, includes parallel sets of analysis activities for the two categories of cumulative impact considerations.

A primary task necessary for cumulative impact analysis is monitoring for both categories of impacts. To ensure the proper consideration of past development activities, monitoring identifies and tracks development actions by type, by location, and over time. Further, monitoring also includes the collection of data on a set of socioeconomic system parameters that describe factors affecting the nature and rate of development activity. Monitoring is also essential within the natural systems context. For the systems surrounding a proposed project, time series data are useful in identifying existing environmental conditions and in providing a database for understanding systems' responses, thresholds, and interactions.

The next major set of tasks requires modeling of both development patterns and natural systems' responses. Within the context of other development activities, data on past activities and socioeconomic system parameters are used to develop and calibrate regional land use development models. Outputs of these models provide forecasts of the type and nature of future development actions, yielding a more comprehensive picture of the incremental effect ("insult") of a project in relation to other past, present, and foreseeable future development. In addition, the enhanced land use modeling efforts can identify a particular project's effect in shifting the structure or type of regional development, or in changing existing rates of growth.

For natural systems, modeling efforts focus on understanding the responses of those systems when perturbed by development activities. "Crowding" can be examined by determining the recovery time (or space) needed for a particular system when perturbed by a development activity. Other, more complex, responses can also be modeled for the variety of natural systems affected. Included in these responses are those unanticipated effects resulting from exponential or discontinuous functional relationships, as well as system-wide changes such as time-delayed effects, cycling, or structural alterations. A final issue to be captured in these modeling efforts includes the responses resulting from interactions across natural systems. Models based upon ecosystems, rather than more narrowly

[2]For a complete discussion of several of these approaches and techniques, see Contant and Wiggins (1989). In addition, specific general approaches have been proposed by Stakhiv (1988), Sonntag et al. (1987), Peterson et al. (1987), and others.

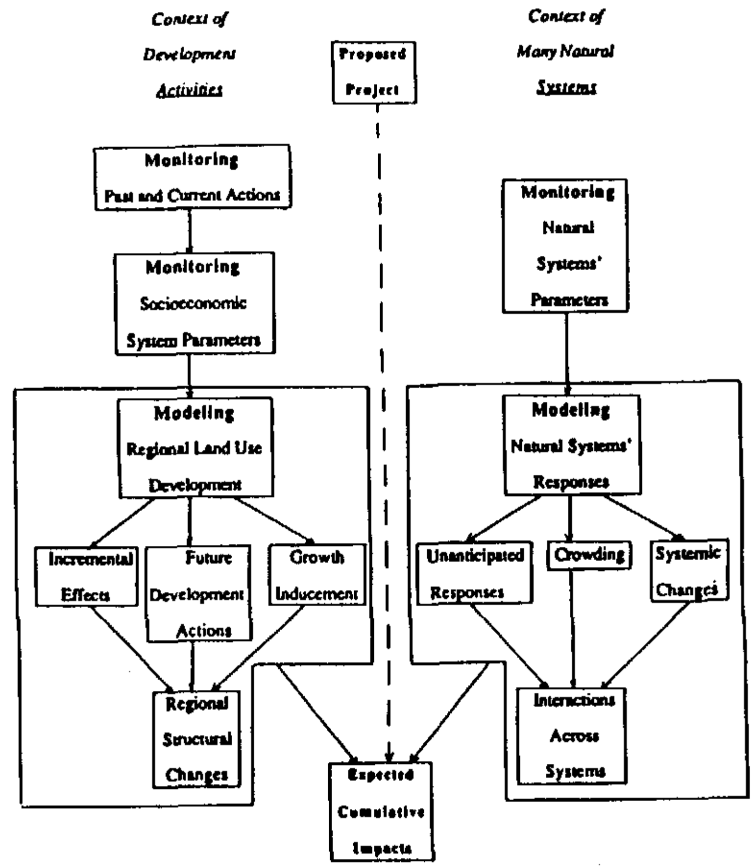


FIGURE 1. Approach for cumulative impact analysis

defined natural systems, can enhance capabilities in making these cross-system, cross-media impact predictions (Proett 1987; Hunsaker et al. 1989).

With this analysis approach, cumulative impact considerations can be more rigorously and more thoroughly included in individual project assessments. By understanding the importance of the context of a particular project, monitoring and modeling of both development activities and natural systems become essential elements in cumulative impact analysis. An emphasis on monitoring activities improves the capability of the analysis approach in describing existing conditions (for development activities and environmental systems) as a baseline for future comparisons and assessments. Expanding the scope of modeling to include more sophisticated methods enhances the consideration of cumulative impacts resulting from nonlinear, discontinuous, synergistic, or cross-media effects. These improvements should result in more comprehensive assessments and more thorough inclusion of cumulative impacts in project-level decisions.

Implications for Cumulative Impact Consideration

Incorporating this new assessment approach into project-level decisions will require significant improvements in existing administrative and managerial systems. Previous attempts at cumulative impact assessment have taxed the scientific and organizational capacity of most governmental or consulting offices (Contant 1984; Dickert and Tuttle 1985; Cowart 1986; Stakhiv 1988). These shortcomings should not lead to ignoring the problems of cumulative impact analysis; rather, limitations should be identified clearly and potential solutions identified.

A major limitation in most cumulative impact analyses is the lack of detailed monitoring information on previous development projects and several key environmental parameters (Dickert and Tuttle 1985). By identifying the need for this information on both sides of the parallel structure for cumulative impact analysis, new or improved information management systems will be needed. Fortunately, recent developments in geographic information systems (GIS) and remote sensing may provide the opportunity for improving the monitoring and tracking of project data and environmental systems' conditions (Johnston et al. 1988; Contant and Wiggins 1989; Hawkes et al. 1989).

Limitations resulting from the lack of scientific understanding of natural systems' phenomena further constrain the capabilities of most cumulative impact analyses. Vastly improved modeling efforts for natural systems are essential to *incorporate cumulative impact considerations in project level decisions*. Some of these improvements will require greater investments of resources; others will require a shift in the types of systems studied. For example, understanding wetland cumulative impacts will require greater expenditures for basic research and data collection (Preston and Bedford 1988). Additional research on understanding the interactions between several natural systems will necessitate analyses of a new level of systems, known as ecosystems. Present research on ecosystem models is producing promising, but admittedly limited, results (Hunsaker 1989; Hunsaker et al. 1989).

Similarly, more comprehensive modeling efforts are necessary in understanding and forecasting the complex processes in socioeconomic systems that produce land use development. This improved understanding would aid in predicting future development activities, in identifying growth-altering projects, and indicating changes in economic development pressures. Outputs of these models, however, must be provided at the appropriate scale, and indicate the various locational and spatial interactions that occur in regional development processes. Recent advancements in combining GIS and land use models may provide the data and scientific capabilities to make the required spatial land use and development forecasts (Densham and Goodchild 1989; Harris 1989).

A final set of limitations noted by most analysts reflects the inability of existing managerial systems to control expected cumulative impacts (Peterson et al. 1987; Hirsch 1988; Stakhiv 1988). In many cases, management of cumulative impacts relies on "yes/no" decisions about a project with modifications made to the original design to mitigate cumulative impacts. This limited form of impact management does not reflect the complex nature of the accumulation of impacts, nor the idiosyncrasies associated with the context within which the impacts occur.

Incorporating new management mechanisms for controlling expected effects is the critical final step to a complete discussion and control of cumulative impacts. Some suggested approaches have included an additional layer of review specifically for cumulative impact considerations (Peterson et al. 1987), greater use of programmatic impact assessments (Hapke 1985), or use of a "graduated scale" for both project reviews and modifications (Contant and Wiggins 1989).

At the heart of most of these suggested efforts is an attempt to resolve the mismatch that is often present between the level at which a cumulative impact occurs and the jurisdiction through which control efforts can be exercised (Beanlands et al. 1986). Under present conditions, even the most well-developed efforts to control cumulative impacts within a series of jurisdictions can be thwarted by inaction by a single entity within the impact area (Deakin 1986). Cumulative impacts that are felt at a regional scale (Stakhiv 1988) can only be addressed through planning processes directing development at that same scale (James et al. 1983; Sonntag et al. 1987). Therefore, adequate control of cumulative impacts requires regional planning and cooperation.

This call for regional planning is not as simple as one might think, however. Cowart (1986), in his discussion of Vermont's Act 250, suggests that a state land use program designed to control cumulative impacts is not enough. Proper planning processes are necessary to monitor development activities, define the relevant policy goals, determine appropriate management strategies, and adopt the proper control actions. Under these enhanced regional planning conditions, an enlightened and proper control of cumulative consequences is feasible.

Combining improved monitoring and enhanced modeling with more imaginative management can lead to more thorough and rigorous cumulative impact analysis at the project level. Added resources will be necessary to accomplish these improvements, but the resolution to some environmental problems may rest on our ability to identify and control the effects of individual projects with benign direct impacts, but cumulatively significant impacts. Recognizing the importance of projects' natural and developmental contexts is a significant first step in improving cumulative impact analysis.

References

Barney, P. E. 1981. The programmatic environmental impact statement and the national environmental policy act regulations. *Land and Water Law Review* 16(1):1–31.

Baskerville, G. 1986. Some scientific issues in cumulative environmental impact assessments. In *Cumulative Environmental Effects: A Binational Perspective*, G. E. Beanlands, W. J. Erckmann, G. H. Orians, J. O'Riordan, D. Policansky, M.H. Sadar, and B. Sadler (eds.). The Canadian Environmental Assessment Research Council and The United States National Research Council, Canada: Minister of Supply and Services.

Beanlands, G. E., Erckmann, W.J., Orians, G. H., O'Riordan, J., Policansky, D., Sadar, M. H., and Sadler, B. (eds.). 1986. *Cumulative Environmental Effects: A Binational Perspective*, The Canadian Environmental Assessment Research Council and The United States National Research Council, Canada: Minister of Supply and Services.

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

Clark, W. C. 1986. The cumulative impacts of human activities on the atmosphere. In *Cumulative Environmental Effects: A Binational Perspective*, G. E. Beanlands, W. J. Erckmann, G. H. Orians, J. O'Riordan, D. Policansky, M. H. Sadar, and B. Sadler (eds.). The Canadian Environmental Assessment Research Council and The United States National Research Council, Canada: Minister of Supply and Services.

Contant, C. K. 1984. *Cumulative Impact Assessment: Design and Evaluation of an Approach for the Corps Permit Program at the San Francisco District*. Ph.D. thesis. Stanford, CA: Stanford University.

Contant, C. K., and Wiggins, L. L. 1989. Toward defining and assessing cumulative impacts: Practical and theoretical considerations. Paper presented at the Scientific Challenges of NEPA: Future Directions Based on 20 Years of Experience Conference, Knoxville, TN.

Cowart, R. H. 1986. Vermont's Act 250 after 15 years: Can the permit system address cumulative impacts? *Environmental Impact Assessment Review* 6:135–144.

Deakin, E. 1986. *Mitigating Traffic from New Development: Six Issues*, Berkeley, CA: Institute of Transportation Studies.

Densham, P. J., and Goodchild, M. R. 1989. Spatial decision support systems: A research agenda. In *Proceedings to the GIS/LIS Conference*, Orlando, FL, vol. 2, pp.707–716.

Dickert, T. G., and Tuttle, A. E. 1985. Cumulative impact assessment in environmental planning: A coastal wetland watershed example. *Environmental Impact Assessment Review* 5:37–64.

Environmental Law Reporter (ELR). 1973a. *Scientists' Institute for Public Information, Inc. v. Atomic Energy Commission*. Vol. 3, p. 20525.

Environmental Law Reporter (ELR). 1973b. *Indian Lookout Alliance v. Volpe*. Vol.3, p. 20739.

Environmental Law Reporter (ELR). 1976. *Kleppe v. Sierra Club*. Vol.6, p. 20532.

Hapke, P. 1985. Thomas v. Peterson: The Ninth Circuit breathes new life into CEQ's cumulative and connected actions regulations. *Environmental Law Reporter* 15:10289–10297.

Harris, B. 1989. Beyond geographic information systems: Computers and the planning professional. *Journal of the American Planning Association* 55:85–90.

Hawkes, C. L., Beattie, J. B., Cain, S. L., Culver, S. R., Jenkins, P. L., Johnson, T. L., Reynolds, M. T., Vance-Miller, D. L., West, R. L., and Wynn, S. L. 1989. Quantitative cumulative impact analysis, resource protection goals, and a geographic information system used to evaluate impacts of places and lode mining in three Alaska national parks. Paper presented at The Scientific Challenges of NEPA: Future Directions Based on 20 Years of Experience Conference, Knoxville, TN.

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

Hunsaker, C. T. 1989. Ecosystem assessment methods for cumulative effects at regional and global scales. Paper presented at The Scientific Challenges of NEPA: Future Directions Based on 20 Years of Experience Conference, Knoxville, TN.

Hunsaker, C. T., Graham, R. L., Suter II, G. W., O'Neill, R. V., Jackson, B. L., Barnthouse, L. W. 1989. *Regional Ecological Risk Assessment: Theory and Demonstration*, Oak Ridge National Laboratory, Publication No. 3273, March.

James, T. E., Ballard, S. C., and Devine, M. D. 1983. Regional environmental assessments for policy making and research and development planning. *Environmental Impact Assessment Review* 4(1):9–24.

Johnston, C. A., Detenbeck, N. E., Bonde, J. P., and Niemi, G. J. 1988. Geographic information systems for cumulative impact assessment. *Photogrammetric Engineering and Remote Sensing* 54:1609–1615.

Merson, A. and Eastman, K. 1980. Cumulative impact assessment of western energy development: Will it happen? *University of Colorado Law Review* 51:551–586.

Peterson, E. B., Chan, Y. -H., Peterson, N. M., Constable, G. A., Caton, R. B., Davis, C. S., Wallace, R. R., and Yarranton, G. A. 1987. *Cumulative Effects Assessments in Canada: An Agenda for Action and Research*. Canadian Environmental Assessment Research Council, Canada: Minister of Supply and Services.

Preston, E. M., and Bedford, B. L. 1988. Evaluating cumulative effects on wetlands functions: A conceptual overview and generic framework. *Environmental Management* 12(5): 565–583.

Proett, M. A. 1987. Cumulative impacts of hydroelectric development: Beyond the cluster impact assessment procedure. *Harvard Environmental Law Review* 11: 77–146.

Roots, E. F. 1986. Closing remarks: A current assessment of cumulative assessment. In *Cumulative Environmental Effects: A Binational Perspective*, G. E. Beanlands, W. J. Erckman, G. H. Orians, J. O'Riordan, D. Policansky, M. H. Sadar, and B. Sadler (eds.). The Canadian Environmental Assessment Research Council and The United States National Research Council, Canada: Minister of Supply and Services.

Sonntag, N. C., Everett, R. R., Rattie, L. P., Colnett, D. L., Wolf, C. P., Truett, J. C., Dorcey, A. H. J., and Holling, C. S. 1987. *Cumulative Effects Assessment: A Context for Further Research and Development*. Canadian Environmental Assessment Research Council, Canada: Minister of Supply and Services.

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

U.S. Council on Environmental Quality. 1978. National Environmental Policy Act—Final regulations. *Federal Register* 43(230): 55978–56007.

U.S. Council on Environmental Quality. 1980. *The Global 2000 Report to the President*. Washington, DC: Government Printing Office.

Vlachos, E. 1985. Assessing Long-Range Cumulative Impacts. In *Environmental Impact Assessment, Technology Assessment, and Risk Analysis*, V. T. Covello (ed.). Berlin: Springer-Verlag.

Chapter III

Integrated Coastal Management

DEVELOPING A SUSTAINABLE VISION

Most indicators suggest that, throughout the country, human impact upon urban coastal areas continues at a level of severity that threatens the biological integrity of many marine systems and seriously impairs their capacity to produce a full range of goods and services valued by people.

Given the importance of coastal areas to society, managing their social and economic uses in a sustainable fashion should be a central tenet of government policy. In the absence of controls, coastal resources are unpriced and widely accessible, and the market fails to reveal social values or restrain use. There is a need for a complete system of resource valuation capable of identifying the consequences and opportunity costs associated with various patterns of resource use. Unfortunately, such a comprehensive tool does not exist. Some work has been done, particularly in the area of human uses of water resources, but it is significantly incomplete, especially with respect to the full range of human values, the consequences of health effects, and ecological values. Thus, there is no comprehensive set of economic tools capable of developing and implementing an optimal coastal management strategy that meets society's goals.

In the absence of such a capability, reversing trends of degradation in an effective and efficient manner is more difficult. It requires, at the least, that society formulate a clear vision of the coast's future and identify the tasks necessary to achieve that vision. The concept of integrated coastal management (ICM) is a starting point for that vision. This concept is associated here with two general objectives: 1 restore and maintain the integrity of coastal ecosystems, and 2) to maintain important human values ar uses associated with those resources.

Recently, the Environmental Protection Agency, through its Science Advisory Board, explored problem of making its management programs more relevant to ecological imperatives in *Reducing Risk: Setting Priorities and Strategies for Environmental Protection* (EPA 1990). In a substantial sense the methodology for integrated coastal management set forth in this report is a further step in the general direction proposed in the EPA report.

OBJECTIVES OF INTEGRATED COASTAL MANAGEMENT

- To restore and maintain the integrity of coastal ecosystems, and
- To maintain important human values and uses associated with those resources.

PRINCIPLES AND METHODOLOGY FOR A SYSTEM OF INTEGRATED COASTAL MANAGEME

Important changes in the two decades since the passage of the Clean Water Act suggest that integrated coastal management is achievable. Two developments, one social and the other technical, a: worth highlighting:

- Today, a far greater proportion of the public cares about the environment in general and coastal areas in particular than 20 years ago. This awareness is a potentially powerful political force tl can be effective in driving admittedly complex processes to useful conclusions.

- Additionally, important technical progress has been made over the past 20 years in source reduction, treatment systems, outfalls, and modeling of systems. Although scientific knowledge about coastal processes is far from complete, it is far advanced over that of a generation ago. Beyond mere (

scientists and engineers also understand in a more sophisticated way the nature of how coastal processes operate. Finally, modern computing and other data management tools have given scientists and others for the first time the capacity to organize, analyze, and display this complex of information in a way which is accessible to technical analysts and lay persons alike.

Integrated coastal management is an ecologically-based approach to environmental management and is therefore in many ways a departure from the technology-driven strategy which has characterized the national effort since the FWPCA of 1972, especially in urban areas where a major emphasis has been appropriately on the construction of sewage treatment plants. In fact, to a certain extent ICM is analogous to the water-quality-based efforts which preceded the 1972 amendments and which were thought to have been largely ineffectual. Two decades have passed and it is now time to reconsider the validity of an ecologically-based strategy especially in highly complex situations. As noted above, the public attention to environmental issues and the scientific understanding is vastly more pronounced than was the case in the 1960s, and therefore the basic likelihood of environmental action is enhanced. In addition, there is now a well developed regulatory system, with permits, inspection and enforcement procedures, capable of assuring that identified obligations are, in fact, met. This system is now being complemented by the development of an economically-based incentive system which may add even greater force to the compliance imperative. Perhaps most critically, it is now necessary to assure that scarce public economic resources are spent on those management options that will have the maximum positive impact. Society can no longer afford to spend large amounts of money solving unimportant problems.

The following discussion of integrated coastal management sets forth a set of interlocking management principles and a related process. The fundamental objective of this system is to allow for improved identification of important priorities and better allocation of resources to the identified problems. An integrated system is interlocking and iterative, that is, each of the principles builds on and influences others. The following discussion identifies these six principles and suggests how they can be applied in a coherent and logical fashion.

Principles

The following principles should underlie the development of an integrated coastal management system and the institutions necessary to implement it:

1. Coastal management's overall objective is to maintain ecological processes and meet human needs for goods and services. Accordingly, management actions need to be developed on the basis of tl best science available about ecological functions as well as a comprehensive understanding of human ne and expectations, which are both tangible and intangible.

2. Management objectives should be expressed as water and sediment quality, and other environmentally based goals. Using environmentally based goals allows flexibility in the methods used achieve those goals, while assuring that ecosystems are protected at the desired level.

3. Comparative assessment of risk, and available management concepts should drive the selecti of management options. This approach will assure that a rational basis is used for focusing action and resources on the most important problems. A dynamic and iterative planning process should integrate risk-based analysis with evolving scientific understanding and human expectations. The process is a continuing one and allows for incremental decision making when that is the prudent course. On the ot hand, it also provides a context for making the high cost or risk decision most likely to be correct in th face of scientific uncertainty. This integration requires a stable institutional base for the planning proc

4. A transdisciplinary perspective is critical to coastal problem solving. While specific discipli such as chemistry, biology, and hydrology and their integration through environmental engineering principles will always be central to this transdisciplinary perspective, increasingly fields such as economi law, and sociology are important components of marine management. Maintaining systems for routine exchange and analysis among professions is essential to help managers gain a comprehensive understan

PRINCIPLES OF INTEGRATED COASTAL MANAGEMENT

- Management actions are based on the best available scientific information about ecological functions, as well as a comprehensive understanding of human needs and expectations which are both tangible and intangible.
- Management objectives are expressed as water and sediment quality and other environmentally-based goals.
- Comparative assessment of risk, and available management concepts drive the selection of management options.
- A transdisciplinary perspective is critical to coastal problem solving.
- The ICM process functions in a context that is responsive to scientific uncertainty about the functions of coastal systems.
- ICM is driven by both science and engineering, and public expectations.

of coastal problems. With a linked analytical system, disciplines could assist more effectively in formulating a transdisciplinary management response. Such a system should integrate scientific, engineering, and other information into management practice in a timely and flexible fashion.

5. This system should function in a context that is responsive to scientific uncertainty about the functions of coastal ecosystems. Accordingly, it is based on the premise that how these natural systems respond to stresses from pollution events, overfishing, sedimentation, or encroachment on habitat is not often well understood or subject to human intervention. Management actions instead must recognize that it is the human activities at the source of these stresses that must be reduced or changed. Managing from this perspective elevates coastal preservation concerns into day-to-day public policy considerations such as

how to meet housing needs, transportation needs, and other sustenance needs while protecting the environmental quality of the coasts.

6. Integrated coastal management is driven both by science and engineering, and public expectations. Especially important, therefore is the requirement that there be significant opportuniti public participation and involvement throughout the process. Appropriate public values are not onl economic or recreational, but also those arising out of an ethical and aesthetic concerns for protectio the environment.

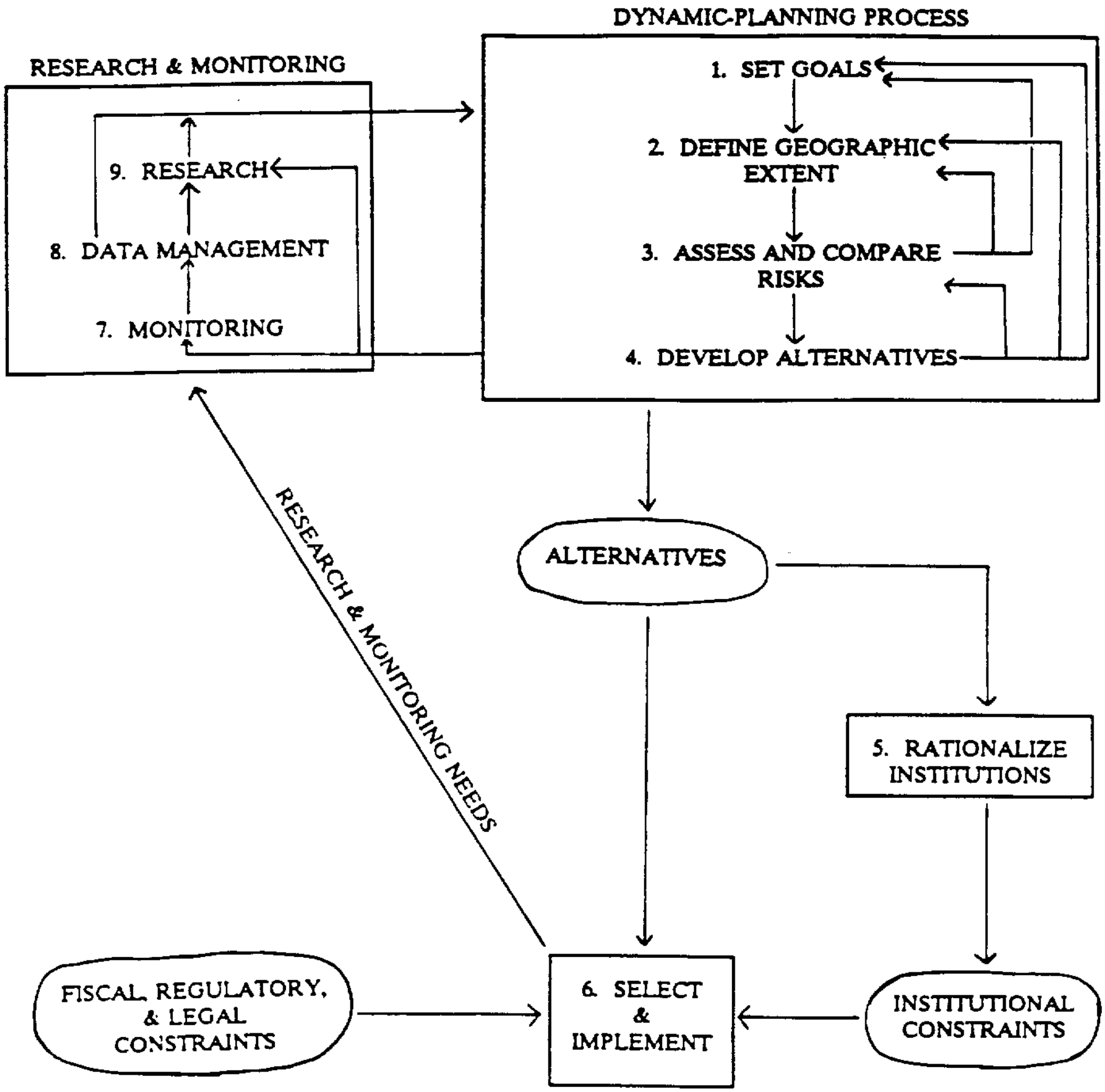
Process

The process of integrated coastal management is composed of the following elements which applied in the sequence they are discussed. Elements at the bottom of the list are not less importan earlier ones and should feed back into the former. The flow chart in Figure III-1 gives a simple sche view of the relationship among the elements.

1. Set Goals. At the outset of the development of a management plan, it is important to de a well-informed understanding of the goals and expectations held for a coastal region and the range environmental problem areas that require further attention. Management has two primary objective protect the fundamental functions and biological richness of the ecosystem, and to maintain importa human uses. The starting point of integrated management is to identify the problems threatening th goals. Two important tasks must be completed to meet these objectives: 1) define critical (importa environmental processes in time and space using existing information and data; and 2) define and rationalize the variety of human expectations about uses and benefits to be derived from the coast.

Environmental Processes. All coastal systems are not the same. There will be variations in t functions of natural processes. In a particular system the important elements of these processes mu identified with sufficient precision so that management can protect them. The definition of processe

1 Figure III-1 Process of Integrated Coastal Management



issues inevitably leads to a system for setting priorities, forcing decisions about issues that are more important to address than others in order to maintain ecological and human health.

Human Expectations. Societal values and needs with respect to a particular coastal system als vary from region to region. Often there are conflicts among various interests, and changes in values a needs will occur over time. Initial management steps must seek to understand the existing range of expectations. This understanding will further contribute to an appreciation of those elements of the natural systems that are important for protection

Once important environmental processes and human expectations have been identified, the anthropogenic conditions that threaten their maintenance can be determined.

2. Define Geographic Extent of Concerns. It is important to address environmental problem the scale on which they occur. Thus, integrated coastal management must be based on adoption of a relevant environmental domain with appropriate aquatic, terrestrial, and atmospheric components. T starting point for defining the geographic extent of an issue often will be the coastal area of concern, or all of its watershed, and other areas as dictated by important related marine and terrestrial process For example, concerns about pelagic fisheries likely would suggest concern over a large area. The pr of defining the geographic extent of an issue should also take into account sources of problems and c demands on the system, such as for the products produced in the coastal area, which create stress.

Only rarely will there be a perfect coincidence between either the variety of ecological domai the various sources of degradation, or the boundaries of administrative jurisdictions. However, approximations of such areas, both as a means to facilitate identification and location of the most important environmental processes and most immediate or significant sources of degradation, can be developed at a scale which is consistent with appropriate management actions. An overall goal in adopting relevant environmental domains is to minimize the number of significant causes and effects taking place outside the domain and maximize the effectiveness of management measures that can be taken within the domain.

In many situations the result of this analysis may produce both a variety of environmental domains and parallel sets of variable jurisdictional boundaries. As the problem analysis progresses, decisions can be made about the most logical way in which to draw areas for specific management and concern as a function of knowing what problems are important and what management options can be used to address them.

3. Assess and Compare Risks. An assessment and comparison of risks to ecological systems and human health across the full spectrum suggested above should be completed before management options are selected. The risk comparison should guide decisions for setting priorities. Ideally, risk management decisions should place the burden of control on activities that may significantly harm humans or the environment. While risk assessment methodologies are not fully developed and comparisons of human risks with ecological ones are difficult to make, even a qualitative examination driven by the goals set for coastal management will substantially improve the priority setting process. The continued refinement of risk assessment and comparison methods using the basic concepts of dose, exposure, and hazard is crucial to integrated management.

When an integrated understanding of environmental degradation and deterioration in human use has been developed, choices for priority attention can be made on the basis of the relative importance of issues within the total complex of problems. In addition to deciding which problems are important to solve, this analysis should also include a component that attempts to define an understanding of what level of protection or management is required in order to meet established goals.

An integrated understanding of the sources of degradation provides the basis for choosing those control options that will yield the greatest net benefits. A comprehensive display of the relative risks and their causes is the essential ingredient to a strategic approach to management. It is also critical to decisions about allocating societal resources to those problems that are of most immediate importanc

The science of natural systems often will be understood imperfectly. A comparative risk analy can contribute to maximizing the probability that more important problems will be addressed first. However, uncertainty may occasionally be so significant as to not allow for a clearly rational choice; a social values may appropriately demand that preventive action be taken. Where such value laden cho are made, they are likely to be better informed if made within the context of strategic assessments of comparative risks and with input from the public who ultimately provide the necessary resources.

4. Develop and Compare Alternatives for Risk Management. Coastal problems cannot be managed successfully as separate issues, such as pollution or wetland loss or fisheries depletion. The at least two reasons for the need to integrate such pressing issues. In the first instance, apparently separate issues are interrelated. Thus, fisheries declines could be due to overfishing, pollution, or fai reproduction due to loss of tidal wetlands. Secondly, resources are scarce. In the foregoing case of fisheries loss, it would be ideal to define the most likely cause of the loss and then design manageme options addressing that specific problem. Thus, risk management strategies must be devised to addre most important elements in the complex structure of problems in an integrated fashion--one that ass important sources of risk and achievable management alternatives.

Risk management decisions should be made through the consideration of priority problems i light of available management options, resources, administrative and legal structures, and other facto relevant to solutions. Consideration must also be given to the tradeoff between expenditures and be both with respect to different solutions to the same problem and solving different problems.

Not all problems need to be addressed using the same kinds of strategies. While a standards based regulatory system may be essential to providing a basic framework of governance, other techni such as economic incentives, streamlined management, land use policy, and education will be appropriate for particular problems.

Too often society perceives that the problem of a coastal environment is singular or one-dimensional. Thus, enormous efforts are made to improve a sewage treatment plant or to eliminate the presence of a particular chemical, yet problems remain. As suggested above, more frequently, degradation in resources is due to the combined effects of a number of human actions or to a condition which is not the most obvious. While correcting one problem might mitigate environmental harm for a brief period of time, long-term protection and restoration may not be accomplished. This situation does not mean that all problems need to be addressed with equal vigor simultaneously. It does mean that choices must be made about which problems to solve in a context that considers the multiplicity of causes and effects and maximizes the expenditure of resources on those issues which are important.

These four steps described together constitute the Dynamic Planning Process. This process whereby values, ecological processes, comparative risks, and strategies are developed and assessed must be considered as a dynamic and continuing planning process. Such a process is needed to capture interactions where one action may lead to another, to recognize new problems, to respond to new knowledge, and to recognize and correct mistakes. This process is an iterative exercise in which choices are made about how to anticipate and resolve conflicts and set priorities among multiple uses before environmental harm is done. In order to give reality to the planning process, it must be tied to some system of allocating uses in the coastal environment. These systems can include a broad spectrum ranging from land use controls, to regulatory systems with permits, to protected area programs such as marine sanctuaries.

The dynamic nature of the planning process is the core concept that allows for feedback between the various elements of the methodology. At one scale there are a series of interactions among the planning process, the conduct of scientific activity and the establishment of implementation programs. Within the planning process itself there are numerous iterative steps between the various functions.

Perhaps the strongest connections exist between the comparative risk assessment and comparative risk management activities.

5. Rationalizing Institutions. Institutions and mechanisms of governance must be rationalize a fashion which is capable of meeting the demands that an ecosystem-based management initiative wil dictate. Ideally, one might strive to create a single management institution responsible for all aspects management within an entire coastal ecosystem. In a world with existing institutions, government boundaries and variable ecological boundaries, however, the development of an ideal institution will r be possible. However, an improved system of coordination, and communication among institutions ca far to remedy the current, highly fragmented governance structure.

It also may be necessary to create new institutions and structures to meet certain managemen needs. For example, a significant land use management objective may only be able to be implemente with a change in the relation between local and state governments. While theoretically achievable, if change is politically impossible, in fact, it imposes a constraint on management options.

6. Selection and Implementation. Once alternative management options have been develope choices must be made about the alternative to be implemented. Inevitably the choices result from a political decision making process that may involve executive and legislative authorities among several jurisdictions, and entities having different missions and levels of authority. This political process is a integral part of coastal management. In the first instance, fiscal, regulatory, or institutional realities pose constraints that make certain options impossible to achieve. Inflexibility in environmental laws regulations make it difficult to implement an ICM approach effectively. The existence of such constr will force reconsideration of management options. In addition, failures to use scientific and related information may result in insufficient attention to the coastal problems so that hard choices are avoid Finally, if public expectations and values are ignored it may be difficult to implement recommended alternatives.

FOR REVIEW ONLY--DO NOT CITE, QUOTE, OR REPRODUCE

7. Monitoring. Integrated coastal management must include a comprehensive monitoring effort that focuses on factors of significant ecological, human health and resource use importance, or the processes that are crucial to them, and the control measures which have been put in place. In general, such a monitoring system will not only measure the status of water quality from a chemical and physical sense but will also take the pulse of the biologic regime. The results of monitoring function as a feedback mechanism to modify management actions, direct new research, and provide information for public accountability. The products of monitoring are the essential glue that allows integrated coastal management to take advantage of the new factors of public concern about and the technical capacity to understand environmental problems.

8. Information Management. Monitoring and research develop the data that drives an integrated data management system. Integrated coastal management can only be accomplished if monitoring and other data for environmental systems are managed in a way that allows managers and other interested parties to appreciate and make decisions about the whole. As noted, in the last decade, for the first time, significant capacity has been developed which permits a much greater degree of comprehensive understanding. This new capacity consists of three essential elements: significant increases in raw data; enhanced models for analyzing this data; and powerful hardware and software to manage the data. While data and models will never be complete or perfect, the increased capacity to manage them does provide an ability to integrate available data in a fashion that substantially enhances the prospect of integrated management. It is this information that helps to continually refine the understanding of the nature of particular coastal problems, expands the capacity to carry out comparative risk assessment and management, and ultimately allows for the development of new options for implementation when needed. Finally, it provides the picture as to whether efforts have succeeded or failed.

9. Research. An ongoing research program to continually refine the capacity to carry out the various elements of the dynamic planning process is essential. Attempting to carry out each step of the

process will suggest questions to which there are only incomplete answers. Furthermore, research coup with monitoring will develop new information that might suggest additional questions. The system mu: foster the formulation of these issues and the making of decisions to provide for appropriate research efforts. The need for more research need not become an excuse for doing nothing. To the contrary, display of problems, comparative risks, and management options with an understanding of uncertainty (allow decisions to proceed where problems are severe.

CONCLUSION

Integrated coastal management is a rigorous and difficult process. It is needed for situations tl are scientifically or governmentally complex, costly, risky or otherwise fraught with a degree of uncertai Accordingly, it need not be used for those problems which, upon initial examination, present a relativel simple solution.

While integrated coastal management may be useful most often in complex ecological systems extend far beyond the ~~limits~~ of an urban area, it is also a useful analytical and management methodolo when decision-makers are faced with problems having a predominantly urban theme. In urban areas, sources of human perturbation of the marine environment and their effects are often highly complex. Urban areas also rarely affect adjacent marine resources in splendid isolation from events in freshwatei watersheds or the distant ocean. Finally, public resources are always scarce and must be allocated to correcting those problems having the highest likelihood of important environmental benefits. Integrat coastal management provides a context for considering all of these complexities and then deciding wha important to be done in the urban setting.

Integrated coastal management is an iterative process. As discussed, there is continual feedbac among the various components of the methodology. Equally importantly, the entire process can be us for a particular situation with an increasing level of precision over time. For example, a quick analysis

FOR REVIEW ONLY--DO NOT CITE, QUOTE, OR REPRODUCE might be carried out in a matter of days using existing information. The result of such a rapid exercise can set up the dimensions of a more protracted process by identifying gaps and problems that require focus.

For the elected official this apparently complicated process should have utility because it will produce clearer choices which allow, and may even force, the political process to allocate resources to the most important problems. In essence, the process allows the political decision-maker to strike a balance between the expectations of various publics with respect to the facts as presented by technical professionals and reach a conclusion about implementing achievable management options. While those responsible for political choice and implementation need not necessarily understand, or even participate in, every aspect of the science or planning of integrated coastal management, it is a process that produces rational choices and allows for their refinement and modification over time.

REFERENCES

U.S. Environmental Protection Agency (EPA). 1990. Reducing Risk: Setting Priorities and Strategies for Environmental Protection. Washington, D.C.: Science Advisory Board (A-101). SAB-EC-90-021.

INTEGRATED COASTAL MANAGEMENT BIBLIOGRAPHY

Born, S.M., and A.H. Miller. 1988. Assessing Networked Coastal Zone Management Programs. Coastal Management 16:229-243.

Caldwell, L.K. Perspectives on Ecosystem Management for the Great Lakes. **Cite needed.**

Christie, W.J., et al. 1986. Managing the Great Lakes Basin as a Home. Journal of Great Lakes Research 12(1):2-17.

Colborn, T.E., et al **(need other authors)**. 1990. Great Lakes, Great Legacy? Washington, D.C.: The Conservation Foundation and the Institute of Research on Public Policy.

Derthick, M. 1974. Between State and Nation. Washington, D.C.: The Brookings Institution.

Donahue, M.J. 1986. Institutional Arrangements for Great Lakes Management: Past Practices and Future Alternatives. Lansing, Michigan: Michigan Sea Grant.

Donahue, M.J. 1988. The Institutional Ecosystem for Great Lakes Management: Elements and Interrelationships. The Environmental Professional 10:98-102.

Eichbaum, W.M., and B.B. Bernstein. 1990. Current Issues in Environmental Management: A Case Study of Southern California's Marine Monitoring System. Coastal Management 18:433-445.

Haigh, N., and F. Irwin. 1990. Integrated Pollution Control in Europe and North America. Washington D.C.: The Conservation Foundation and The Institute for European Environmental Policy.

Hansen, N.R., et al (need other authors). 1988. Controlling Nonpoint-Source Water Pollution - A Citizen's Handbook. Washington, D.C.: The Conservation Foundation and The National Audubon Society.

King, L.R., and S.G. Olson. 1988. Coastal State Capacity for Marine Resources Management. Coastal Management 16:305-318.

19. THE REGULATION AND MANAGEMENT OF BOTTOMLAND HARDWOOD FOREST WETLANDS: IMPLICATIONS OF THE EPA-SPONSORED WORKSHOPS

James G. Gosselink
Marine Sciences Department, Louisiana State University
Baton Rouge, LA 70803

Lyndon C. Lee
L.C. Lee & Associates, Inc.
3016 W. Elmore Street, Seattle WA 98199

Thomas A. Muir
Division of Endangered Species and Habitat Conservation
U.S. Fish and Wildlife Service
Washington, D.C. 20240

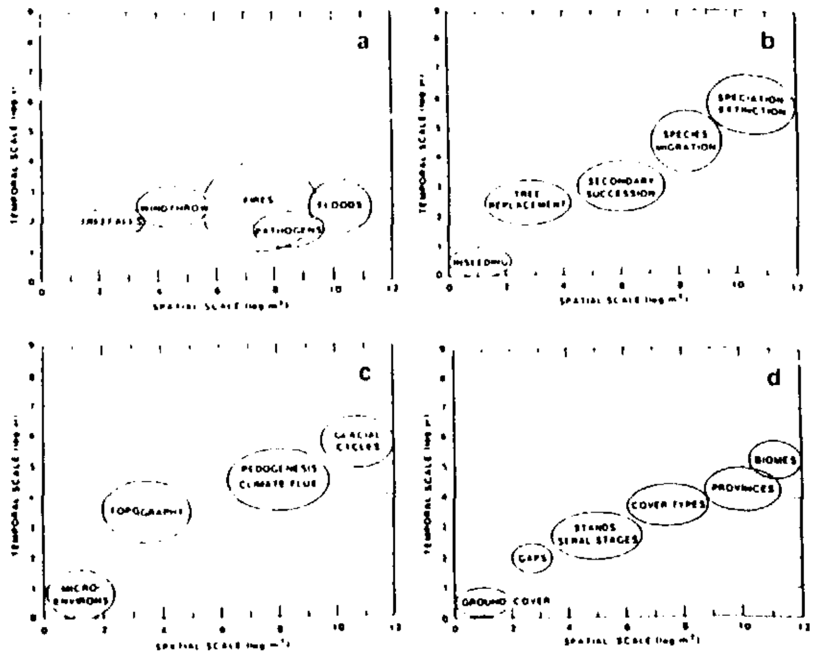


Figure 3. (a) Disturbance regimes, (b) forest processes, (c) environmental constraints, and (d) vegetation patterns, viewed in the context of space-time domains (from Urban et al. 1987, page 120; Copyright © 1987 by the American Institute of Biological Sciences, reprinted with permission).

functional unit. Typically, ecosystems ecology deals with such concepts as external, physical, forcing functions; nutrient cycling; trophic structure; and community diversity. These concepts have little meaning at the lower level of the hierarchy. Bottomland forest ecosystem characterization enables one to estimate the "cascade" effect of human activities; for example, to trace the impact of a hydrologic modification to the forest vegetation and from there through the forest's trophic levels.

A final level of bottomland forest systems organization is that of the landscape. While there is considerable overlap between concepts concerning ecosystems and landscapes (see Gosselink et al. Chapter 17), the latter term is usually reserved for large heterogeneous areas composed of several ecosystems that are spatially and temporally linked and that function as an integrated unit. The importance of spatial patterns in landscapes is explicitly recognized. Landscapes have their own properties, related to large areas and to long time frames. For example, a river is a complex system of many parts that function together as an integrated whole. Headwaters, upland slopes, floodplains, terraces, and river channels are all spatially and structurally integrated and interrelated. River systems cannot be fully understood by a study of these individual parts; their interrelationship is a fundamental property of the system.

The concept of a spatial and temporal hierarchy of bottomland forest systems organization has a number of implications that influence our understanding of these systems and the way we manage them.

1) Management of individual processes or species generally ignores the integrated nature of bottomland hardwood forest systems. As a result, related processes are ignored, and the resulting management actions often have unforeseen and undesirable side effects. For example, flood control projects are too often designed to minimize on-site flooding, with little regard to impacts on water quality and biota, or even on resulting downstream flooding and/or secondary effects to the biological integrity of downstream reaches. *Since bottomland flooding regimes determine the* quality of most of the valued services of bottomland forest systems, such a focus maximizes flood control at the expense of other services.

There has been considerable effort devoted to the development of evaluation systems for rating bottomland hardwood forests and other wetland ecosystems. Evaluation protocols such as the Habitat Evaluation Procedure (HEP, U.S. Fish and Wildlife Service 1980), the Golet Evaluation System (Golet and Larson 1974) and the Wetland Evaluation Technique (WET, Adamus et al. 1987) were developed with the idea of rating the services of a specific wetland site, or of comparing the relative value of services provided by alternative sites. They all focus on the first level of bottomland forest systems organization, site-specific processes.

If models are correctly bounded, the WET and Golet procedures deals qualitatively with second level (ecosystem) processes such as primary production, flood reduction, and water quality protection. No current rating procedure adequately addresses third level (landscape) processes. Indeed the context value of a specific site (that is, its context in the landscape) can only be evaluated if that context is assumed to persist through time. This presumption is unwarranted in the absense of an *implemented natural resource management plan for the entire watershed* landscape. For example, the value of a riparian site for flood abatement changes if other flood detention areas are isolated from the river by levees, if a dam is constructed across the river upstream, or if land clearing increases runoff.

2) Bottomland hardwood forest systems operate as integrated functional units. This statement is related to the previous point. Rather than managing to maximize one service at the expense of other services, evaluation and management procedures should recognize the integrated functioning of bottomland hardwood forest systems, and manage to optimize for the greatest combined value of *all* services. At a minimum, management decisions should be based on an explicit recognition of the balance between gains in one service and the loss of others. An excellent example of the *integrated functioning* of bottomland hardwood forest systems is the functional role of the elevation gradient found at many sites. As elevation changes across a floodplain, flooding frequency, depth, and duration also change. As a result, soil characteristics and vegetation patterns also change in relatively predictable ways (see Chapters 12 and 14). Biotic and geochemical processes related to soil saturation, inundation, and accessibility (primary productivity, nutrient recycling, decomposition, fish reproduction) also change predictably across this gradient (Gosselink et al. 1981). One issue discussed at length in the first workshop was whether classifying bottomland hardwood forest sites according to flooding (and related vegetation and soil) zones was a useful management devise (see Clark and Benforado 1981). Despite considerable evidence that ecological processes change predictably across the gradient, and despite consensus about the utility of the zonal concept for organizing and communicating some types of information about bottomland forest systems, workshop participants rejected the idea of using zones as a management tool precisely because the concept ignored the integrated nature of the bottomland ecosystems along complex environmental gradients. Repeatedly individuals and workgroups emphasized that the functions of each bottomland "zone" (as defined in Chapters 12 and 14) changed as water levels changed. The land-water interface moves up and down the zones with changes in river stage, and as this occurs the ecological processes associated with that interface move. Thus, while the idea of rating zones (for example, giving highest priority for preservation to the annually flooding zone, and progressively lower priority to higher zones), seemed to be a reasonable regulatory innovation, it ignored points repeatedly made in the workshops: (a) zones are interdependent, that is, the quantity and quality of water, materials, and biota in a particular zone depend on changes that occur as these items cross and interact with other portions of the floodplain ecosystem; (b) functional processes move among zones, depending on flooding depth; and (c) different zones perform different (or very similar) services and support different (or very similar) flora and fauna at different seasons, depending on flooding.

3) The regulatory focus on an individual site ignores the context of that site in the landscape. The ecological value of a site depends in part on its position in the landscape. For example, construction in a bottomland hardwood forest site upstream from a reservoir could threaten an urban water supply. Or, clearing a bottomland hardwood forest stand near a city could have a larger adverse effect on hunters, than clearing a similar stand many miles away from an urban area. Although the concept of landscape context is broadly recognized, from a wetlands regulatory perspective it has been difficult to address. Current §404 regulations consider ecosystem processes, but generally the focus has been on site-specific functions and services. Early evaluation approaches and procedures dealt strictly with site-specific conditions but not the landscape contexts in which wetland sites exist. More recently, WET (Adamus et al. 1987) began to broaden the focus of functional assessments by evaluating the opportunity and significance of a particular wetland process which may occur in a wetland, but landscape contextual considerations and their influence on the functional attributes of a particular wetland at both site-specific and landscape scales are narrowly interpreted.

4) Important ecological processes occur at landscape scales. Landscapes, as integrated ecological units, are characterized by processes

that have meaning only at landscape scales. Fortunately, these processes can often be measured at specific points in the landscape, the measurement reflecting the functional integrity of the whole system. For example, the hydrograph at a specific point on a river is a function of precipitation over the upstream portions of the watershed, land use on the watershed, the frequency and intensity of upstream human activities that affect water flows, watershed slope, and soil characteristics. Similarly, water quality at a specific site on a river, reflects upstream loading, which in turn is controlled by all the above parameters. Large, continuous tracts of bottomland hardwood forest were identified by both the wildlife and ecosystem processes workgroups as landscape structures with functions and attributes that transcend the average size of individual §404 permit sites. Large forested bottomland tracts consist of a diverse array of subsystems that have the potential to support a full set of species with narrow habitat requirements, especially those species adapted to forest interiors. These tracts maintain spatially heterogeneous patch-dynamic processes whereby shade-tolerant and shade-intolerant species can occur in all life stages. They enhance genetic interchange and diversity, which are especially important to maintain viable populations of large, terrestrial organisms. They also maintain faunal assemblages that have community and trophic integrity.

5) A site-specific focus cannot deal adequately with cumulative effects. Cumulative effects are, by their nature, landscape-level processes. They can be characterized as occuring in five general categories: time and space crowded, synergistic, indirect, and nibbling (Beanlands and Duinket 1983). The additive impacts (which result from nibbling), occur as a result of human activities on many different sites, which taken together have significant effects on the structural and functional integrity of the environment. Since evaluation of the environment of a single site through assessment techniques such as HEP or WET considers only the impact of modifying that site, it cannot and does not consider what happens to the whole landscape when many different sites are modified. Thus, almost by definition, a site-specific focus does not consider the landscape, and therefore ignores cumulative effects.

RECOMMENDATIONS FOR BOTTOMLAND HARDWOOD FOREST MANAGEMENT

Hierarchical considerations, as discussed above, lead to a number of recommendations for management of forested bottomland wetlands.

1) Regulation and management procedures must focus on the landscape as well as on site-specific impacts. Section 404 of the Clean Water Act, and attendant regulations and guidelines (33 CFR § 230, 40 CFR § 320-30), provide for regulation of dredge and fill activities in Waters of the United States, including wetlands. Section 404 does not regulate development of upland sites. Nevertheless, because uplands, wetlands and waterbodies (within river basins or hydrologic units), function as an integrated whole, effective management of bottomland hardwood forest wetlands requires that upstream and upslope ecosystems be managed as part of an integrated landscape. As discussed above, landscape management is necessary to: (a) maintain landscape-level ecological properties; (b) maintain the functional integrity of individual sites within the landscape, since these sites are influenced by what occurs around them; and (c) control cumulative impacts.

2) A landscape focus requires preplanning. As part of the federal Section 404 permit process, judgments about human actions in wetlands are made by permit specialists in response to individual permit requests prepared by project proponents. If regulatory actions are to maintain and restore the physical, chemical, and biological integrity of the nations's waters, as required in the Clean Water Act, they must be made in the context of plans for the entire landscape. This follows from previous arguments: if decisions are made on individual sites from site information alone, there can be no effective management of the landscape. And if the landscape degrades because of the cumulative effects of many site-specific decisions, the individual sites will also degrade, because they are part of the landscape. This implies that decisions about sites should be made only in the context of landscape plans. Such plans must be formulated and articulated prior to local decisions (Gosselink and Lee 1989, Lee and Gosselink, 1988). Although site evaluations are primarily reactive, landscape evaluations must be anticipatory, setting the conditions for site development.

3) The planning process requires an iterative sequence of ecological assessment, goal-setting, and planning for spatially discrete implementation. Assessment of the ecological condition of a landscape unit is a necessary precursor to its management. Assessment provides information about the condition and potential of the landscape system, from which goals and plans can be formulated (Gosselink et al. Chapter 17, Gosselink and Lee 1989). Such assessments should focus on landscape-scale processes, not details of individual sites. (Tables 6 and 7 in Chapter 17 list such processes and suggest indices by which to measure them.) Goal-setting forces all interested parties to reach consensus about the desired future of the total resource, not selected aspects of it. Finally, planning is the process of implementing the goals through prioritized actions at specific locations within the assessment unit. Within this context, permit decisions should be based on the "direction" of the impact of the proposed action with respect to the goals (Lee and Gosselink 1988). Generally, permits would be approved if cumulatively they move the landscape system toward stipulated goals. Permits would be denied if proposed projects move the landscape system away from approved goals. This kind of process requires monitoring of the system and provision of an "institutional memory" so that actions are recorded as they occur, and implementation strategies can be revised as the landscape system approaches the goals.

RECENT DEVELOPMENTS IN WETLANDS REGULATION

The bottomland hardwood forest workshops were held during a time when both the scientific and regulatory communities were becoming increasingly concerned with the issue of cumulative impacts. Since then, several workshops and/or symposia had cumulative impacts as their major focus and a number of articles on the subject were published (Bedford and Preston 1988a, CEARC 1986, Gosselink and Lee 1989, Williamson and Hamilton 1989). One of the workshops, held by the U.S. Environmental Protection Agency (EPA) in January 1987, was initiated specifically to develop more systematic cumulative impact assessment methods within a consistent conceptual framework. In the Preface to the published report of the workshop, Bedford and Preston (1988a) pointed to the "fundamental incongruity" that confronts those who regulate or study wetland ecosystems. "The scale at which they observe human impacts on wetland resources to be accumulating is far greater than the scale at which they ask questions or make decisions. Entire wetland landscapes have been altered inadvertently through the cumulative effects of numerous localized individual actions. Insights gained through research conducted at one site and on one process cannot provide straightforward answers about the consequences of multiple interacting processes operating at the scale of watersheds and landscapes" (p. 561).

In subsequent articles in this EPA workshop publication this theme is repeated many times. In their summary article Bedford and Preston (1988b) stated: "The primary conclusion to be drawn from the articles and the workshop is that improving the scientific basis for regulation will not come merely from acquiring more information on more variables. It will come from recognizing that a perceptual shift in temporal, spatial, and organizational scale is overdue. The shift in scale will dictate different -- not necessarily more -- variables to be measured in future wetland research and considered in wetland regulation" (p. 752).

The excellent articles resulting from the EPA workshop focus primarily on documentation of cumulative impacts to hydrology, water quality and life support functions of wetland ecosystems; and on conceptual approaches to measuring these impacts. The emphasis was on how to "scale up" research to make it consistent with the scale of management problems. The workshop did not address how, given appropriate technical information, cumulative impacts can be managed.

In contrast, the recommendations of the *National Wetland Policy* Forum, which met during 1987 and 1988, focused on action programs to solve a number of wetland problems, including the cumulative loss of wetlands. The Forum was an EPA-commissioned effort to develop national policy recommendations by individuals representing a cross-section of groups most interested in wetland issues. Participants included representatives of environmental organizations, academic interests, developers, farmers, foresters, and local, state and federal agencies. In its final report (Conservation Foundation 1988) the Forum recommended a national goal of "no overall net loss of the nation's remaining wetlands base in the short term, and an increase in the quantity and quality of the nation's wetlands resource base in the long term" (p.3).

One of a number of specific recommendations for implementing the no net loss goal is advance planning. "A major element in these reforms is increased emphasis on advance planning to guide our wetlands protection and management programs. The Forum recommends all states undertake the preparation of State Wetlands Conservation Plans to provide a basis for all subsequent acquisition, regulation, and other wetlands protection and management activities. These efforts, which should reflect local land-use plans and other societal values, should result in the nation's wetlands programs anticipating needs and problems rather than merely reacting to them" (Conservation Foundation 1988, p. 4).

The no net loss policy, if implemented, would elevate the status of wetland mitigation, since presumably dredge and fill permits that involved "unavoidable losses" would impose conditions for wetland creation and/or restoration to offset the losses. Currently, mitigation is carried out without much regard for whether created wetlands replace the services of the developed site in a landscape context (Larson 1987, Kusler and Kentula 1989). Federal guidelines state a preference for on-site, in-kind (i.e., local) replacement of wetlands that are impaired, but it has been extremely difficult for regulators and wetland scientists to address the question of functional equivalence, especially as related to position and function of wetlands within watersheds. Advance planning, which establishes a blue print for the future of a landscape unit, could identify key sites for wetland creation and/or restoration, that would enhance overall landscape integrity (Gosselink et al. 1990).

REGULATORY STRATEGIES TO IMPLEMENT PLANNING WITH A LANDSCAPE FOCUS

Wetland Regulation under Section 404 of the Clean Water Act

In the United States, the principle regulatory tool to protect wetlands is Section 404 of the Clean Water Act, and associated U.S. Army Corps of Engineers (ACE) and EPA regulations and guidelines (33 CFR §§ 320-330; 40 CFR §230). Aside from initial uncertainties about the regulatory intent of the Act and the incremental clarification of the regulations through successive court challenges (Anonymous 1988), the chief hindrance to effective regulation is probably the incompatibility of cumulative impact issues with site by site permit review. By definition, a site-specific focus cannot limit cumulative impacts, except in the most restrictive case of denying all §404 permit requests. That is, in the absence of landscape-level planning, there can be no effective mechanism to decide how much development is enough, or to allow development of one site but not another, if the site evaluations are equivalent. Under current administration of §404, the development of one site often sets a precedent for the approval of all similar permit requests (or types of projects), since the criteria for permit approval are usually the same regardless of landscape context. This inevitably leads to an "all or none" syndrome, in which environmental activists are pitted against development interests over isolated issues and local sites, rather than focusing on site-specific project impacts in the context of an overall plan.

Inability of the §404 program to manage cumulative impacts is compounded by the issuance of nationwide permits, particularly Nationwide Permit 26 (33 CFR 330.5 (a)(26)). This general permit, which allows up to 10 acres of fill to be placed into headwater wetlands (wetlands located in headwater reaches of tributary streams with a mean annual flow of less than 5 $ft^3\ sec^{-1}$) and in isolated wetlands (wetlands that are not bordering, neighboring, or contiguous to Waters of the United States), effectively exempts about 7 million ha of wetlands from regulation (Anonymous 1988), and may have led to significant cumulative impacts to water quality, hydrology, and food web support functions throughout the range of forested wetlands in the Southeastern and Mid-Atlantic states . Headwaters are critically important because they form the primary contact between terrestrial and aquatic ecosystems in landscapes that support bottomland hardwood wetlands. They also exist as the headward-most member of bottomland hardwood wetland continua.

Originally, the Nationwide Permit program was designed to reduce paperwork and agency response time to permit requests that were repetitive (e.g., Nationwide Permit 3 - repair and replacement activities), which had potentially minor impacts (e.g., Nationwide Permit 1 - aids to navigation), or which potentially impacted relatively small areas (e.g., Nationwide Permit 26 - headwaters, isolated or intermittent waters). Dredge and fill activities permitted under Nationwide Permit 26 usually proceed without requirements for mitigation. Thus, leniency in Nationwide Permit 26 permitting of projects proposed for headward extensions of watersheds is, by definition, bound to result in incremental, unmitigated cumulative impacts. This situation is exacerbated by the fact that incremental "nibbling" of headwater and isolated wetlands is common for three major reasons: (1) unscrupulous practices on the part of project proponents, who have been known to design projects so that individual parcels (in development projects) are less than 10 acres (e.g., 100 acre tracts split into multiple parcels of 9.9 acre); (2) high personnel turn-over rate within regulatory agencies, which leads to failure to track the cumulative effects of Nationwide Permit 26 permitting over time within a given watershed; (3) the slow pace of incremental wetland development (from an institutional viewpoint) that can occur over many years. Even with the best institutional memory, the cumulative effects of Nationwide Permit 26 activities are difficult to except after years of incremental change in a watershed.

Long Range Opportunities for a Landscape Focus and Planning

In the four years since the EPA-sponsored bottomland hardwood forest workshops, the idea that planning is a necessary precondition of cumulative impact management has gradually gained recognition. For example, the report of the National Wetlands Policy Forum states "To be effective, the nation's wetlands protection and management programs must anticipate rather than react.....They should consider the whole, not just the individual parts. In short, the programs should be based on comprehensive planning for wetlands protection and management...." (Conservation Foundation 1988:19). In addition, planning, which sets the conditions for management of the wetland resource, has been distinguished from regulation, which is one (albiet probably the strongest) of a number of tools available to implement planning. These tools run the gamut from programs of persuasion, offering incentives and disincentives, through regulatory prohibition, to acquisition. A difficulty in both planning and implementation is that they can involve many different agencies at federal, state, and local levels, as well as public and private interests, whose individual missions and interests must be merged to achieve a consensus about the future of the resource and to implement the plan. Each agency works within relatively restrictive regulations and guidelines developed as the best way to satisfy its mission. In this context it is appropriate to ask two questions about our institutional capability to address cumulative impacts. First, what agency, if any, has either the responsibility or the authority to initiate comprehensive planning? Second, within the existing regulatory programs of EPA (as the nation's lead environmental protection agency), what flexibility exists for comprehensive planning?

Federal Authority for Comprehensive Natural Resource Planning

In the United States no federal agency, or even group of agencies, has either the mission or the explicit authority to undertake comprehensive natural resource planning at landscape scales. This is in part because of the nature of the political system in this country. Generally, the legislative branch responds to particular issues with legislation tailored to address that issue. Legislation, therefore, is usually focused on a single resource (e.g., water quality, rare and endangered species), and agencies with responsibility for implementing the legislation are constrained by that focus. They do not have a mandate for managing the landscape to optimize all resources. A recent report on the nation's floodplain management activities (Johnston 1989), states (with reference to natural floodplain values) "....federal, state and local programs to manage these natural values are often not focused on the floodplain, but on the particular resource or activity. For example, programs have been developed to protect water quality, but they are not focused on managing floodplains for water quality protection...... Floodplain management and/or protection of natural floodplain values is typically not an explicit program objective" (pages 7-20 to 7-21).

Two reasons for the reluctance of federal legislators to address comprehensive planning are: (1) the strong tradition in the United States against land-use planning, which is seen as abrogating rights of individual ownership of land, and (2) the historical rights of states and local jurisdictions in these matters (i.e., environmental federalism). As a result, federal environmental legislation has generally avoided any hint of land-use planning. Rather, it has attempted to protect and enhance the public resources of air, water and federally owned or managed lands. Any restrictions on the use of private lands have been enacted to protect these public resources, and only after bitter battles in the Congress and in the courts. The requirement for a permit for wetland development, as required under § 404 of the Clean Water Act, is an excellent example of this type of legislation. The Act's purpose is to maintain and restore the physical, chemical, and biological integrity of the nation's waters, which include wetlands (40 CFR 328.3 (a) 1-7). Thus, regulation of dredge and fill activities in privately owned wetlands is justified in the Act because wetlands have been shown to function in several ways that directly influence the maintenance of water quality. In contrast, land management practices in uplands, which can also strongly affect downslope water quality, are not usually regulated. (We refer here to such practices as land clearing, not to the production of point source effluents, which <u>are</u> regulated.)

Despite these political considerations, several Acts passed by the Congress during the late 60's and 70's do provide possibilities for natural resource planning on landscape scales. The 1970 National Environmental Policy Act (NEPA, 42 U.S.C. 4371 *et seq.*) broadly recognized natural environmental values and incorporated them into federal decision-making for large projects. Although the requirement to address cumulative impacts is specific in NEPA, in practice the Environmental Impact Statement is largely reactive. Environmental Impact Statement review is not usually viewed as a comprehensive planning process, and thus has never effectively addressed cumulative impacts.

An earlier Act, the Water Resources Planning Act of 1965 (WRPA), has greater potential for comprehensive planning. Although it was passed to address the widespread problem of flood losses, in combination with NEPA it has the potential to encourage the development of comprehensive resource management plans. The WRPA created the Water Resources Council, which was charged with (1) assessing the adequacy of basin plans and establishing principles and standards for federal participation in river basin planning; and (2) creation, operation and termination of interstate government river basin planning commissions. The Principles and Standards issued by the Water Resources Council (1973) established three "accounts" by which proposed actions were to be evaluated; (1) a "National Economic Development" account, which, as implied, evaluated the economic utility of the project; (2) an "Environmental Quality" account, and (3) an account called "Social Well Being". This document was a major attempt to describe a procedure for standardized basin-level planning that addressed multiple management objectives including both economic development and environmental quality (Field 1979)

The Water Resources Council was active in promoting the establishment of River Basin Commissions. Several were established in the late 60's and early 70's, and were required to prepare comprehensive, coordinated plans for their region or basin.

The ambitious programs set forth by the WRPA did not fare well. In 1982, funding was withdrawn from the Water Resources Council and River Basin Commissions. The Water Resources Council was disbanded, and the River Basin Commissions gradually closed their operations. The 1983 revision of "Principles and Standards" dropped the "Environmental Quality" and "Social Well-Being" accounts. As a result the "National Economic Development" account provides the sole basis for project justification (Interagency Task Force on Floodplain Management 1986). In the absence of the Water Resources Council, the WRPA and Executive Orders related to floodplain and wetland management (Executive Orders 11988 and 11990) have been coordinated by an Interagency Task Force on Floodplain Management (created in 1975 to develop a Unified National Program for Floodplain Management, and formerly chaired by a Water Resources Council representative). While this task force has sponsored several important initiatives it does not consider itself a policy group, and does not have the influence of the Water Resources Council.

Another Act that has potential for comprehensive natural resource management is the Watershed Protection and Flood Prevention Act of 1954, administered by the Soil Conservation Service. This Act authorized the Soil Conservation Service to participate in comprehensive watershed management projects in cooperation with states and their subdivisions. Eligible projects are limited to watersheds of less than 250,000 acres (Stembridge, undated). Although the goals of the Act were water and erosion management, it is currently being used in a much broader context, for example as a mechanism for wetlands protection.

Generally, responsibility for land-use planning is reserved for the states and local authorities. State and local governments are in a stronger position than the federal government to institute comprehensive planning (1) through their authority to initiate zoning and other land use restrictions, and (2) because they are closer to the local problems and thus able to tailor plans to local conditions. Therefore, federal legislation, such as the WRPA and the CWA, does not preempt this authority, but through a system of incentives encourages state assumption. For example, the CWA encourages states (through grants for compliance) to assume responsibility for regulation of local water quality, including (in the 1987 revisions), the development of plans for control of non-point pollution sources. The report of the National Wetlands Policy Forum reflects this trend. "The Forum recommends that: state and local governments and regional agencies, with the support and cooperation of the relevant federal agencies, undertake wetlands planning to achieve the goal of no net loss, and that Congress allocate adequate funds to assist with these efforts" (Conservation Foundation 1988, p. 19).

In summary, there is at present little Congressional, executive, or federal agency encouragement or authority for comprehensive multiple objective landscape planning, of the kind needed to manage cumulative impacts. In some respects (for example, with the Water Resources Council and the WRPA) there has been a retreat, since the 1970's, in the executive branch's willingness to deal with comprehensive planning. Some agencies are working to institute such planning within the authority of existing statutes and regulations. However the nature of these statutes often limits the scope of planning efforts. Nevertheless, some existing statutes provide mechanisms to address cumulative impacts. In the following section we examine EPA's opportunities for cumulative impact management through its mandate for wetland protection.

EPA Opportunities for Landscape Planning and Cumulative Impact Management

Advance Identification. Section 404 of the Clean Water Act requires EPA and the ACE to regulate dredge and fill activities in wetlands. Nationwide the ACE has authority to evalute and issue permits for dredge and fill activities, but EPA has authority to veto ACE decisions under §404(c) of the CWA. The procedure, as described in regulations issued by the ACE and EPA (33 CFR §§ 320-330; 40 CFR §230) is largely reactive (that is permit evaluations are usually initiated in response to an individual application). Because the §404 permit process focuses on individual permit sites it fails to address landscape contextual problems, and thus provides no mechanism to manage cumulative impacts.

EPA, however, has authority for planning in the Advance Identification program, as described in Section 230.80 of the 404(b)(1) Guidelines (40 CFR Part 230). The ACE has a similar authority under their "Special Area Management Plans". Under §230.80 EPA and the permitting authority (ACE or an approved State agency), act jointly to identify wetlands and other waters of the United States as possible future disposal sites or areas generally unsuitable for disposal site specification. The results are informational, not regulatory. Any person may still submit an application for a §404 permit regardless of the designation of the site. The results of the advance identification process simply put the applicant on notice about the relative probability of obtaining a permit. Thus this program is used to improve the public's understanding of wetlands and to provide a degree of predictability to the regulated community that does not exist within the context of individual permit reviews. It also provides EPA regional offices opportunities to coordinate more effectively in local or state planning processes. Documents developed in the advance identification process contain wetlands assessment and other data that are potentially useful in other wetland protection activities such as the development of local zoning and regulations, the identification of valuable sites for purchase or zoning easements, and for public education efforts.

Although the §404 (b)(1) guidelines are specific as to the purpose of Advance Identifications (i.e., designation of wetlands as unsuitable for disposal site specification) EPA's Advance Identification guidelines (Office of Wetland Protection 1989) clearly encourage a broad perspective on ecosystem protection and management. For example, the program can be used in combination with other portions of the §404 (b)(1) guidelines to address large geographically-based issues. Advanced identifications can be focused to reduce or reverse regional trends of wetland losses, and/or for assessing cumulative and secondary impacts within designated boundaries (§230.80 (g) & (h)). The preferred units for advanced identification program are watersheds or ecosystems because components of these systems functionally relate to one another. Broad participation by all agencies and groups with interests in the geographic area is usually encouraged, beginning early in the advance identification process (Office of Wetland Protection 1989). Thus the advance identification program may be used as a means to initiate comprehensive planning, even though EPA's authority under Section 404 is limited to jurisdictional wetlands.

The potential of Advance Identifications for use as a planning mechanism for cumulative impact management is just beginning to be realized, with the recent issuance of guidelines by the EPA Office of Wetland Protection. The number of advance identification programs initiated across the U.S. has increased rapidly in the past three years, and several have been completed. They vary in size from several to over 100,000 ha. Documents prepared for these programs are limited to the authority under §230.80 to identify wetlands unsuitable for disposal. It is not clear, therefore, to what extent they are being used for more comprehensive planning. One difficulty is that no agency has authority to expand the advance identification program into a comprehensive multiple objective project that can result in the development of natural resource protection goals and bring to bear many different regulatory and non-regulatory tools to implement the goals. Some broader authority is needed to accomplish this. It could be located at the state level, but most states are not organized to handle programs that cut across federal and state agency lines. The Advance Identification guidelines (Office of Wetland Protection 1989) provide a broad perspective, and specific recommendations for comprehensive planning, but their effective implementation appears to require broader authority and more focused support from EPA.

Interim Opportunities for Cumulative Impact Management. As EPA and cooperating federal, state and private sector groups gain more experience with and more confidence in Advance Identifications, we can expect better recognition and management of cumulative impacts in wetlands. In the short term, the §404(b)(1) requirements for alternatives analysis, impact assessment and impact minimization can provide opportunities to improve the permit review process and specifically to address cumulative impacts. However, there are distinct administrative, conceptual, and practical limits regarding the extent to which cumulative impact assessments can be effectively incorporated into individual §404 permit reviews. These limits are defined mostly by agency time and by staffing constraints. As discussed above, agency personnel are significantly influenced by lack of clear definition of "goals" for management of an individual permit site in the context of the landscape in which it occurs, by lack of "institutional memory" on the part of regulatory agencies, by lack of familiarity with the rapidly evolving field of landscape ecology, and by the unavailability of landscape-scale data in a usable and accessible form.

Goals set the context for assessment of cumulative effects and their perceived impacts on the condition of wetlands. Even though the §404(b)(1) Guidelines direct federal agencies to review "cumulative effects" (230.11(g)) and "secondary effects" (230.11(h)) in the context of individual permit reviews, only broad direction is given in the §404(b)(1) Guidelines, the ACE Regulations (33 CFR 320-330) and from the Clean Water Act itself regarding standards or criteria for review of cumulative and secondary effects. Thus, while the agencies are required by their guidelines to review cumulative and secondary effects, they currently have no specific standards for doing so. In addition, while most agency managers and staff are relatively well trained in basic biology, jurisdictional delineation of wetlands, and the administrative or regulatory details of individual §404 permit review, they are relatively unfamiliar with the new and rapidly evolving field of landscape ecology. This lack of command of the subject matter often leads to review of §404 permits in the context of what is familiar to agency staff and management (i.e., site-specific review), and not necessarily what is most technically valid (i.e., site specific reviews combined with landscape-level reviews).

In practice, the importance of cumulative and secondary effects has been recognized by relatively innovative regional offices of the EPA, ACE and U.S. Fish and Wildlife Service. Such recognition is reflected in documents (such as letters of denial or written recommendations for project revisions) that address cumulative and secondary effects. However, most of these agency comments are developed in reaction to an individual permit request and without substantive technical support in the form of data and/or credible models that document cumulative or secondary effects. The lack of sound technical substantiation of such impacts is open to successful challenge by individuals with knowledge of current methods and data.

The main point to emphasize is that assessment of cumulative impacts needs to be carefully and slowly implemented, and incorporated into the standard procedures used by regulatory agencies during their reviews of wetland-related projects. Implementation should be incremental, along with "business-as-usual" considerations of individual permit reviews. Because introduction of landscape-level reviews into the §404 process is experimental, agencies need to develop and maintain a dual track approach to permit review. When possible, data from cumulative impact assessment efforts initiated under regional Advance Identifications or Special Area Management Plans should be used to substantiate claims of cumulative or secondary effects related to individual permit reviews (e.g., EPA Region Three Recommended Section 404c determination for the Ware Creek, VA Project, USEPA, Philadelphia, PA, 1989). Simultaneously, agencies need to devote increased time and resources to Advance Identifications and/or Special Area Management Plans that rely principally on landscape-level analyses that focus on management of cumulative impacts and anticipatory approaches to permit reviews (Gosselink and Lee 1989).

Another effective tool in cumulative impact management is the "layering" of State §401(Water Quality) authority over individual as well as nationwide permit reviews. This approach has proven to be an effective tool consistent with the goals of environmental federalism as articulated by the current (Bush) administration. For example, states with §401 permitting authority can effectively veto issuance of individual §404 or Nationwide Permit 26 permits through denial of state wetlands permits or §401 certification.

A final suggestion regarding current approaches for dealing with cumulative effects relates to two major concerns articulated throughout the bottomland hardwood forest workshops by agency personnel: (1) the need for a unified approach to the definition of wetlands under federal jurisdiction, and (2) the need to deal consistently with agricultural and silvicultural exemptions in conformity with §404(f)(1) and (f)(2).

With regard to development of a consistent, unified approach for definition of wetlands, the "Federal Manual for Identification and Delineation of Jurisdictional Wetlands", issued January 10, 1989 (Federal Interagency Committee 1989), promises to go a long way towards shifting the focus of §404 review from questions concerning jurisdiction to more substantive issues. This is a positive step for the national wetlands protection process. It has particular relevance for dealing with cumulative impacts because agreement on wetland jurisdictional boundaries allows participants in the §404 process to devote more time and effort to deal with identification of project purpose, siting alternatives outside jurisdictional wetlands, minimization of impacts, and mitigation of unavoidable impacts, as required in the §404 Guidelines. With the recent emphases on landscape ecology and "no net loss" of wetlands, such efforts are bound to incorporate landscape-level thinking into the planning processes associated with individual permit reviews and with Advance Identifications and Special Area Management Plans.

Section 404(f)(1) & (2) of the Clean Water Act address agricultural, silvicultural and ranching exemptions for "on-going and established" operations (40 CFR 232.3). The new wetlands delineation manual clarifies regulatory jurisdiction over these exemptions. Combined with the Swampbuster provisions of the Food Security Act of 1985, consistent application and enforcement of §404(f)(1) & (2) can work to maintain bottomland hardwood forest patch size and structural integrity, help to limit non-point source inputs to Waters of the United States, and maintain reasonably natural hydrologic connections with functioning (even though farmed or forested) wetland ecosystems.

The National Wetlands Policy Forum (Conservation Foundation 1988) identified a number of pressing wetlands issues that need to be addressed. The Forum suggested that while some of the present regulatory deficiencies can be addressed by modifying present regulations, others probably require new legislation at both federal and state levels. Recognizing the slow pace of major legislative initiatives we have, in this chapter, focused on the possibilities for improving wetlands protection within the present statutory and regulatory framework. Summarizing these possibilities, a three-pronged approach, operating within current regulatory constraints, can produce significant improvements in management of cumulative impacts in bottomland hardwood forests.

1) Gradually incorporate landscape ecology principles and cumulative impact evaluation into current wetland permit reviews that are completed under federal or similar regional, state, or municipal authorities;
2) Expand the use of advance planning authority under §230.80 of the EPA §404 Guidelines, and the ACE Special Area Management Program, to address landscape level planning and cumulative impacts assessment;

(3) Continue to explore ways to use the comprehensive planning authority of the Water Resources Planning Act of 1965 and the Watershed Protection and Flood Prevention Act of 1954, to incorporate comprehensive planning at landscape scales. This approach integrates the multiple missions of the many federal, state, and local authorities and private interests that have a stake in wetlands. In the final analysis only comprehensive planning can lead to effective management of cumulative impacts, and effective protection of forested bottomland resources.

LITERATURE CITED

Adamus, P. R., and L. T. Stockwell. 1983. A method for wetland functional assessment. U.S. Army Waterways Experiment Station, Vicksburg, MS. Tech. Rep. D-77-223. 72 p.

____________, E. J. Clairain, Jr., R. D. Smith, and R. E. Young. 1987. Wetland evaluation technique (WET); Vol. II: Methodology. Operational Draft Technical Report Y-87-__. U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Anonymous. 1988. A guide to federal wetlands protection under Section 404 of the Clean Water Act. Natural Resources Law Institute, Lewis and Clark Law School, Portland, OR. Anadromous Fish Law Memo 46: 1-34.

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

Bedford, B. L., and E. M. Preston, eds. 1988a. Cumulative effects on landscape systems of wetlands. Environmental Management 12 (5): 561-775.

____________________.. 1988b. Conclusions. Pages 751-771 in B. L. Bedford and E. M. Preston, eds. 1988. Cumulative effects on landscape systems of wetlands. Environmental Management 12 (5): 561-775.

Brinson, M. M., R. L. Swift, R. C. Plantico, and J. S. Barclay. 1981. Riparian ecosystems: their ecology and status. U.S. Fish Wildl. Serv., Biol. Serv. Prog., Washington, D.C. FWS/OBS-81-17. 151 p.

CEARC (Canadian Environmental Assessment Research Council) and NRC (United States National Research Council. 1986. Cumulative environmental effects: a binational perspective. Minister of Supply and Services Canada. 174 pp.

Clark, J. B., and J. Benforado, eds. 1981. Wetlands of bottomland hardwood forests. Elsevier Sci. Publ. Co., N.Y. 401 pp.

Conservation Foundation. 1988. Protecting America's wetlands: an action agenda. The final report of the National Wetlands Policy Forum. Washington, D.C. 69 pp.

Federal Interagency Committee for Wetlands Delineation. 1989. Federal manual for identifying and delineating jurisdictional wetlands. U.S. Army Corps of Engineers, U.S. Environmental Protection Agency, U.S. Fish Wildl. Serv., and U.S.D.A. Soil conservation Service. Washington, D.C. Cooperative Technical Publication. 76 pp. & Appendices.

Field Associates, R. M. 1979. Regional water resources planning: a review of level B study impacts. Prepared for the U.S. Water Resources Council, Washington, D.C.

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

Golet, F. C., and J. S. Larson. 1974. Classification of wetlands in the glaciated Northeast. U. S. Fish Wildl. Serv., Washington, D.C. Resources Publ. 116. 56 pp.

Gosselink, J. G., S. E. Bayley, W. H. Conner, and R. E. Turner. 1981. Ecological factors in the determination of riparian wetland boundaries. Pages 197-219 *in* J. B. Clark and J. Benforado, eds. Wetlands of Bottomland Hardwood Forests. Elsevier Sci. Publ. Co., N.Y. 401 pp.

____________, G. P. Shaffer, L. C. Lee, D. Burdick, D. Childers, N. Taylor, S. Hamilton, R. Boumans, D. Cushman, S. Fields, M. Koch and J. Visser. 1990. Can we manage cumulative impacts? Testing a landscape approach in a forested wetland watershed. Bioscience (in press).

Gosselink, J. G., and L. C. Lee. 1989. Cumulative impact assessment in bottomland hardwood forests. Wetlands 9:83-174.

Interagency Task Force on Floodplain Management. 1986. A unified national program for floodplain management. Federal Emergency Management Agency, Washington, D.C. FEMA 100/March 1986.

Johnson, R. R., and J. F. McCormick. 1979. Strategies for protection and management of floodplain wetlands and other riparian ecosystems. U. S. Forest Service, U. S. Dept. Agriculture, Washington, D.C. General Tech. Rep. WO-12.

Johnston Associates, L. R. 1989. Status report on the nation's floodplain management activity. An interim report. Prepared for the Interagency Task Force on Floodplain Management, Washington, D.C.

Kusler, J. A., and M. E. Kentula. 1989. Wetland creation and restoration: the status of the science. Vols. I and II. U. S. Environmental Protection Agency, Corvallis, OR. EPA/600/3-89/038.

Larson, J. S. 1987. Increasing our wetland resources. Proc. Nat. Wildl. Fed. Symp., 5 Oct 87. Washington, D.C.

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

Office of Wetland Protection. 1989. Use of Advance Identification authorities under Section 404 of the Clean Water Act. Guidance to EPA regional offices. U.S. Environmental Protection Agency, Office of Water, Washington, D.C. IM-89-2.

Roelle, J. E. 1987. Workshop summary. Pages 156-171 *in* Roelle, J. E., G. T. Auble, D. B. Hamilton, G. C. Horak, R. L Johnson, and C. A. Segelquist, eds. Results of a workshop concerning impacts of various activities on the functions of bottomland hardwoods. U.S. Fish Wildl. Serv., National Ecology Center, Fort Collins, CO. NEC-87/15. 171 pp.

Soulé, M. E., and B. A. Wilcox, eds. 1980. Conservation biology: an evolutionary-ecological perspective. Sinauer Associates,

Stembridge, J. E., Jr. undated. The role of Conservation Districts in urban floodplain management. Coast Environment Resources Institute, OR. Working Paper No. 4.

Turner, M. G. 1989. Landscape ecology: the effect of pattern on process. Annual Rev. Ecol. Systematics 20: 171-197.

U.S. Fish and Wildlife Service. 1980. 101 ESM -Habitat as a basis for environmental assessment. 102 ESM - Habitat evaluation procedures. Division of Ecological Services, Washington, D.C.

Urban, D. L., R. V. O'Neill, and H. H. Shugart, Jr. 1987. Landscape ecology. Bioscience 37 (2): 119-127.

Wharton, C. H., W. M. Kitchens, E. C. Pendleton, and T. W. Sipe. 1982. The ecology of bottomland hardwood swamps of the Southeast: a community profile. U. S. Fish Wildl. Serv., Biol. Serv. Prog., Washington, D.C. FWS/OBS-81/37.

Williamson, S.C., and K. Hamilton. 1989. Anotated bibliography of ecological cumulative impacts assessment. U.S. Fish. Wildl. Serv., Washington, D.C. Biol. Rep. 89(11). 80 pp.

Water Resources Council. 1973. Principles and standards for planning water and related land resources. Federal Register 38 (174): 10 Sep 1973, 24778, with 2 appendices.

Landscape Conservation in a Forested Wetland Watershed

Can we manage cumulative impacts?

James G. Gosselink, Gary P. Shaffer, Lyndon C. Lee, David M. Burdick, Daniel L. Childers, Nancy C. Leibowitz, Susan C. Hamilton, Roel Boumans, Douglas Cushman, Sherri Fields, Marguerite Koch, and Jenneke M. Visser

More than one-half of the 40 million ha of wetlands in the coterminus United States is forested (Frayer et al. 1983). Most of these wetlands (57%; Abernethy and Turner 1987) are in the southeastern United States. They are characterized as permanently, semipermanently, or intermittently flooded and are dominated by cypress (*Taxodium* spp.), tupelo (*Nyssa* spp.), and oak (*Quercus* spp.). The broad Mississippi River alluvial floodplain, which extends from the Gulf of Mexico to southern Illinois, historically supported the largest United States expanses of forested wetlands, but since the 1950s these areas have been rapidly converted to the production of cotton, corn, and soybeans (OTA 1984). Brinson et al. (1981) estimated the loss of riparian forest at more than 70% since presettlement days. Abernethy and Turner (1987) calculated a 23% loss since the 1950s. In the Mississippi River alluvial floodplain alone, bottomland forests decreased from 4.8 million ha in 1937 to 2.1 million ha in 1978, a 55% loss (MacDonald et al. 1979).

> **Prompt action is needed for landscape planning to be cost effective**

This rapid wetland loss is of intense concern to environmental interests. Forested riparian wetlands perform a number of valuable services for humans, including moderation of downstream flooding, maintenance of good water quality, and provision of diverse habitats for wildlife (Wharton et al. 1982). These are largely public benefits. Aside from the economic return for resource harvests—timber and wildlife—benefits do not generally accrue to the land owner, but rather to individuals or groups downstream. Wetlands have therefore been recognized as public resources and are federally protected under the Clean Water Act (CWA) of 1972 (Public Law 95-217). Despite this protection, freshwater wetland loss, and particularly forested wetland loss, continues. The cumulative impact of many individual actions, no single one of which is particularly alarming, threatens the integrity of whole wetland landscapes. Analysis of potential solutions to this problem makes an interesting study of cumulative environmental degradation, a phenomenon that is increasingly widespread.

This article addresses general issues in environmental planning related to the cumulative impacts of human activities on the environment. We focus specifically on wetlands, although the problem is more general, and the issues addressed and methods discussed have broad application. To set the stage, we introduce the legal and administrative framework for wetland regulation, the nature of cumulative impacts, and the use of ecological principles (specifically landscape ecology principles) in environmental planning. Next, we assess the cumulative impact of human activities in the Tensas River basin, Louisiana, and show how the assessment can be used for planning purposes. Finally, we discuss the generality of this ap-

James G. Gosselink is a professor in the Marine Sciences Department and Gary P. Shaffer is an assistant professor in the Department of Experimental Statistics at Louisiana State University, Baton Rouge, LA 70803. Lyndon C. Lee, formerly an assistant research professor at the Savannah River Ecology Laboratory of the University of Georgia and a senior ecologist in the Office of Wetland Protection, US Environmental Protection Agency, Washington, DC, is now with L. C. Lee Associates, Inc., Seattle, WA 98119. David M. Burdick is a postdoctoral investigator in the biology department at Woods Hole Oceanographic Institution, Woods Hole, MA 02543. Daniel L. Childers is an assistant research professor at the Baruch Institute for Marine Biology and Coastal Research, University of South Carolina, Georgetown, SC 29442. Nancy C. Leibowitz is a wetland scientist with NSI Technology Services Corp., Corvallis, OR 97333. Susan C. Hamilton and Roel Boumans are research associates in the Coastal Ecology Institute at Louisiana State University. Douglas Cushman, formerly a student at Louisiana State University, now resides in West Linn, OR 97068. Sherri Fields in an intern at the US Environmental Protection Agency in Washington, DC. Marguerite Koch is a research environmentalist at the South Florida Water Management District, West Palm Beach, FL 33416. Jenneke M. Visser is a postdoctoral research scientist in the Coastal Ecology Institute at Louisiana State University.

proach and its applicability in other contexts.

Federal jurisdiction in forested wetlands

Discharges of dredged or fill material into the waters of the United States are regulated under Section 404 of the Federal Water Pollution Act Amendments of 1972 (33 U.S.C. Section 1251), as amended in the CWA of 1977 and again in 1987 (Public Law 100-4). In practice, the term *waters of the United States* has been defined to include wetlands and, specifically, most bottomland hardwood forests.[1] Although "normal" forestry practices are statutorily exempt under Section 404(f) of the CWA, clearing of forested wetlands for conversion to uplands results in discharges regulated under Section 404.[2] Large areas of wetland forests were cleared before 1972, when activities in wetlands became regulated under the Federal Water Pollution Act. For both legal and technical reasons, clearing has continued despite the protection of the CWA.

It has taken years and a series of court decisions (Natural Resources Law Institute 1988) to clarify uncertainties about the geographic jurisdiction of Section 404 and the types of activities it exempts. For example, the US Army Corps of Engineers, which jointly administers the Section 404 program with the US Environmental Protection Agency (EPA), agreed, only as recently as 1984, to apply nationwide the decision in Avoyelles Sportsmen's League v. Marsh (National Wetlands Newsletter 1984). This decision, which clarified the jurisdictional definition of a wetland and what constitutes a regulated activity, was not reflected in the regulations that guide permit processing until November 1986.[3]

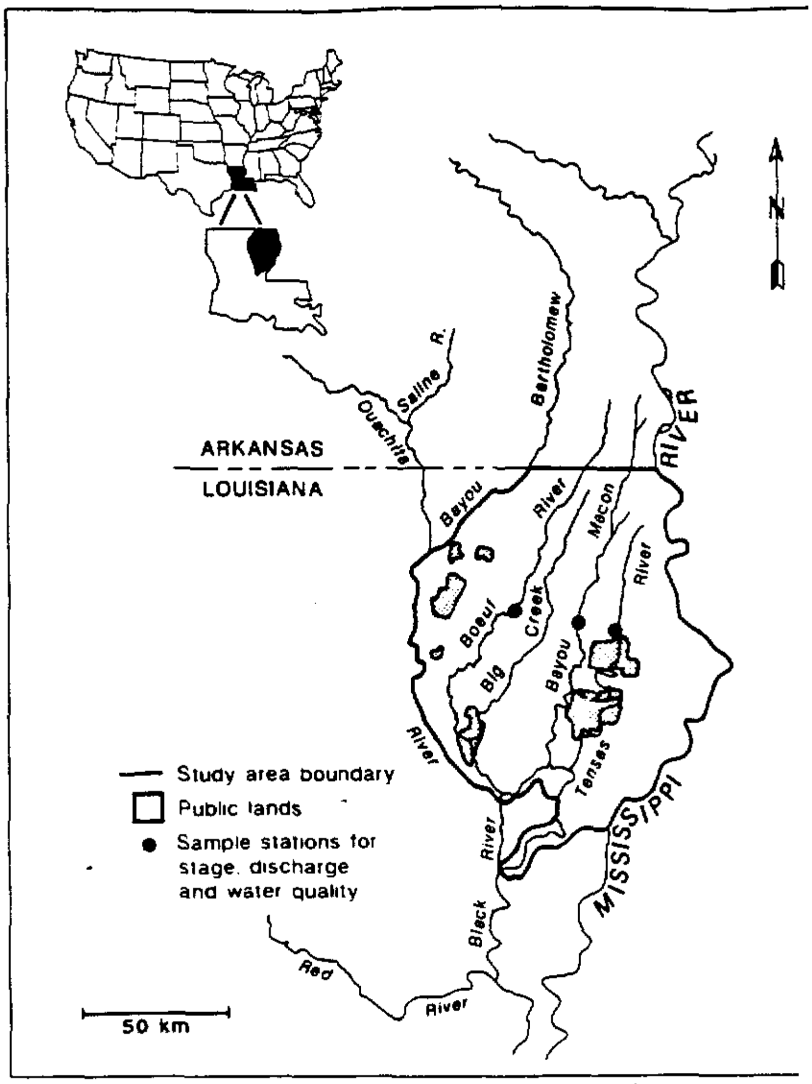


Figure 1. Location of the Tensas basin in northeastern Louisiana. The basin is bound on the east by the Mississippi River, on the south by the Tensas-Cocodrie levee, and on the west by the Bayou Bartholomew watershed boundary and the Ouachita and Bla River levees. The northern boundary is political, the Arkansas-Louisiana border. Wat flows generally from north to south through the study area. Principal streams are t Boeuf and Tensas Rivers (tributaries of the Ouachita River) and Bayou Macon tributary of the Tensas River).

Cumulative impacts

An important technical hindrance to protection of forested wetlands has been the difficulty of managing the cumulative impacts of incremental clearing of small tracts (Lee and Gosselink 1988). *Cumulative impact* is defined in the Council on Environmental Quality regulations (which implement the National Environmental Policy Act of 1969)[4] 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 other actions. Cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time. (40 C.F.R. §§ 1508.7 and 1508.8)

[1] 33 C.F.R. 328.3(b) and 40 C.F.R. 232.2(r); Avoyelles Sportsmen's League v. Marsh, 715 F.2d 897, 903 n.12. 5th Cir. 1983.
[2] 33 C.F.R. Section 323.4(c) and 40 C.F.R. Section 232.3(b), Avoyelles Sportsmen's League v. Marsh, 715 F.2d 897, 903 n.12. 5th Cir. 1983.
[3] 33 C.F.R. §§ 320–330.

[4] 42 USC. 4321–4347.

The CWA and regulations for implementation of Section 404 by both EPA (40 C.F.R § 230) and the Army Corps of Engineers (33 C.F.R. §§ 320–30) require consideration of cumulative impacts, but for a number of reasons (Beanlands et al. 1986, Horak et al. 1983) they are seldom evaluated in permit review processes.

Wetland forest conversion to agriculture is a typical cumulative impact. Historically, the incremental clearing of 10 ha to as much as 2000 ha in an individual permit has been perceived to have no significant ecological impact on a total forest system of several million hectares, and the cumulative effect of many such permitted activities has been ignored.

This failure can be understood if the current regulatory process is contrasted with the kind of process required for cumulative impact assessment. The Section 404 permit process focuses on the impact of a proposed activity at an individual wetland permit site. In contrast, cumulative impacts are landscape-level phenomena that result from decisions at many individual permit sites, as well as activities that are not regulated under Section 404 (Gosselink and Lee 1989). Hence cumulative impacts are often external to the focus of individual permit review. In addition, the current permit process is largely reactive; that is, the decision about whether or not to approve an activity on a site is made in response to a permit request, not in advance of it.

If cumulative impacts are to be managed, the decision at an individual site will have to be governed by earlier decisions made about the allowable extent of modification of the whole landscape unit. Thus cumulative impact management has the potential to change current wetland regulatory practices in two significant ways: it raises the focus of management from site-specific to natural landscape units, and it imposes planning on the current Section 404 process, which is largely reactive. It should be noted that EPA has authority for planning under the regulations addressing advance identification procedures (40 C.F.R. § 230.80).

Techniques for addressing cumulative impacts presented in recent workshops and publications (Beanlands et al. 1986, Bedford and Preston 1988,

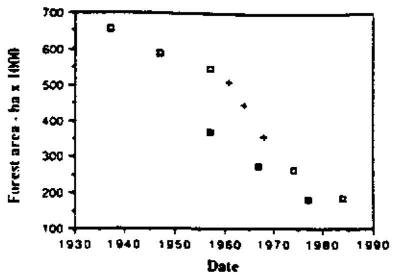


Figure 2. Bottomland forest areal changes in an area covering seven parishes in the Tensas basin since 1935. Open squares from US Forest Service surveys (Earle 1975, May and Bertelson 1986), closed squares from MacDonald et al. (1979), and crosses from Yancey (1970). The seven-parish boundary approximates the boundary of the Tensas basin and covers an equivalent area.

Clark 1986, Vlachos 1985) include one or more of the following: checklists of characteristics or processes to be considered in the analysis, matrices of interactions among human activities and environmental conditions, nodal networks that depict likely impacts from disturbances, and simulation models of ecosystems and responses to human activities (Risser 1988). In general, these approaches build on widely accepted techniques of conventional environmental impact analyses (McAllister 1980, Munn 1975, Westmann 1985). However, there is not currently any methodology for cumulative impact assessment, or even any conceptual approach that is generally accepted by scientists and managers. Risser (1988) recommends continuing the search for a suite of cumulative impact techniques underlain by a strong conceptual basis that incorporate recent dramatic advances in our understanding of landscape ecology as well as new methods (e.g., remote sensing techniques) for landscape analysis.

Gosselink and Lee (1989) described a methodology for cumulative impact assessment and management that incorporates both planning and a landscape-level focus. It has three components:

- Assessment. The characterization of cumulative effects on both the ecological structure and the functional ecological processes in a designated landscape unit.
- Goal-setting. Agreement by public consensus on environmental goals for the assessment area, based on the assessment and consistent with regulations under the CWA.
- Implementation. The development of specific plans to implement the goals, based on the landscape structure and function of the area that is assessed.

The need to manage cumulative impacts on a landscape scale requires that the boundaries of an assessment unit encompass an area that is, to the extent possible, ecologically closed to water and nutrient flows (so that forces external to the basin can be minimized) and also large enough to satisfy the home range and habitat requirements of the farthest-ranging animal species of interest (e.g., the black bear or the Florida panther). The latter requirement has a double benefit. It ensures that the analysis will address the protection of the target species, and in so doing it also encompasses a diverse group of species with smaller ranges. Pragmatic considerations such as political jurisdiction and map scales also influence the choice of boundaries. Gosselink and Lee (1989) recommend boundaries that enclose areas of 1 million ha or more that are natural hydrologic watersheds or drainage basins.

To characterize an area this large, the proposed cumulative impact assessment methodology focuses on a limited number of landscape indices that reflect ecological structure and hydrologic, water-quality, and biotic functions. By *landscape indices*, we mean simple, measurable properties that integrate ecological processes over large areas. For example, a stream water-quality record reflects water chemistry conditions in the watershed above the sample station. Long-term data records are used, thus enabling a time-series analysis of system change.

Landscape ecology and natural resource conservation

Troll (1950) defined landscape ecology as the study of the physicobiological relationships that govern the different spatial units of a region. It deals with large areas and the interaction of parts within these areas. Thus the emphasis is on the pattern of the landscape and how pattern influences *ecological processes or functions* (Forman and Godron 1986, Turner 1989).

Of particular relevance to cumulative impacts is the study of island biogeography, a field pioneered by MacArthur and Wilson (1963, 1967), and the application of that knowledge to the design of ecological preserves (Diamond 1975, Harris 1984, Margoulis and Usher 1981, May 1975). These studies are concerned with the size and shape of patches in the landscape, their isolation from each other, and the influence of these factors on species diversity. Whereas in the pioneering studies the patches were islands isolated by water, in applications to natural preserves the patches studied were forests, isolated by grasslands, agricultural fields, or other human barriers.

Diamond (1975) summarized five landscape principles for natural reserves:

- Species richness increases with forest area.
- For a given total forest area, one large reserve will support more native, interior species than two or more smaller ones.
- For a given forest area, disjunct patches that are close will support more species than patches that are farther apart.
- Disjunct forest patches connected by strips of protected habitat are preferable to isolated patches (the protected corridors facilitate animal movement between patches and provide gradual ecotones between similar habitat types).
- Other things being equal, a circular reserve is preferable to a linear one, because the former maximizes dispersal distances within the reserve and minimizes the edge relative to the interior.

We use these principles in the following cumulative impact discussion, although we recognize that considerable controversy surrounds the application of insular biology to applied conservation issues (Noss and Harris 1986, Schonewald-Cox et al. 1983, *Simberloff and Abele 1982*, Simberloff and Cox 1987, Soulé and Simberloff 1986).

Assessing cumulative effects in a river basin

We have assessed cumulative effects in the Tensas River basin in northeastern Louisiana. We illustrate the feasibility of a landscape-level assessment and the utility of this approach for natural resource conservation planning.

Description of the Tensas basin. The Tensas basin study area is composed of alluvial bottomlands of the Mississippi River. This river overflow area is defined as wetlands for jurisdictional purposes under Section 404 of the CWA. Boundaries correspond with Louisiana Department of Environmental Quality Hydrologic Segments 0809-0812, which agree closely with US Geological Survey (USGS) Hydrologic Units 0805001-0805003. The basin comprises approximately 1 million ha (Figure 1), an area large enough to sustain a viable population of black bear (Nowak 1986). Thus the choice of boundaries did not preclude the option of managing the basin for the largest endemic mammals.

The climate of the study region is characterized by mild winters and warm summers. Temperatures range from an average of 9° C in winter to 27° C in summer, with an annual average of 18° C. Rainfall averages 132 cm/yr and is heaviest during winter and spring months, coincidental with peak flows of the Mississippi River. Historically during high Mississippi River stages, local rivers backed up, flooding much of the Tensas basin (Lower Mississippi Region Comprehensive Study Coordinating Committee 1974). Man-made levees prevent this backwater flooding today.

The relatively flat and poorly drained land is typical of the topography of the Mississippi River alluvial plain. Soils in the eastern portion of the basin were formed recently by the Mississippi River. Those in the western portion were derived from older sediments of the Ouachita River. Under natural conditions, these highly fertile soils, primarily of the Sharkey soil series, support vigorous forest stands. Lower Mississippi River *floodplain forests are among the most* productive fish and wildlife habitats in the United States (Glasgow and Noble 1971, US Army Corps of Engineers 1974, US Department of Agriculture n.d.). They are located in the Mississippi Flyway and are therefore important to migrating, wintering, and resident bird populations (Bellrose 1980).

The study area is characterized by extensive agricultural and timber resources. Although Sharkey soils are highly fertile, they are difficult to farm. Drainage is necessary to establish crops (US Army Corps of Engineers 1984). Historically, more than 90% of the study area was forested wetland. However, with improved technology and federal economic incentives, large areas of forest have been cleared and converted to production of soybeans, cotton, rice, and corn (Stavins 1986).

The Tensas basin contains one of the largest remaining tracts of forested wetland in the Mississippi Valley, the 50,000-hectare Singer Tract. *This land, owned by the Chicago Mil*l and Lumber Company, remained an old-growth forest until the 1940s when the last portion was logged. The cutover area has since been managed for timber production, and a second growth forest has developed. However, this land is currently being leased in small parcels (200–400 ha) to local agricultural interests. Recently, the federal government purchased part of the tract to establish the 20,000-hectare Tensas River National Wildlife Refuge (Figure 1). In addition, Louisiana acquired 7779 ha adjoining the refuge and established the Big Lake Wildlife Management Area (US Army Corps of Engineers 1984). Besides the National Wildlife Refuge and the Big Lake Wildlife Management Area, relatively few tracts of land are publicly owned. In all, public areas total approximately 44,000 ha. Within them, forests are harvested commercially, but no land is cleared.

Ecological characterization. We used a number of indices to characterize the Tensas basin at a landscape scale. These indices, a subset of those suggested by Gosselink and Lee (1989), were based on forest structure (and land use), stream stage/discharge, water-quality records, breeding bird surveys, and Christmas bird counts. They employ historical data sets (ap

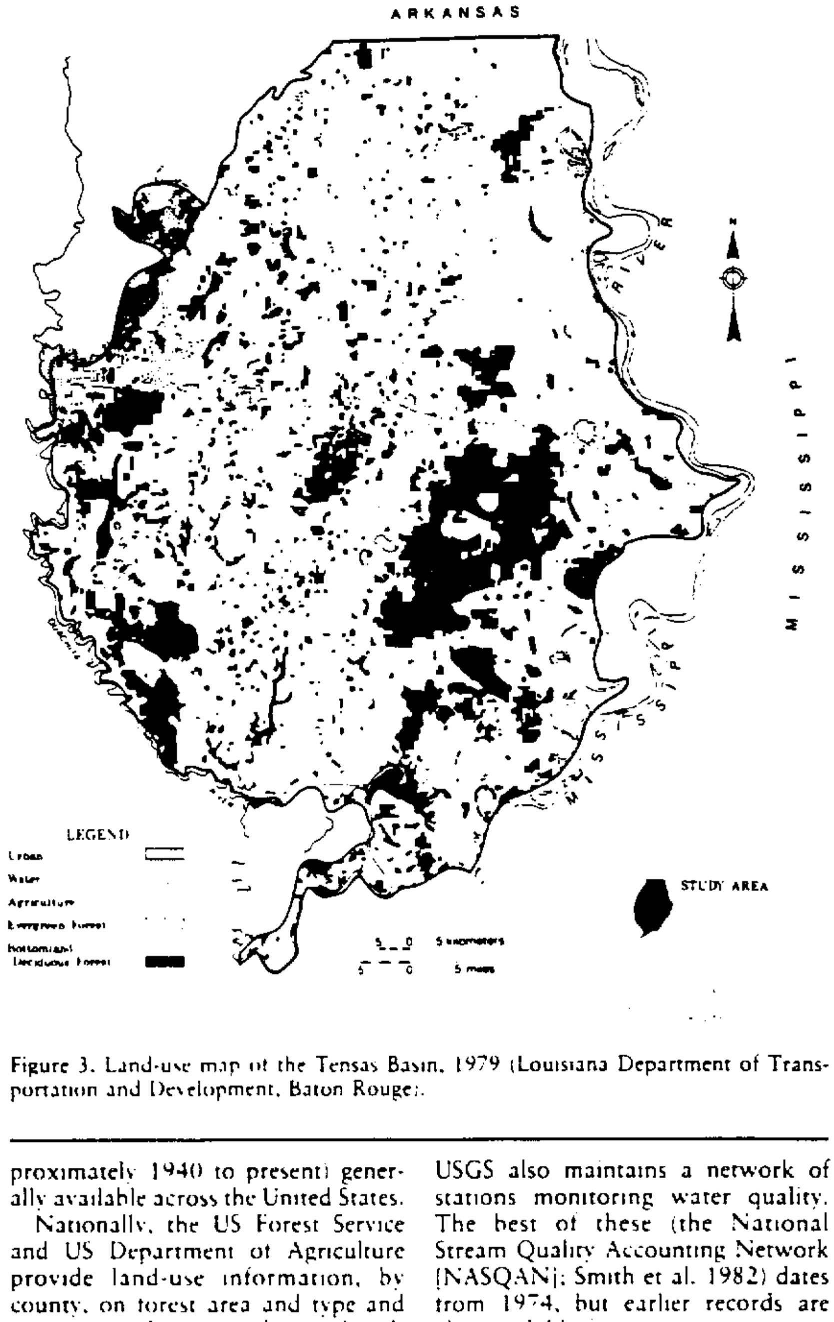


Figure 3. Land-use map of the Tensas Basin, 1979 (Louisiana Department of Transportation and Development, Baton Rouge).

proximately 1940 to present) generally available across the United States.

Nationally, the US Forest Service and US Department of Agriculture provide land-use information, by county, on forest area and type and on crop production and area. In addition, many states have recently produced maps that display land-use information.

Records of stream stage and discharge, collected by the ACE and USGS, cover extensive areas throughout the United States (Hutchinson 1975, 1977). Records for some streams provide continuous data from the turn of the century; other records cover 50-year periods. The USGS also maintains a network of stations monitoring water quality. The best of these (the National Stream Quality Accounting Network [NASQAN]; Smith et al. 1982) dates from 1974, but earlier records are also available for some stations.

In general, long-term data on biota are scarce. The best records are of bird abundance: the US Fish and Wildlife Service breeding bird surveys (Bystrak 1981) and the Audubon Society Christmas bird counts (Drennan et al. 1985). Breeding bird surveys have been conducted since 1967, whereas some Christmas bird count records go back 85 or more years. All these data sets were appropriate for this prototype study, because they provide historical records of structural and functional changes in the basin that generally reflect landscape-level processes rather than site-specific ones. However, these data sets may be short, fragmented, or spatially incomplete. Additional data sets, collected by local agencies, may be available in some localities.

Forested wetland area and distribution. In the Tensas basin, approximately 10,000 ha of forest have been cleared per year since 1937. Only approximately 157,000 ha remain, roughly 15% of the original forested area (Figure 2). Most of this forested area is in four patches, each 10,000–30,000 ha, in the east-central part of the basin (Figure 3). The largest contains the Tensas National Wildlife Refuge and the state Big Lake Wildlife Management Area. There are four additional patches between 3000 and 10,000 ha. The rest of the forested area is in small patches (mostly less than 300 ha) widely scattered throughout the basin (Figure 4).

This forest patch distribution contrasts sharply with the situation in 1957, when two large patches accounted for 326,500 ha of the total 560,000 ha then forested (Figure 4). In the intervening period, the large forest patches were fragmented by conversion to agriculture, and small patches disappeared for the same reason. As a result, the total number of patches remained at approximately 500, most of them less than 300 ha in size.

Fifty-five percent of the stream edges were forested in 1957, and less than 15% by 1987 (Figure 5). This figure does not show dimensions less than 250 m, and narrow forested borders were probably classified as agricultural fields. However, the error introduced is probably minimal; on-site observations and detailed analysis of several 1:24,000 photo-ortho quad maps revealed that most agricultural fields in the basin extend to stream banks without any intervening forested strip.

Hydrology. Throughout the Tensas basin, public works projects have generally increased runoff efficiency. This efficiency is shown by LANDSAT imagery from the peak flood time in January 1983 (the highest river stage of the decade), which

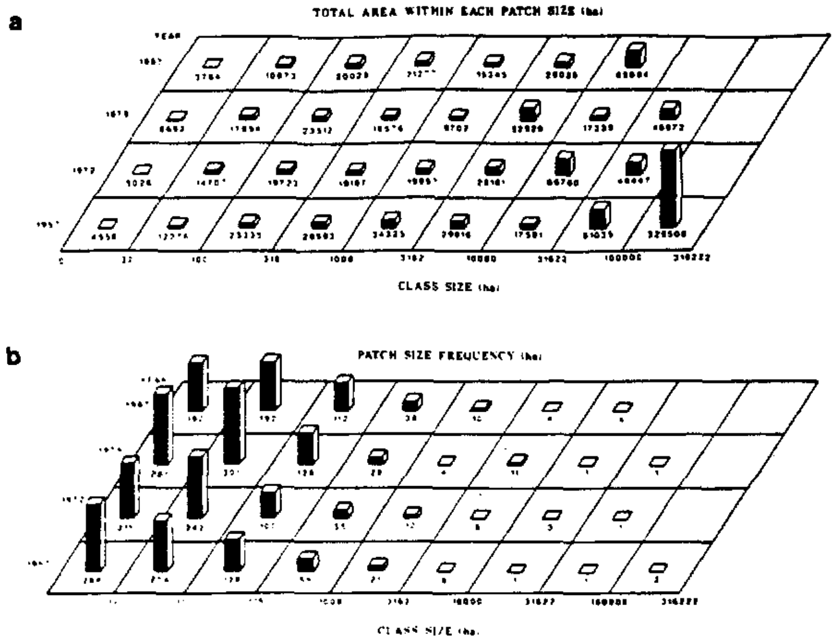


Figure 4. Bottomland forest (a) patch size and (b) frequency distribution, 1957–1987. The horizontal axis groups the patches by size classes, on an exponential scale (e.g., $10^{1.5}$ ha, 10^{2} ha, . . . $10^{5.5}$ ha). The height of each bar is proportional to the area within each size class (a) or the number of patches within a size class (b), which is given below each bar. See Gosselink et al. (1989) for details of methodology.

shows no significant flooding anywhere in the basin, except in the southern portion along the Ouachita River.

We analyzed river-stage records and associated discharge data from the late 1930s to the present for the Boeuf River at Girard, Bayou Macon at Delhi, and the Tensas River at Tendal (Figure 1). The stage/discharge characteristics of the three major streams have changed significantly during the past 50 years (Figure 6).

The stage/discharge relationship is an index of the stability of a stream

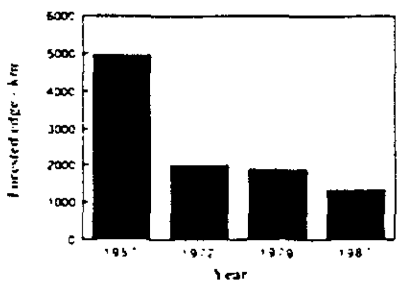


Figure 5. Temporal changes in the length of streams bordered by forest in the Tensas Basin, Louisiana. See Gosselink et al. (1989) for methodology.

system. It tends to be relatively constant in undisturbed watersheds (e.g., Belt 1975). In the Tensas basin, each stream behaved differently, probably reflecting differences in the structural management of the three streams rather than effects of forest clearing.

Bayou Macon is an example of increased hydrologic efficiency, probably the most common change in the Tensas basin. The effects of channel improvements initiated in 1957 and continuing until 1967 are clearly seen at the gauge (Figure 6b). Beginning in 1963, peak discharge increased sharply, whereas peak stage appears to have decreased slightly (Figure 7). There is no indication that runoff from the increasingly cleared watershed increased during 1953–1962. Therefore, the large increase in peak discharge, beginning in 1963, is more likely an indication of a hydrologically more efficient stream whose flood peaks are sharp and short. As a result of these changes, bottomland flooding in the Bayou Macon subbasin is now shorter and more sporadic than it was in the past.

STREAM WATER QUALITY. Except for data on turbidity, long-term stream water-quality records from the basin are lacking. Monthly total Kjeldahl nitrogen and total phosphorus data from stream water samples, taken at the same locations as the water-level gauges, were available for 1978–1986 from the USGS benchmark and NASQAN programs. These data were analyzed by using techniques for time-series analyses described by Hirsch et al. (1982; see Childers and Gosselink 1990 for details).

We chose to examine phosphorus and nitrogen, because they are key nutrients involved in primary production and excellent indicators of stream eutrophication (Lund 1965). Phosphorus is a particularly good index of water quality for several reasons. First, it is the most common nutrient limiting aquatic plant growth in freshwater systems (Hecky and Kilham 1988, Hutchinson 1957, Kuhl 1962). Second, as a common fertilizer constituent, it is often a good index of agricultural disturbance. Finally, soluble inorganic phosphorus is quickly adsorbed to soil particles and immobilized, making it a good indicator of erosion from the adjacent

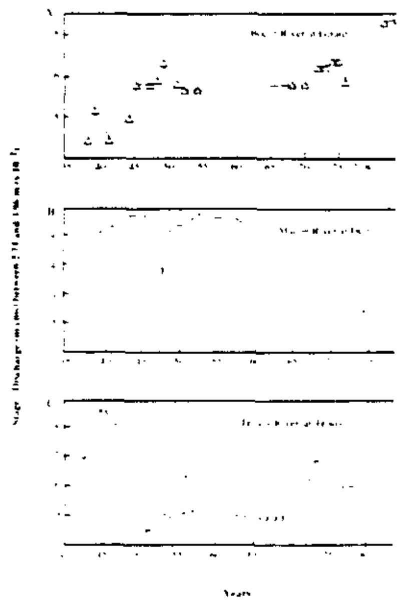


Figure 6. Changes from 1935 to 1985 in the slopes of discharge rating curves between the 2.74 and 3.96 m river water level (relative to the US Army Corps of Engineers datum), at Bayou Boeuf, Macon River, and Tensas River sites. The water-level range reflects discharge rate when overbank flooding begins (cms, cubic meters per second).

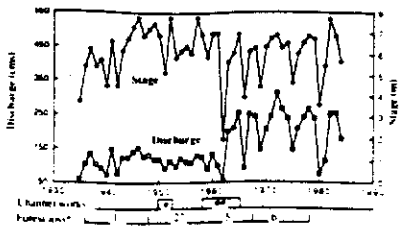


Figure 7. Maximum discharge and stream water level (stage), 1940–1985, Bayou Macon at Delhi, Louisiana, with associated structural changes in the watershed. (#, snagging and clearing; ##, channel improvements; *, percent of total area of East and West Carroll parishes; cms, cubic meters per second).

watershed (Smith et al. 1987, Wetzel 1975).

Unfortunately, the short period of record precluded meaningful time-series analysis of phosphorus. Instead, we used turbidity data that had been collected since 1958 at the same three stations. Because of the adsorption of phosphorus to soil particles, the relationship between turbidity and total phosphorus is fairly close (Hirsch et al. 1982). In data from the three streams, 39%–57% of the variability in phosphorus can be attributed statistically to turbidity (Figure 8).

To filter out the variation in turbidity caused by differences in seasonal discharge, which might otherwise obscure long-term trends, we performed a regression analysis of turbidity against water stage (a proxy for discharge; Figure 9) and plotted the residuals against time (Figure 10). We show nutrient data for Bayou Macon only; data from the other two streams are similar. The residuals represent turbidity concentrations independent of flow (flow-adjusted turbidity; Hirsch et al. 1982).

The seasonal range of flow-adjusted turbidity values has increased since 1960, but the long-term slope is flat (Figure 10b). Absolute turbidity concentrations have increased during the same period (Figure 10a), although the regression accounts for only 2% of the variation in the data. This difference between the slopes of the flow-adjusted and the absolute data indicates that turbidity increases are primarily due to hydrologic changes (e.g., runoff and stream modifications) in the basin above the sampling site.

Phosphorus concentrations in excess of 0.1 mg/l are associated with predictable biotic community changes in running streams (i.e., eutrophication; Mackenthun 1973, US EPA 1976). Phosphorus has exceeded 0.1 mg/l in 96% of the approximately 90 samples taken from all three streams in the past eight years. On the basis of the work of Omernik (1977), who showed that phosphorus levels usually exceed 0.1 mg/l when more than 50% of the watershed is disturbed, these elevated concentrations are not surprising, because approximately 75% of the watershed had already been cleared when phosphorus sampling was initiated (1978).

The positive slope of turbidity as a function of stream stage also indicates stream eutrophication (Figure 9). A positive slope represents an enrichment phenomenon that accompanies erosion from cleared areas of the watershed. In forested watersheds, erosion is minimal, and turbidity often decreases with increasing discharge, because suspended particles are diluted by the large water volume (Hirsch et al. 1982).

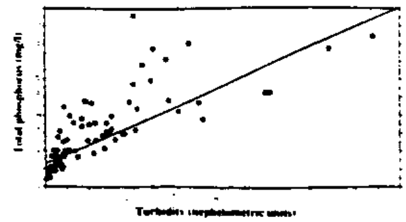


Figure 8. The relationship of total phosphorus to turbidity (nephelometric turbidity units) at Bayou Macon (total phosphorus = 0.0001 [turbidity] + 0.175; r^2 = 0.58, $p < 0.01$).

Table 1. Changes in bottomland hardwood forest-bird abundances from breeding-bird surveys (Route #23, Mer Rouge, LA, 1967–1985) and Christmas bird counts (Ouachita Parish, LA, 1930–1983)

	Breeding bird surveys				Christmas bird counts	
		Abundance changes*				
Bird groups	Number of species	+	0	−	Number of species	Abundance changes[†]
Passerines	27	3	15	9	29	−4
Woodpeckers	4	0	2	2	7	−1
Waterbirds	5	0	5	0	3	0
Hummingbirds	1	0	1	0		
Raptors					4	−1
Vultures					2	−1
Total	37	3	23	11	45	−7

* + = increasing with time ($p < 0.1$); 0 = no significant change; − = decreasing with time.
[†] − = linear regression ($p < 0.1$) indicating species decline with decreasing forest.

BIOTA. During this century, a number of species endemic to the Tensas basin have become locally extinct: the red wolf (*Canis lycaon* Schreber), Florida panther (*Felis concolor* Coryi.), ivory-billed woodpecker (*Campephilus principalis*), and perhaps Bachman's warbler (*Vermivora bachmanii*). Local extinctions are continuing. We analyzed breeding bird surveys[5] conducted for 14 of the years between 1967 and 1985 by the same observer and Christmas bird counts[6] from a site in Ouachita Parish for 1930–1935 and 1973–1983 (Burdick et al. 1989).

Breeding-bird survey data show a significant decline of between three and four species per decade in the number of bird species that use forests. The breeding-bird survey data also reveal that 11 bird species declined in density out of a total of 37 bottomland forest-dependent species for which there were enough sightings to analyze trends (Table 1). Those species declining included such area-dependent species as the pileated woodpecker (*Dryocopus pileatus*), red-headed woodpecker (*Melanerpes erythrocephalus*), Acadian flycatcher (*Empidonox virescons*), great-crested

[5] Obtained from US Fish and Wildlife Service, Laurel, MD. Birds seen or heard were counted for exactly three minutes at 50 stops, 0.8 km apart, along a standardized route.
[6] National Audubon Society, published annually in *American Birds*, formerly in *Bird Lore*.

flycatcher (*Myiarchus crinitus*), eastern wood-pewee (*Contopus virens*), orchard oriole (*Icterus spurius*), and red-eyed vireo (*Vireo olivaceus*). Three species increased in abundance: the fish crow (*Corvus ossifragus*), first noted in 1977 when a reservoir was constructed in the area, and two species characteristic of forested field edges, the Carolina wren (*Thryothorus ludovicianus*) and the rufous-sided towhee (*Pipilo erythrophthalmus*).

Similarly, Christmas bird count data reveal that out of a total of 45 species analyzed, 7 species declined significantly since 1930 as the forested area declined (Table 1). These species include the red-shouldered hawk (*Buteo lineatus*), red-headed woodpecker, and the white-breasted nuthatch (*Sitta carolinensis*). No species significantly increased in abundance.

To eliminate the influence of other factors correlated with wetland forest loss, we compared densities of bird species that use forested wetlands, using breeding bird survey routes with differing amounts of adjacent forest cover. Excluding three gregarious species (red-winged blackbird, common grackle, and mourning dove) that also use fields extensively, as adjacent forest area decreased there were significantly fewer forest species and lower densities of these species (Burdick et al. 1989).

Planning for cumulative impact management

In the three-step process of assessment, goal-setting, and planning, the latter two steps represent a transition from a fairly objective scientific or technical characterization of the ecological condition of a landscape to a value-laden prescription and management plan implementation for that landscape unit. Goal-setting should be an expression of public values that incorporates many considerations, including compliance with existing statutes and a balance between a healthy environment and economic development. Indeed, there is no single correct solution to a planning problem. Each plan reflects the values of those involved in its development.

In practice, setting goals and devising plans for a landscape unit are probably most successful when they involve the participation of all federal, state, and local agencies with jurisdiction; private landowners; environmental groups; and interested public citizens. On one hand, wetland regulation is hindered by private land ownership, fragmented and uncertain authority, and competing or conflicting interests (Natural Resources Law Institute 1988). On the other hand, broad participation offers the opportunity to educate and to reach consensus, using all the diverse regulatory and nonregulatory approaches available to solve planning problems.

To simulate this broad participation, we invited representatives from a number of state and federal agencies, environmental groups, and interested public citizens to a one-day planning session.* We reviewed the results of the Tensas assessment and engaged in a three-step exercise that involved determining the ecological health of the Tensas basin, setting goals for the basin environment according to its current health, and describing how those goals could be implemented. Goals and plans were developed in broad outline at that meeting and modified subsequently by the authors. We present them to illustrate the utility of this process for management of cumulative impacts.

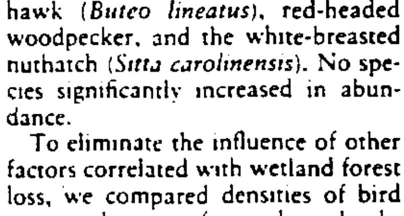


Figure 9. The relationship of turbidity (NTU, nephelometric turbidity units) to discharge at Bayou Macon (turbidity = 16.75(water level) - 35.13; $r^2 = 0.24$, $p < 0.01$). A positive slope, indicative of a disturbed watershed (Smith et al. 1982), is seen at all three sites in the Tensas Basin. Water level was used as a proxy for discharge.

*Participants included individuals from the FWS, Army Corps of Engineers, Soil Conservation Survey, US Geological Survey, EPA, Louisiana Department of Wildlife and Fisheries, Louisiana Geological Survey, several Louisiana State University departments, and two private environmental groups.

The ecological status of the Tensas basin. In setting goals, we first formally interpreted the ecological health indicators in the Tensas basin. Is the patient in bad shape? How bad? For example, if the current environment is considered healthy (i.e., current cumulative effects of human activities are not considered detrimental and the environment is adequate to support good water quality and a diverse native biota, then goal-setting should focus on protecting an appropriate level of existing resources for future generations. If, on the other hand, cumulative human effects are considered to be deleterious, then appropriate goals are to redress these impacts and restore a healthy environment.

We judged the environment of the Tensas basin to be seriously degraded, primarily by two types of activities that are both cumulative and interacting. Public works projects have reduced the area of the basin previously subject to flooding during normal spring high-water periods and bottomland forests have been converted to cropland.

Public works projects generally reduced the hydroperiod of the forests making the land more suitable for farming and stimulating bottomland conversion (McCabe et al. 1981

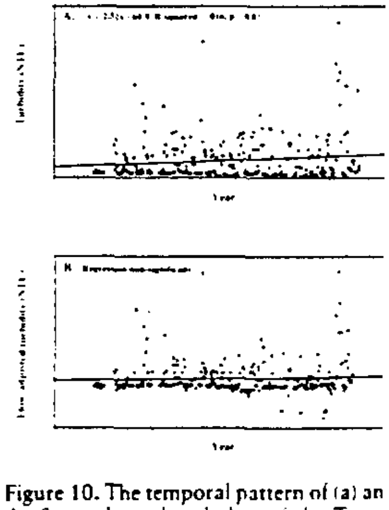


Figure 10. The temporal pattern of (a) an (b) flow-adjusted turbidity of the Tens River (NTU, nephelometric turbidit units). Flow-adjusted turbidity values ar the residuals of a regression of turbidit on water level (i.e., turbidity from whic variation due to flow has been remove see Figure 9).

Stavins 1986). This forest conversion, especially the loss of streamside buffer strips, led to poor water quality through increased erosion and fertilizer runoff from the cleared land. The highly eutrophic (US Environmental Protection Agency 1976, Mackenthun 1973) streams in the basin no longer meet the CWA goal of water quality that provides for the protection and propagation of fish, shellfish, and wildlife (33 U.S.C. §1251(a)).

Land clearing also contributed to reduction in the diversity of indigenous flora and fauna. Although black bears still exist in the Tensas Wildlife Refuge, the area of large forest patches is marginal for support of a viable population. Forest bird species with narrow niche requirements are also declining in number and population size. In sum, the area is not supporting a balanced indigenous population of flora and fauna, as called for in the CWA (several top carnivores have become extinct), but whether it can continue to support the extant species is also doubtful.

Goals for the Tensas basin. The CWA provides a strong statutory incentive for environmental goal-setting. Its goals are "to restore and maintain the chemical, physical, and biological integrity of the waters of the United States" (33 U.S.C. § 1251), and to protect "balanced indigenous populations" of shellfish, fish, and wildlife (33 U.S.C. § 1311(h) (2)). The ACE regulations for Section 404 permit review (33 C.F.R. §§320–30), temper these goals with consideration for human development needs, and require a "public interest review" based on a broad-reaching benefit:cost analysis (33 C.F.R. § 320.4). EPA's regulations for implementing Section 404, however, provide for overriding ACE decisions strictly on environmental grounds (33 U.S.C. § 1344(c)).

The goals we set for the Tensas basin were a refinement of CWA goals. They are as follows:

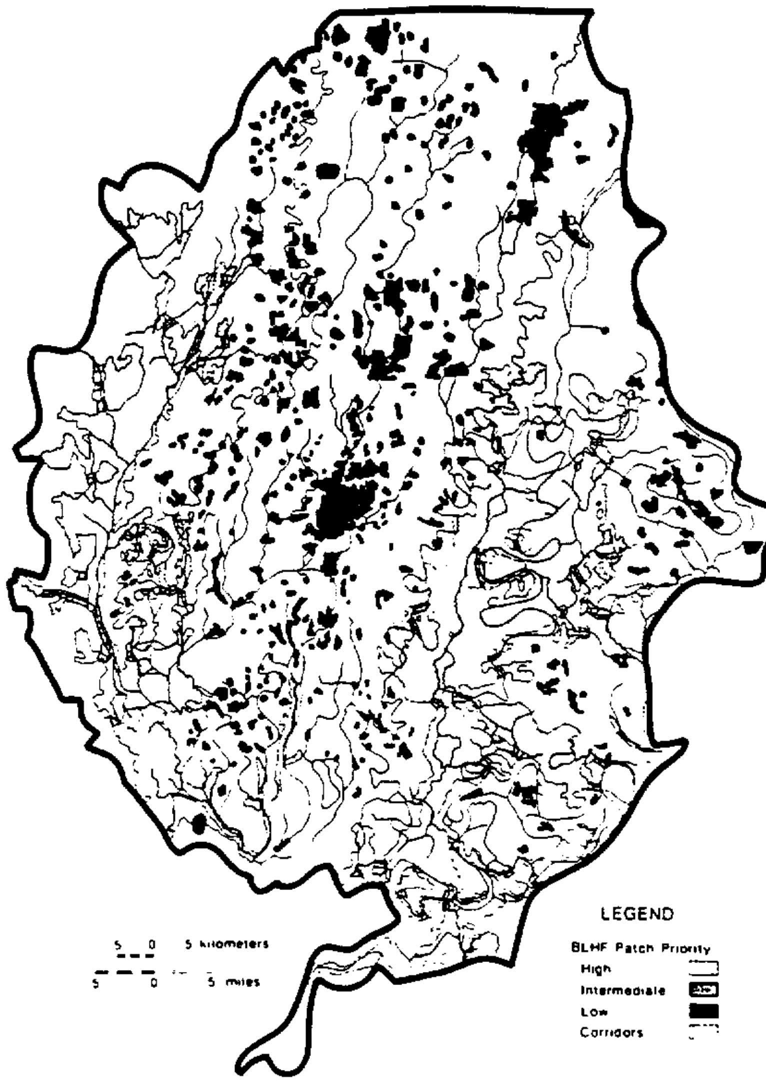


Figure 11. A priority plan for bottomland hardwood (BLHF) use in the Tensas Basin. Key elements are conservation of large forest tracts and acquisition and restoration of forested riparian corridors to increase effective patch size.

- No further net loss of forested wetlands. Of the historic forested wetlands in the basin, 85% has been converted to farmland, and this loss is related to unacceptable degradation of water quality, hydrologic function, and biotic diversity. This goal is consonant with the primary goal of "no net wetland loss" recently recommended by the National Wetland Policy Forum (Conservation Foundation 1988) and affirmed by the administration of President George Bush.
- Improve water quality to full compliance with EPA's suggested minimal standards, as indicated by phosphorus (Mackenthun 1973, US Environmental Protection Agency 1976). The current poor water quality threatens stream fauna and possibly human health, and limits recreational use of the basin's water resources.
- Return stream hydrology in the remaining large forest patches to the natural pattern of spring flooding. Changed hydrology threatens the remaining forested wetlands. Therefore, future actions should aim at restoring historic flooding patterns in forested tracts.
- Conserve existing biota, especially those species that require large forested areas and/or forest interiors. The black bear is a target species. Preserving and enhancing conditions that support a viable black bear population will also provide suitable con-

ditions for most other species now in the basin.

Implementation of goals. Goals provide a vision of the future, but unless they can be implemented they have little practical significance. In this section, we discuss strategies for implementation of the Tensas basin goals based on landscape ecology principles. We also present specific plans to show how the strategies can lead to decisions about how to use and regulate specific tracts of land.

The first goal provides an overall limit to further wetland loss and an implied mechanism for managing the landscape through wetland restoration and creation. The goals can all be approached through appropriate management of the landscape. The ecological degradation in the basin is due not only to the areal loss of bottomland forest, but also to the *pattern of that* loss. For example, forest fragmentation is a major factor in erosion of biotic diversity (Diamond 1975, Harris 1984, Margoulis and Usher 1981, May 1975), and cleared stream banks contribute disproportionately to water-quality degradation (Lowrance et al. 1984, Peterjohn and Correll 1984). Therefore, our prescriptions focus on landscape pattern and use principles based on island biogeography as applied to nature preserves.

There is an implicit assumption that if an appropriate landscape pattern is preserved, the ecological structures and processes associated with individual sites will generally also be preserved. Some impacts, such as pollutants that affect only particular species, clearly violate this assumption. But for forested wetlands, where agricultural conversion is the primary impact, the assumption seems reasonable.

Three primary landscape management strategies can be used to implement the last three goals. They are as follows:

- Conserve and restore large blocks of bottomland upland forest, appropriately interspersed with smaller tracts. This strategy would improve water quality by minimizing the dramatic increase in erosion that accompanies land disturbance (Murphree et al. 1976, Ursic 1965); maintain and improve the floodwater storage and peak stage reduction attributed to freely flooding forested wetlands (Carter et al. 1979); preserve forest area–sensitive stenotopic species by favoring them over more opportunistic generalists found in abundance along forest edges, which need less protection (MacArthur and Wilson 1963, 1967); and preserve sufficient habitat for survival of mammals and birds with large home ranges (Soulé and Wilcox 1980).
- Conserve and restore continuity between forest patches by creating or conserving forested corridors, particularly along streams. Forested stream edges improve water quality by filtering nutrients and sediments in overflow waters and in runoff from adjacent uplands (Lowrance et al. 1984, Peterjohn and Correll 1984). Protected floodplain corridors also store floodwaters, reducing the impact of downstream floods (Carter et al. 1979). This strategy also increases the effective size of forest patches by providing corridors through which animals can move safely from one small patch to another (MacArthur and Wilson 1963, 1967).
- Maintain and restore forest contiguity across the floodplain from stream to upland. This strategy reinforces the provision of linear corridors along streams to ensure protection of the whole range of bottomland zones (Larson et al. 1981), thus enhancing floral and faunal diversity and providing for safe lateral movement of animals across the bottomland from stream to upland.

Effectively implementing this general approach to management of forested wetland landscapes also requires concrete plans. Although a major objective of the approach is to improve ecological functions by enhancing the spatial pattern of forest fragments, much can also be achieved at the local level by stream-edge reforestation and crop production techniques that are minimally intrusive on the environment.

Figure 11 illustrates major spatial elements of a plan for implementing the Tensas basin goals, based on the 1979 land-use map of the basin (the most recent map available when the plan was prepared). The plan is designed to enhance forested wetlands in the two areas where they are most concentrated (Figure 3), the large east-central block of patches and the crescent of fairly large patches along the western border of the basin. These areas are not only the largest remaining forested wetland tracts, but they also present the best opportunities for enhancement of the basin ecosystem through corridor development. They fulfill Diamond's principles for conservation of biota in natural reserves (Diamond 1975); they address the habitat requirements of the black bear.

To develop the plan displayed in Figure 11, we worked outward from the largest forest patches, linking adjacent patches with corridors (most along streams) that represent primary sites for forest restoration. Expansion of the two forest patch-corridor complexes was halted when additional corridors contributed little to overall forest area in the complex. Approximately 400 ha of corridors among forest patches would increase the effective size of the largest forest complex from 50,000 ha to more than 100,000 ha. In the same way, along the western edge of the basin, 600 ha of appropriately placed corridors would form a 63,000-ha forest complex.

The plan assigns the two forest patch complexes the highest level of regulatory protection. Intermediate-level protection is assigned to forest patches of intermediate size surrounding the highest-priority areas (Figure 11). This priority follows from Diamond's principle that close disjunct patches support more species than patches farther apart. Finally, the small, isolated patches scattered throughout the basin are given the lowest priority of protection, which is that currently provided under Section 404 of the CWA. These small patches contribute little to stream water quality and hydrologic buffering, and they support primarily forest-edge and opportunistic species that are plentiful and do not need additional protection.

The goal of no net wetland loss can be implemented in the context of this plan through a number of different approaches, alone and in combination. First, Section 404 of the CWA provides a powerful tool for management, because almost all of the remaining forest is jurisdictional wetland. The advance identification provisions of the guidelines implementing Section 404 (40 C.F.R. § 230.80) allow EPA to identify, before any permit is requested, critical wetland areas (e.g., the high-priority areas of the Tensas plan) as generally unsuitable for classification as a disposal site. Because most mechanized forest clearing undertaken to convert forested wetlands to agriculture is, for legal purposes, a "discharge,"[8] this process is an important means of protecting critical wetland areas before site-specific controversies develop. EPA also has authority to override Army Corps of Engineers decisions and to deny or restrict the use of any defined area for disposal-site specification under Section 404(c). In the past, EPA has used this authority sparingly.

Second, the agencies reviewing Section 404 permit applications can also require permittees to provide restitution for unavoidable wetland loss under current regulations (40 C.F.R § 230.10(d), § 230.75(d), and 33 C.F.R. § 320.4(r)). Where it enhances implementation of this plan, restitution could be required for medium- and low-priority wetlands (assuming that the proposed activity represents the least environmentally damaging practicable alternative; 40 C.F.R. §§ Part 230); the extent of mitigation could be related to the priority ranking. This strategy would not only ensure no net overall loss of wetlands, but it could be used to secure key corridors.

Third, nonregulatory disincentives to discourage wetland forest clearing and incentives to encourage wetland forest conservation are also available. For example, the Swampbuster provisions of the Food Security Act of 1985 cause a farmer to lose nearly all federal subsidies for clearing and planting wetlands. The Conservation Reserve Program of the same act provides an incentive to conserve and enhance streamside corridors by paying farmers rent to set aside cropland as buffer strips.

Fourth, the plan could also be implemented through outright purchase of key forest sites. Public agencies own approximately 44,000 ha in the basin. These sites are not contiguous, and plan implementation should consider purchase of adjoining properties to link public parcels.

Conclusions

We have reported a specific case study undertaken to test a general approach to cumulative impact assessment and management. We applied methodology developed for forested wetlands to a specific 1-million-hectare area in the Mississippi River alluvial floodplain. This pilot study has demonstrated that an anticipatory methodology for cumulative impact assessment and management can provide a focal point for regulatory programs and foster planning to restore and protect ecosystems.

Did the Tensas basin provide a fair test of the method? Although typical of hydrologic basins of the Mississippi River alluvial floodplain, where the most rapid loss of United States wetlands is occurring, the basin is in many other aspects atypical. It is a rural area with no large population centers or intense industrial development pressure and no significant point-source discharges of pollutants. Human activities that dominate the landscape are limited to farming and large flood-control projects. Because the area is almost entirely jurisdictional wetland, regulatory authority under the CWA extends to essentially the entire basin. Most watersheds are a mosaic of wetlands and uplands, and therefore no legislation provides comprehensive regulatory authority over development.

Despite these limitations, four aspects of the methodology appear to be broadly applicable to resource planning. First, the method identifies a process of ecological characterization, goal-setting, and planning that is a general requirement for the containment of cumulative impacts, and perhaps for natural resource conservation in general (Bedford and Preston 1988, Gosselink and Lee 1989).

Second, the method focuses attention at the landscape level (i.e., watersheds and drainage basins) and bases planning on landscape ecology principles. Although a landscape focus is common in conservation ecology (Soulé and Wilcox 1980) and is widely used in Europe (Turner 1989), in the United States it evokes visions of land-use planning, an idea that contravenes cherished notions of the rights of private land ownership and is therefore political anathema. Consequently, in the United States, statutes such as the CWA and regulations pertaining to these statutes are formulated to regulate public resources such as water and air, and land-use restrictions are incidental to that focus. Our approach, conversely, implies that landscape structure (including land use) is intimately tied to ecological process, and that the most direct way to conserve public resources is by careful landscape planning (Turner 1989).

In this study, we used principles derived from insular biogeography to plan forest pattern. These principles, which relate biotic diversity to forest patch dynamics, are particularly appropriate because they appear to apply equally to protection of stream water quality and wetland hydrologic values. In other types of landscapes, other principles may need to be identified. For example, in an estuarine system dominated by bays and marshes, we know of no species whose distribution is related to large, unbroken marsh tracts. Are patch size dynamics important in this kind of system? Numerous studies indicate that hydrology, which is certainly linked to patch size dynamics, is the primary control on estuarine system processes (Clark 1974, Gosselink and Lee 1989). It is not yet known, however, what landscape management principles are appropriate under circumstances such as these.

The third broadly applicable aspect of our methodology is that relatively few widely available long-term data sets on water quality, hydrology, and biota, supplemented with land-cover data and maps, can provide the basis for an analysis at the landscape level sufficient to identify the major structural and functional changes related to human activities and to provide adequate information for a robust analysis of cumulative impacts. This reliance on a few relatively simple indices needs to be evaluated further. A simple characterization method is important, because it puts cumulative impact assessment and landscape-

[8]Avoyelles Sportmen's League v. Marsh, 715 F. 2d 897, 903 n. 12 (5th Cir. 1983).

level planning within the resources of local regulatory offices, which can complete characterizations without great analytical sophistication and excessive outlay of time and money (e.g., six months for the characterization, at less than $0.20/ha). As Bedford and Preston (1988, pp. 571–572) noted, "Improving the scientific basis for regulation will not come merely from acquiring more information on more variables. It will come from recognizing that a perceptual shift in temporal, spatial, and organizational scale is overdue. The shift in scale will dictate different—not necessarily more—variables to be measured in future [research]. . . . The goal has been to simplify without sacrificing scientific rigor."

Fourth, the pilot study shows that appropriate landscape structure can be implemented with existing regulatory and nonregulatory tools to achieve conservation goals. Implementing a cumulative impact assessment methodology, such as the one tested in the Tensas basin, requires a change in both current regulatory focus and practice, but not a qualitative change in the legal and regulatory framework governing wetland protection. In general, federal statutes (particularly the CWA) provide a clear incentive for strong environmental protection, and the regulations implementing those statutes are broad enough to provide for an anticipatory, landscape-level management strategy.

The task ahead is to identify appropriate regulatory and nonregulatory tools for different landscapes and different kinds of conservation approaches. For watersheds that are not predominantly wetlands, a number of other planning vehicles are available. At the federal level, these include the National Environmental Policy Act of 1969 and, especially, the Water Resources Planning Act of 1965. A more restricted vehicle is the Watershed Protection and Flood Prevention Act of 1954, administered by the Soil Conservation Service. But the most powerful land-use authority is vested in the state and in local zoning authorities. If broad participation from all levels of government, as well as private interests, is sought in goal-setting, these tools can be brought to bear on the planning process.

The National Wetlands Policy Forum's recommendations for a "no net wetland loss" policy stimulated a surge of interest in wetland creation and restoration. In the context of landscape planning, restitution for unavoidable wetland loss can be endorsed as a means of restoring functional integrity to the environment. This kind of mitigation is a popular concept, but it has generally been applied on an ad hoc basis in connection with approval of individual permits. Under these circumstances, the mitigated environmental losses are seldom fully redressed (Larson 1988). However, mitigation could be a powerful vehicle for implementing watershed-scale plans, and it has the potential to stem wetland areal and functional losses.

Prompt action is needed for landscape planning to be cost effective. In rapidly changing areas, such as the Tensas basin, options are quickly lost. For example, Figure 11 presents a plan for the basin as it was in 1979. Recently acquired data show that agricultural development has fragmented the largest 1979 forest patch (47,000 ha containing the Tensas National Wildlife Refuge and the Louisiana State Big Game Management Area) into four separate patches. A revised plan to restore the area will be less effective and will cost more than our plan; that is, to build a 100,000-ha contiguous forest area around these public lands will now require the acquisition and restoration of additional forest tracts and corridors.

Acknowledgments

Support for this study was provided by Louisiana State University and by the Office of Wetland Protection, Environmental Protection Agency, Washington, DC. We thank Mark M. Brinson, J. A. Miller, Tom Muir, R. Eugene Turner, and two anonymous reviewers for critical comments on a previous version of the manuscript.

References cited

Abernethy, Y., and R. E. Turner. 1987. US forested wetlands, 1940–1980. *BioScience* 37: 721–727.

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/US National Research Council, Ottawa, Ontario, and Washington, DC.

Bedford, B. L., and E. M. Preston. 1988. Cumulative effects on landscape systems of wetlands. *Environ. Manage.* 12: 561–772.

Bellrose, F. C. 1980. *Ducks, Geese, and Swans of North America*. Wildlife Management Institute and Illinois Natural History Museum, Washington, DC.

Belt, C. B. Jr. 1975. The 1973 flood and man's constriction of the Mississippi River. *Science* 189: 681–684.

Brinson, M. M., B. L. Swift, R. C. Plantico, and J. S. Barclay. 1981. Riparian ecosystems: their ecology and status. FWS/OBS-83/17. US Fish and Wildlife Service, Biological Services Program, Washington, DC.

Burdick, D. M., D. Cushman, R. Hamilton, and J. G. Gosselink. 1989. Faunal changes and bottomland hardwood forest loss in the Tensas watershed, Louisiana. *Conserv. Biol.* 3: 282–292.

Bystrak, D. 1981. The North American breeding bird survey. *Studies in Avian Biology* 6: 34–41.

Carter, V., M. S. Bedinger, R. P. Novitzki, and W. O. Wilen. 1979. Water resources and wetlands. Pages 344–376 in P. E. Greeson, J. R. Clark, and J. E. Clark, eds. *Wetland Functions and Values: The State of Our Understanding*. American Water Resources Association, Minneapolis, MN.

Childers, D. L., and J. G. Gosselink. 1990. Assessment of cumulative impacts to water quality in a forested wetland landscape. *J. Environ. Qual.* 19: 454–463.

Clark, J. 1974. *Coastal Ecosystems: Ecological Considerations for Management of the Coastal Zone*. The Conservation Foundation, Washington, DC.

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/US National Research Council, Ottawa, Ontario, and Washington, DC.

Conservation Foundation. 1988. *Protecting America's Wetlands: An Action Agenda*. Final report of the National Wetlands Policy Forum. The Conservation Foundation, Washington, DC.

Diamond, J. M. 1975. The island dilemma: lessons of modern biogeographical studies for the design of natural reserves. *Biol. Conserv.* 7: 129–146.

Drennan, S. R., et al. 1985. The 85th Christmas bird count. *Am. Birds* 39: 383–388.

Earle, J. M. 1975. Forest statistics for Louisiana parishes. Forest Service Resource Bulletin SO-52. US Department of Agriculture, Washington, DC.

Forman, R. T. T., and M. Godron. 1986. *Landscape Ecology*. John Wiley & Sons, New York.

Fraver, W. E., T. J. Monahan, D. C. Bowden, and F. A. Graybill. 1983. *Status and Trends of Wetlands and Deepwater Habitats in the Coterminous United States, 1950s to 1970s*. Department of Forest and Wood Service

Colorado State University, Fort Collins.

Glasgow, L., and R. Noble. 1971. The importance of bottomland hardwoods to wildlife. Pages 30–43 in *Proceedings of a Symposium on Southeastern Hardwoods*. US Forest Service, Atlanta, GA.

Gosselink, J. G., and L. C. Lee. 1989. Cumulative impact assessment in bottomland hardwood forests. *Wetlands* 9: 83–174.

Gosselink, J. G., G. P. Shaffer, L. C. Lee, D. M. Burdick, D. L. Childers, N. Taylor, S. C. Hamilton, R. Boumans, D. Cushman, S. Fields, M. Koch, and J. M. Visser. 1989. *Cumulative Impact Assessment and Management in a Forested Wetland Watershed in the Mississippi River Floodplain*. CEI-89-02. Center for Wetand Resources, Louisiana State University, Baton Rouge.

Harris, L. 1984. *The Fragmented Forest: Island Biogeography Theory and the Preservation of Biotic Diversity*. University of Chicago Press, Chicago, IL.

Hecky, R. E., and P. Kilham. 1988. Nutrient limitation of phytoplankton in fresh water and marine environments: a review of recent evidence on the effects of enrichment. *Limnol. Oceanogr.* 33: 796–822.

Hirsch, R. M., J. R. Slack, R. A. Smith. 1982. Techniques of trend analysis for monthly water quality data. *Water Resour. Res.* 18: 107–121.

Horak, G. C., C. Vlachos, E. W. Cline. 1983. *Fish and Wildlife and Cumulative Impacts: Is There a Problem?* US Fish and Wildlife Service, Fort Collins, CO.

Hutchinson, G. E. 1957. *Treatise on Limnology*. John Wiley & Sons, New York.

Hutchinson, N. E. 1975 and 1977. *WATSTORE User's Guide*. Open-File Report 75–426, vols. 1 and 2. US Geological Survey.

Kuhl, A. 1962. Inorganic phosphate uptake and metabolism. Pages 211–229 in R. A. Lewin, ed. *Physiology and Biochemistry of Algae*. Academic Press, New York.

Larson, J. S. 1988. Wetland creation and restoration: an outline of the scientific perspective. Pages 73–79 in J. Zelazny and J. S. Feierabend, eds. *Proceedings of the Conference: Increasing Our Wetland Resources*. National Wildlife Federation, Washington, DC.

Larson, J. S., M. S. Bedinger, C. F. Bryan, S. Brown, R. T. Huffman, E. L. Miller, D. G. Rhodes, and B. A. Touchet. 1981. Transition from wetlands to uplands in southern bottomland hardwood forests. Pages 223–272 in J. R. Clark and J. Benforado, eds. *Wetlands of Bottomland Hardwood Forests*. Elsevier, New York.

Lee, L. C., and J. G. Gosselink. 1988. Cumulative impacts in wetlands: linking scientific assessments and regulatory alternatives. *Environ. Manage.* 12: 591–602.

Lower Mississippi Region Comprehensive Study Coordinating Committee. 1974. *Lower Mississippi Region Comprehensive Study*. Vicksburg, MS.

Lowrance, R., R. Todd, J. Fail Jr., O. Hendrickson Jr., R. Leonard, and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. *BioScience* 34: 374–377.

Lund, J. W. G. 1965. The ecology of freshwater phytoplankton. *Biol. Rev.* 40: 231–293.

MacArthur, R. H., and E. O. Wilson. 1963. An equilibrium theory of insular zoogeography. *Evolution* 17: 373–387.

______. 1967. *The Theory of Island Biogeography*. Princeton University Press, Princeton, NJ.

MacDonald, P. O., W. E. Frayer, and J. K. Clauser. 1979. *Documentation, Chronology, and Future Projections of Bottomland Hardwood Habitat Loss in the Lower Mississippi Alluvial Plain*. US Fish and Wildlife Service, Vicksburg, MS.

Mackenthun, K. M. 1973. *Toward a Cleaner Aquatic Environment*. US Environmental Protection Agency, Washington, DC.

Margoulis, C., and M. B. Usher. 1981. Criteria used in assessing wildlife conservation potential: a review. *Biol. Conserv.* 21: 79–109.

May, D. M., and D. F. Bertelson. 1986. *Forest statistics for Louisiana Parishes*. Forest Service Resource Bulletin. US Department of Agriculture, Washington, DC.

May, R. M. 1975. Island biogeography and the design of wildlife preserves. *Nature* 254: 177–178.

McAllister, D. M. 1980. *Environmental Planning and Impact Evaluation*. MIT Press, Cambridge, MA.

McCabe, C. A., H. D. Hall, and R. C. Barkley. 1981. *Yazoo Area Pump Study, Yazoo Backwater Project, MS*. US Fish and Wildlife Service, Jackson, MS.

Munn, R. E. 1975. *Environmental Impact Assessment: Principles and Procedures*. John Wiley & Sons, New York.

Murphree, C., C. Mutchler, and L. McDowell. 1976. Sediment yields from a Mississippi delta watershed. Pages 1–99 to 1–109 in *Proceedings of the Federal Inter-Agency Sedimentation Conference*. Water Resources Council, Washington, DC.

National Wetlands Newsletter. 1984. Court approves settlement agreement in National Wildlife Federation v. Marsh. 6(2): 4. Environmental Law Institute, Washington, DC.

Natural Resources Law Institute. 1988. *A Guide to Federal Wetlands Protection under Section 404 of the Clean Water Act, Anadromous Fish Law Memo* 46. Lewis and Clark Law School, Portland, OR.

Noss, R. F., and L. D. Harris. 1986. Nodes, networks, and MUMs: preserving diversity at all scales. *Environ. Manage.* 10: 299–309.

Nowak, R. M. 1986. *Status of the Louisiana Black Bear*. Report for the Office of Endangered Species, US Fish and Wildlife Service, Washington, DC.

Office of Technology Assessment (OTA). 1984. *Wetlands: Their Use and Regulation*. OTA-O-206. US Congress, Washington, DC.

Omernik, J. M. 1977. *Nonpoint Source–stream nutrient level relationships: a nationwide study*. EPA-600/3-77-105. US Environmental Protection Agency, Corvallis Environmental Research Laboratory, Office of Research Development, Corvallis, OR.

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

Risser, P. G. 1988. General concepts for measuring cumulative impacts on wetland ecosystems. *Environ. Manage.* 12: 585–589.

Schonewald-Cox, C. M., S. M. Chambers, B. MacBryde, and W. L. Thomas. 1983. *Genetics and Conservation: A Reference for Managing Wild Animal and Plant Populations*. Benjamin/Cummings, Menlo Park, CA.

Simberloff, D., and L. G. Abele. 1982. Refuge design and island biogeographic theory: effects of fragmentation. *Am. Nat.* 120: 41–50.

Simberloff, D., and J. Cox. 1987. Consequences and costs of conservation corridors. *Conserv. Biol.* 1: 63–71.

Smith, R. A., R. B. Alexander, and M. G. Wolman. 1987. Water quality trends in the nation's rivers. *Science* 235: 1607.

Smith, R. A., R. M. Hirsch, and J. R. Slack. 1982. *A Study of Trends in Total Phosphorus Measurements at NASQAN Stations*. Water Supply Paper 2190. US Geological Survey, Washington, DC.

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

Soulé, M. E., and B. A. Wilcox, eds. 1980. *Conservation Biology: An Evolutionary-Ecological Perspective*. Sinauer Associates, Sunderland, MA.

Stavins, R. 1986. *Conversion of Forested Wetlands to Agricultural Uses: An Econometric Analysis of the Impact of Federal Programs on Wetland Depletion in the Lower Mississippi Alluvial Plain 1935–1984*. Environmental Defense Fund, New York.

Troll, C. 1950. Die geographische Landschaft und ihre Erforschung. *Studium Generale (Heidelberg)* 3: 163–181.

Turner, M. G. 1989. Landscape ecology: the effect of pattern on process. *Ann. Rev. Ecol. Syst.* 20: 171–197.

US Army Corps of Engineers. 1974. *Environmental Assessment in the Tensas River Basin*. US Army Corps of Engineers, Vicksburg, MS.

______. 1984. *Reevaluation Report, Tensas River Basin Excluding Bayou Macon, Louisiana*. US Army Corps of Engineers, Vicksburg, MS.

US Department of Agriculture, Soil Conservation Service. n.d. *General Soil Maps of East Carroll, Madison, Tensas, and Franklin Parishes*. Louisiana Agricultural Experiment Station.

US Environmental Protection Agency. 1976. *Quality Criteria for Water*. Ep1.2:W 29/34. U. S. Environmental Protection Agency, Washington, DC.

Ursic, S. 1965. Federal Inter-Agency Sedimentation Conference Proceedings. Miscellaneous Publication 970, US Department of Agriculture, Washington, DC.

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. *Environmental Impact Assessment, Technology Assessment, and Risk Analysis*. Springer-Verlag, New York.

Westmann, W. E. 1985. *Ecology, Impact Assessment, and Environmental Planning*. John Wiley & Sons, New York.

Wetzel, R. G. 1975. *Limnology*. Saunders College Publishing, Orlando, FL.

Wharton, C. H., W. M. Ketchens, E. C. Pendleton, and T. W. Sipes. 1982. *The Ecology of Bottomland Hardwood Swamps of the Southeast: A Community Profile*. FWS/OBS-81/37. US Fish and Wildlife Service, Biological Services Program, Washington, DC.

Yancey, R. K. 1970. Our vanishing delta hardwoods. *Louisiana Conservationist* 22: 3–4.

Ecosystem Assessment Methods for Cumulative Effects at the Regional Scale

C. T. Hunsaker, Oak Ridge National Laboratory, Oak Ridge, TN

ABSTRACT

Environmental issues such as nonpoint-source pollution, acid rain, reduced biodiversity, land use change, and climate change have widespread ecological impacts and require an integrated assessment approach. Since 1978, the implementing regulations for the National Environmental Policy Act (NEPA) have required assessment of potential cumulative environmental impacts. Current environmental issues have encouraged ecologists to improve their understanding of ecosystem process and function at several spatial scales. However, management activities usually occur at the local scale, and there is little consideration of the potential impacts to the environmental quality of a region.

This paper proposes that regional ecological risk assessment provides a useful approach for assisting scientists in accomplishing the task of assessing cumulative impacts. Critical issues such as spatial heterogeneity, boundary definition, and data aggregation are discussed. Examples from an assessment of acidic deposition effects on fish in Adirondack lakes illustrate the importance of integrated databases, associated modeling efforts, and boundary definition at the regional scale.

INTRODUCTION

Effective management of our natural resources requires a holistic approach to environmental assessments. Since 1978, the implementing regulations for the National Environmental Policy Act (NEPA) have required assessment of potential cumulative environmental impacts. Cumulative impact assessment, effects assessment for programmatic environmental impact statements (PEIS), and ecological risk assessment share some common goals and needs when applied to large geographic areas or regions. In the United States, a region can range in size from an area the size of several counties to several states. A region should contain a certain degree of homogeneity with respect to the characteristics used to define it (de Blij 1978). The goals of these assessments include making informed decisions and protecting or managing the environment at large geographic scales. The needs of cumulative, programmatic, and risk assessments at the regional scale include (1) regional and national integrated databases, (2) monitoring that characterizes conditions at several spatial scales, (3) quantified relationships between landscape structure and function, (4) mechanistic understanding of the controls on landscape functions at several spatial scales, and (5) models for several spatial and temporal scales.

The common goals of assessments for cumulative impacts, PEISs, and ecological risk are to make informed decisions and to protect or manage the environment for large geographic areas. A cumulative impact assessment should qualitatively or quantitatively assess "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" (CEQ 1978). The PEIS is appropriate for general matters or related actions that are similar in nature or broad in scope and have cumulative impacts (Sigal and Webb 1989). Myslicki (this volume, Chap. 5) points out the advantages of using PEISs to look at cumulative impacts. A regional risk assessment should evaluate the aggregate influence of multiple disturbances on the total resource as bounded by the region of influence for the hazard of interest. Risk assessment goes beyond a cumulative or programmatic assessment in that it must quantify the probability of impact and the associated uncertainty. Thus, a regional ecological risk assessment is the extreme quantification of a cumulative or programmatic assessment and represents what assessments should be striving to achieve.

This paper proposes that a regional ecological risk assessment provides a useful approach for assisting scientists in accomplishing the task of assessing cumulative impacts. A risk assessment approach is independent of scale (i.e., the components of the assessment are developed for the appropriate space and time scales of each individual assessment). The Canadian Environmental Assessment Research Council and the U.S. National Research Council (1986) stated, "neither scientists nor institutions are working at the temporal and spatial scales needed for the assessment of cumulative effects." Thankfully, this statement is no longer true; however, often the research and analyses are being developed in different disciplines. The theory and analyses from landscape ecology (Turner 1989) and research and tools from geography, both of which focus on spatial scale, when combined with a risk assessment approach, hold great promise for future NEPA assessments. This paper discusses in detail the importance of defining the regions/subregions for assessment; this activity relates to our need to quantify relationships between landscape structure and function and to understand the mechanisms that control landscape functions at different spatial scales.

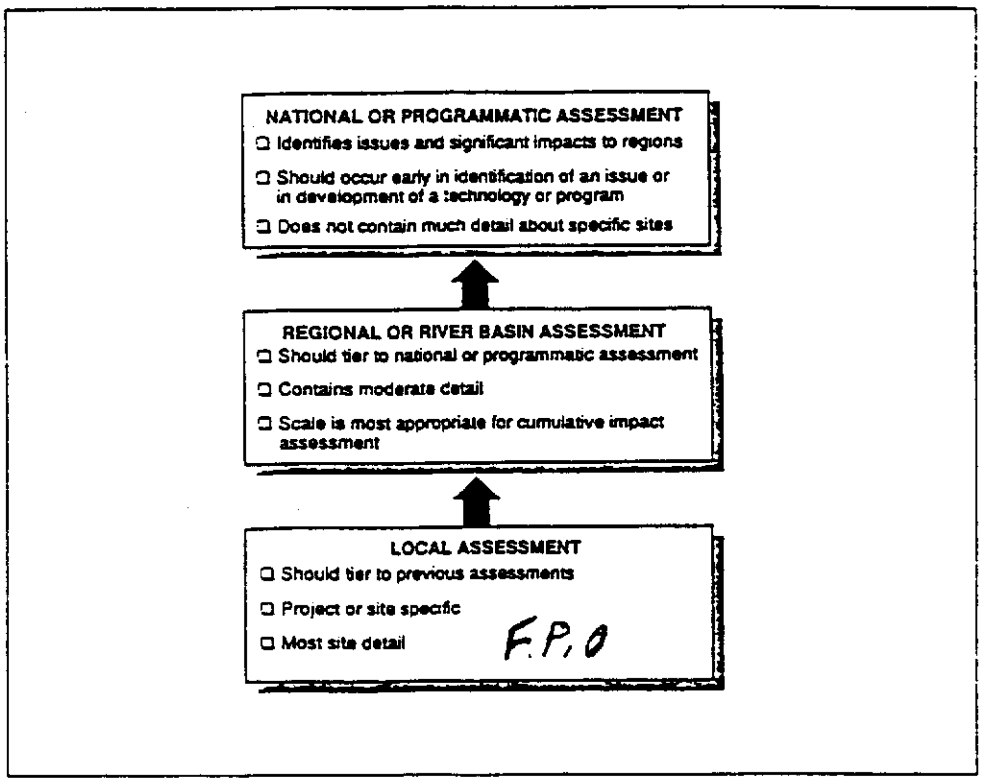


Figure 1. Relationships between national or programmatic, regional, and local assessments. Arrows show the direction for tiering between these assessments.

CUMULATIVE IMPACT ASSESSMENT APPROACH

Spatial and temporal scales are important to the understanding, analysis, and management of cumulative effects. As with regional risk assessments, adequate cumulative impact assessments require an understanding of the contributions to assessment uncertainty from boundary definition (geographic area for assessment), data resolution and aggregation, and spatial heterogeneity of a resource within the assessment area. Cumulative impacts are best addressed at the regional scale, while both national and regional scales are appropriate for programmatic issues (Figure 1). Often cumulative impacts are too complex or extensive to adequately address in most local assessments except in a qualitative manner, unless a regional assessment is available from which to tier. National or programmatic assessments, because of their breadth and often a lack of integrated databases, cannot be expected to address cumulative impacts in much detail (Cada and Hunsaker 1990; FERC 1988a). Several environmental impact statements (EIS) for hydropower development illustrate the ability to quantify cumulative effects at a regional or river basin scale (FERC 1985a, 1985b, 1986, 1988a, 1988b).

The approach outlined for regional ecological risk assessment (Hunsaker et al. 1990) can be used for cumulative impact assessment and programmatic assessments for large geographic areas. Risk assessments can be thought of as having two distinct phases: the definition phase and the solution phase (Figure

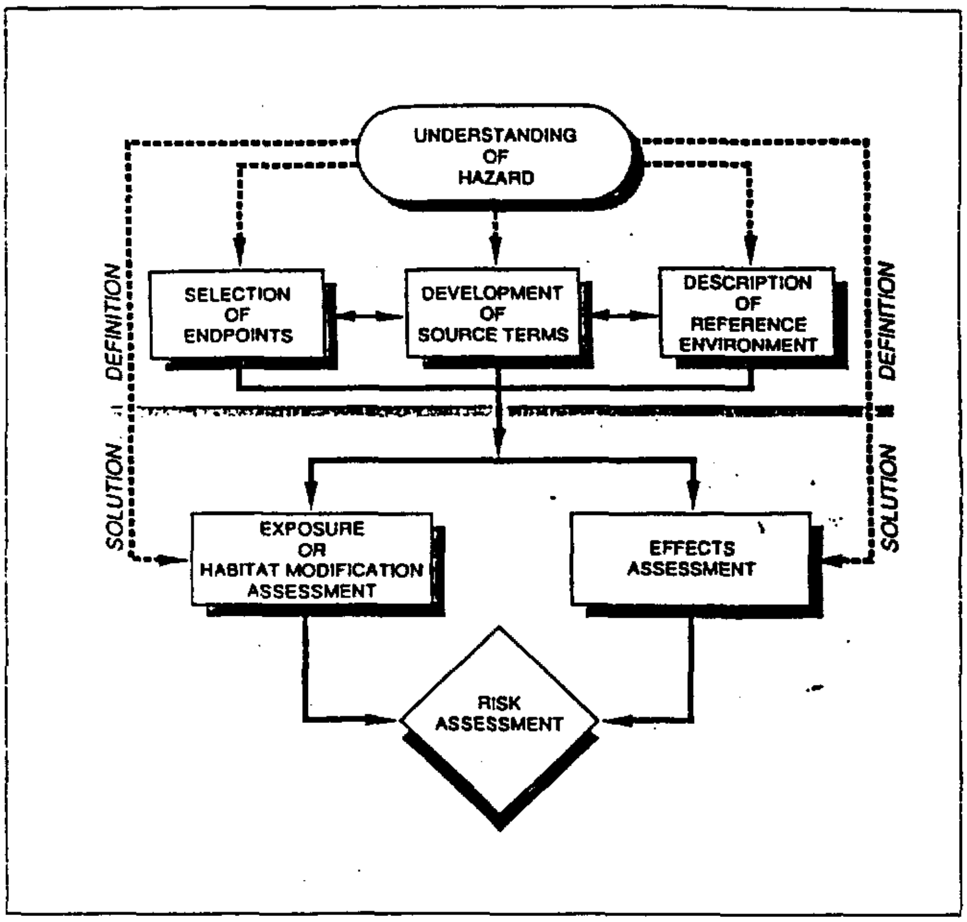


Figure 2. The two phases of regional risk assessment: the hazard definition or scoping of the problem and the problem solution. (From Hunsaker et al. 1990. Regional ecological risk assessment. *Environ. Manage.* 14(3):325–332. With permission.)

2). In the definition phase, the endpoint (entity and its quality of concern), source terms (source and associated magnitude of hazard), and geographic area for assessment (reference environment) are defined. The definition of these three elements should be an iterative process, and the understanding of the hazard should take into account not only the ecological processes of interest, but also the social, economic, and institutional processes significant to the hazard. In the solution phase, exposure and effect are assessed and then combined to determine the risk or probability of a negative event happening.

SELECTION OF REGIONS AND SUBREGIONS FOR ASSESSMENT

For any cumulative effects assessment, the assessor should consider the contributions to assessment uncertainty from boundary definition, data resolution and aggregation, and spatial heterogeneity. For risk assessment, such uncertainty should be quantified. The importance of boundary definition, selection of regions and subregions in this paper, is discussed in detail because appropriate definition of the assessment region can reduce uncertainty; it is also related to data resolution, data aggregation, and spatial heterogeneity. Throughout the discussion, points are illustrated with examples from a demonstration assessment of atmospheric pollutant effects on aquatic ecosystems. The ability to provide such a refined example is possible because of years of research and analyses by many scientists, primarily funded by the National Acid Precipitation Assessment Program (NAPAP 1990).

Demonstration Data and Models

Recent international agreements for controlling atmospheric pollutants have focused on methods to identify and map the distribution and amounts of deposition of atmospheric pollutants that do/do not cause significant harmful effects on the environment (critical loads) (CLRTAP 1989). The United States has emphasized the need to develop critical loads for appropriate geographic areas (i.e., regions/subregions whose endpoints or resources of interest respond in a similar way to the hazard of interest). The examples used in this paper come from that effort to develop and demonstrate an assessment approach for determining and mapping critical loads in the United States. In these examples, the assessment region or reference environment is the Adirondack region in New York State. The endpoint is the proportion of lakes with brook trout, and the source of the acidic deposition hazard is sulfate deposition. The sulfate exposures are for current deposition and 50% of current deposition. The lake acidification model results shown are from an empirical, steady-state model (Henriksen 1984). The probability of fish presence was predicted from surface water pH using an empirical fish response model. These predictions assume that brook trout once existed in the lakes (Baker et al. 1988, 1990).[1]

Three data sets of lake water chemistry are used to evaluate the robustness of the assumption that the variability between the response of lakes to a hazard within a subregion is smaller than between subregions. The Eastern Lake Survey (ELS) (Linthurst et al. 1986) database provides a statistically derived population for lakes ranging from about 4 ha to 2000 ha. In ELS, 128 lakes occur in the Adirondacks. The Direct Delayed Response Project (DDRP) (Turner et al. 1990a) was developed within the statistical sampling frame of the ELS and was designed to project the long-term effects of specified levels of sulfur deposition on a sensitive subset of the ELS. Thirty-seven DDRP lakes occur in the Adirondacks. The Adirondack Lakes Survey (ALSC 1989) has 1280 lakes and is a census of Adirondack lakes 1 ha in size and larger. Using lake location, each lake in each

---

[1] Model development has been a major part of the 10-year NAPAP effort. Models have been extensively applied to assess the regional effects of acidic decomposition (NAPAP 1990; Turner et al. 1990b; Sullivan 1990; Baker et al. 1990; Thornton et al. 1990). In particular, watershed models have been used both alone and in combination with fish response models to protect changes in water chemistry (Thornton et al. 1990) and in the suitability of waters for fish (Baker et al. 1990) resulting from deposition-driven changes in acid-base chemistry. These analyses have been performed for relatively large regions like the Adirondack Mountains. project

database was assigned to its appropriate subregion, and cumulative frequency distributions for each subregion and each database were developed for the probability of fish being present.

Selection of Assessment Subregions

The influence of different databases (resource populations) and current and predicted data distributions for a subregion scheme are discussed with regard to selecting subregions for cumulative effects assessments. I use several existing subregion schemes to illustrate how one can evaluate the appropriateness or usefulness of subregions for an assessment. Regions are divided into subregions to improve the results of the assessment and provide geographic perspective for the policy maker. Three possible ways to divide the region (Adirondacks in the example) into subregions are illustrated using field data (Hunsaker et al. 1986) (Figures 3 through 5).[2]

Aquatic studies classically use the watershed as a physiographic unit of assessment. The Adirondacks can be divided into three large river basins (Figure 3). The Upper Hudson River and the Lake Ontario–St. Lawrence River basins are different with respect to fish presence for all three data sets. The proportion of lakes with a high probability of fish presence is much less for the Lake Ontario–St. Lawrence River basin. This probably results from the basin coinciding with the region of high sulfate deposition in the western Adirondacks. For the purposes of this example, differences in data distributions are determined in a qualitative manner; in actual practice, the cumulative frequency distributions would be drawn with confidence bands to determine statistical differences in distributions. High-elevation lakes tend to have lower pH values and thus are less likely to have trout present (Hunsaker et al. 1986). A useful subregion boundary occurs at the 600–m elevation contour for the DDRP and ELS databases (Figure 4); however, this relationship does not hold for the ALSC. Since the effect of acid deposition on lakes can be affected by soil processes, soil groups provide another logical group of subregions. The haplorthods-haplaquods and cryorthods-cryaquods show different patterns for the endpoint of fish presence for the ELS and the ALSC. The latter are cool soils with high elevations and less buffering capacity; thus, they have a high probability of having low pH lakes and no trout. Soils have complex patterns and the problem of having too many subregions for available data is also illustrated in Figure 5 with the DDRP database.

Data and subregion definition can both contribute to analytic uncertainty. The results of an effects model are likely to be different when different populations of resources are used, such as the different data sets for lakes (ALSC, ELS, and

---

[2] In Figures 3 through 5, pH and fish presence are presented as cumulative proportions. Cumulative frequencies have been converted to cumulative proportions to facilitate comparisons between data sets and subregions. The curves depict the proportion of lakes having a probability of pH of fish presence of x or less. To read a curve for a given subregion using a given data set, pick a value on the horizontal axis and read the proportion of lakes on the vertical axis with a probability of x or less.

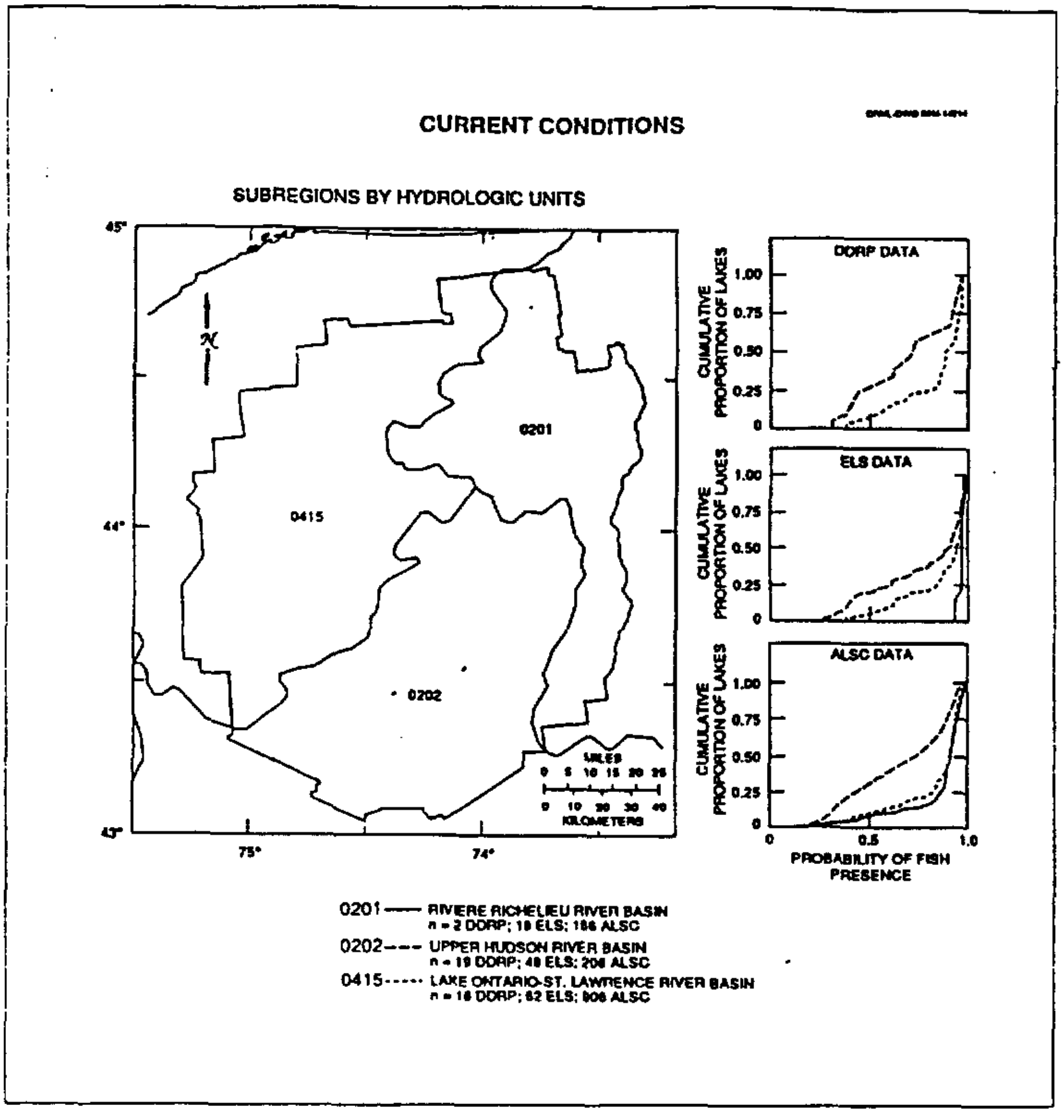


Figure 3. Current probability of brook trout being present in Adirondack lakes within watershed subregions according to major river basins.

DDRP). For example, using the river basins as subregions, the Henriksen model predicted fairly different subregional responses to deposition loads for each database (Figure 6). Model results using lake populations from ELS and ALSC predicted that lakes in the Lake Ontario basin would have a higher probability of fish being present under a 50% deposition reduction from current levels than lakes in the other two basins. When DDRP lakes were used, the model predicted the opposite. For current deposition, the model predicts the Lake Ontario basin to have the highest probability of fish presence for all the lake populations.

The subregion schemes captured spatial differences in response under different deposition scenarios with varying degrees of success. This point is illustrated using the Henriksen model and the ALSC database. This database contains the largest number of lakes, and the Henriksen model could be applied to this data set. The soil order and the river basin subregion schemes best captured spatial

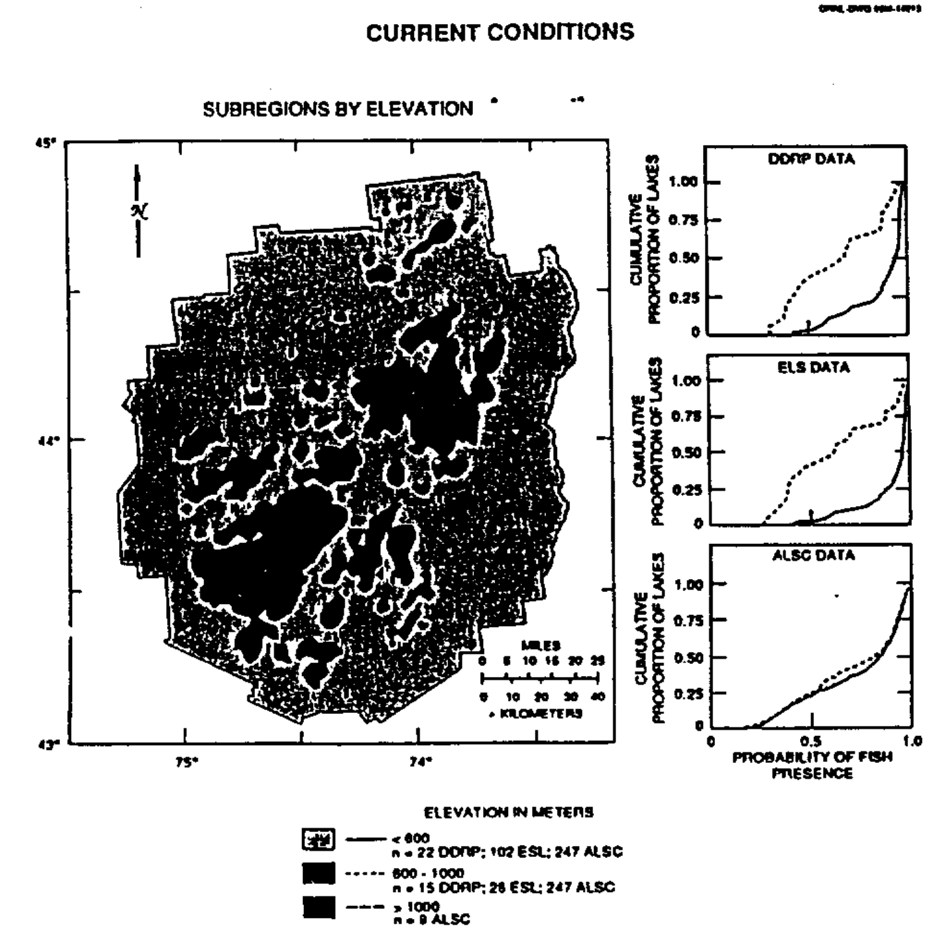


Figure 4. Current probability of brook trout being present in Adirondack lakes within elevation subregions.

differences in response as shown by the vertical separation of the cumulative frequency curves for the subregions (Figure 7). In an actual assessment, confidence bounds would be calculated and graphed for the cumulative frequency distributions and would be used to determine if results were significantly different for different lake populations or subregion designations. For the elevation subregion scheme, lakes in the low- and medium-elevation subregions had very similar response patterns to deposition scenarios for the Henriksen model and the ALSC database. This lack of distinction is supported by the field data for the ALSC; however, field data for ELS and DDRP show a distinction between elevation subregions. If resources in a subregion are not responding differently to an exposure than resources in an adjacent subregion, there may be no reason to keep separate subregions. Thus, one can conclude that selection of appropriate subregions for

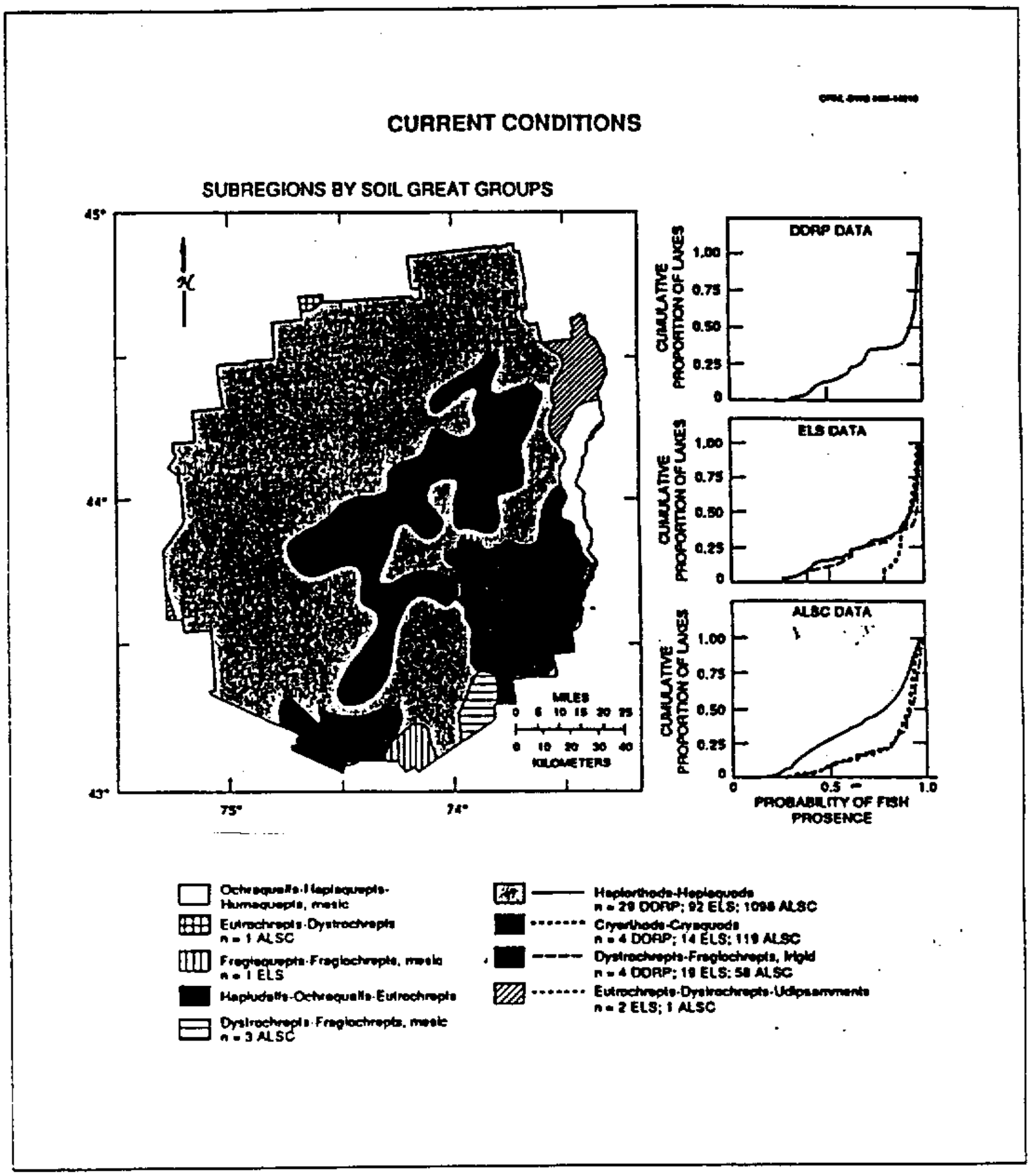


Figure 5. Current probability of brook trout being present in Adirondack lakes within soil subregions. Cumulative frequencies are not shown for subregions with less than ten lakes.

an assessment can differ depending on the databases and data type (field monitoring vs model predictions) used. Confidence can be increased in a subregion scheme if it is supported by both field data and model predictions.

DISCUSSION AND CONCLUSIONS

The approach as outlined by Hunsaker et al. (1990) for regional ecological risk assessment is useful for scoping and performing both cumulative and

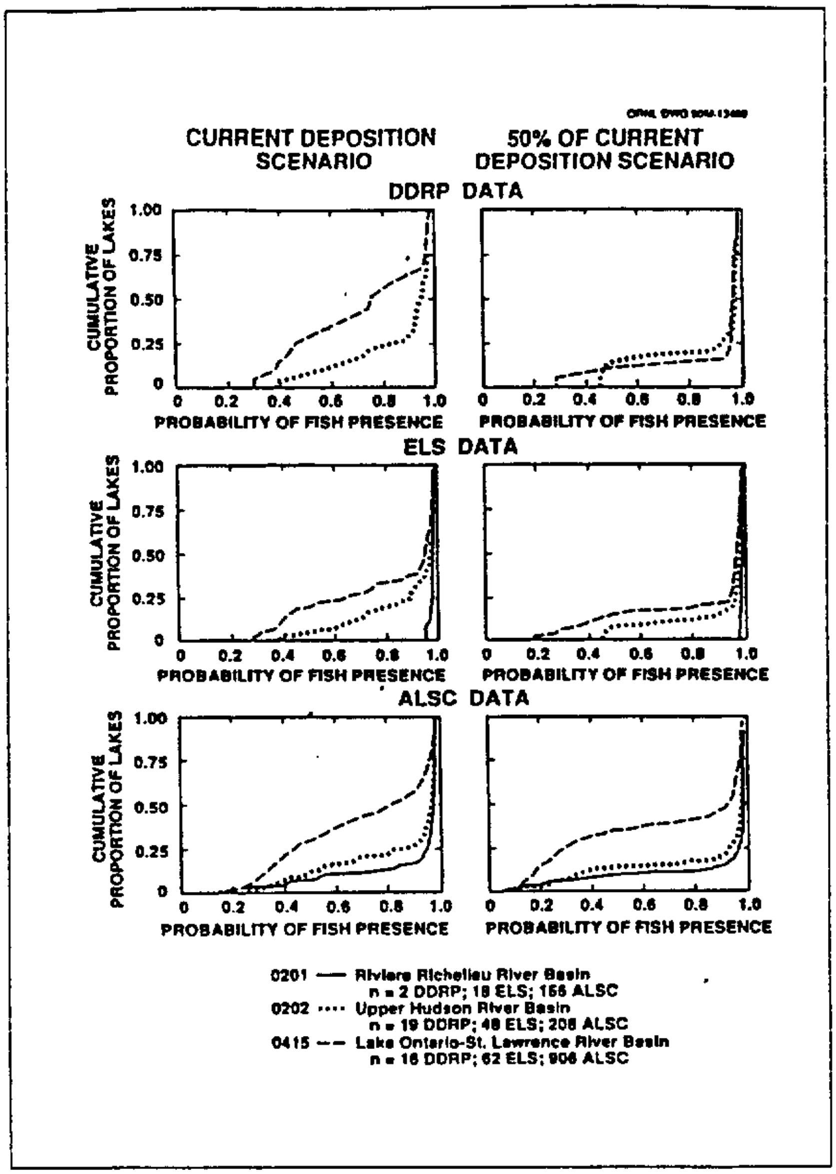


Figure 6. Henriksen model results for presence of brook trout with current deposition and reduced deposition for three databases. Lakes were assigned to river basin. Cumulative frequencies are not shown for subregions with less than ten lakes.

programmatic effects assessments at the regional scale. The definition of the assessment region and subregions is an important component of the assessment process. As shown by the Adirondack examples, different data sets may suggest somewhat different subregion schemes. Defining regions and subregions can improve the assessment by giving policy makers a geographic context and by capturing the spatial variability of endpoint responses. The use of ecologically functional subregions should improve the cost-benefit ratio for control and the accuracy of the sensitivity predictions by fine tuning effects models. Even a logical subregion scheme is not useful if it is so complex that sample size within subregions becomes too small for statistical confidence. Of course, the risk

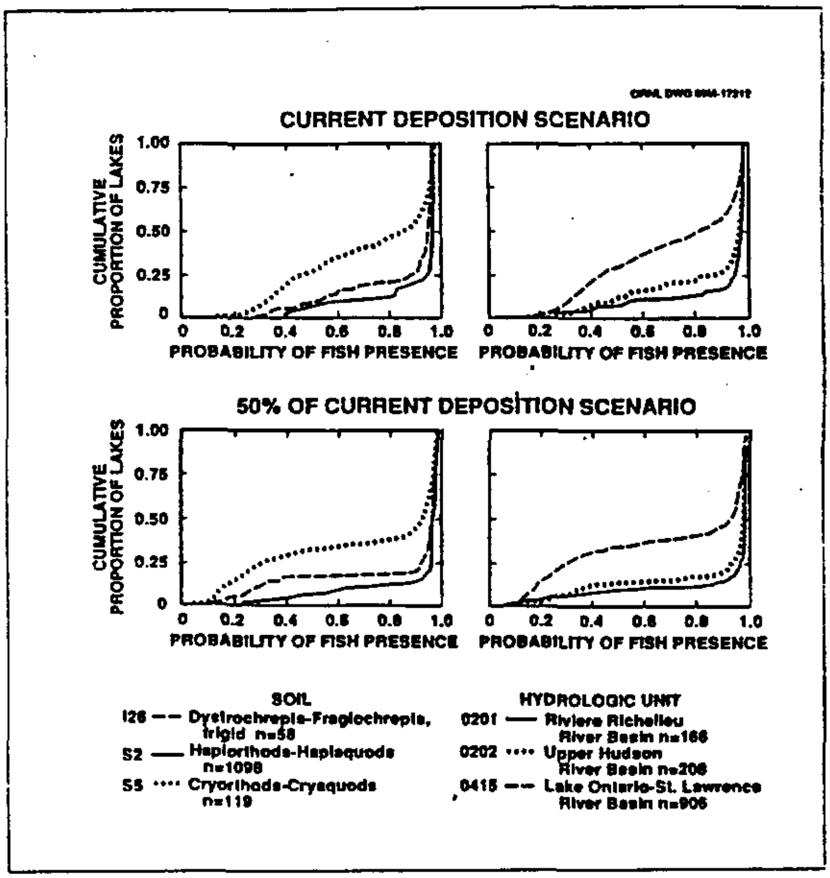


Figure 7. Henriksen model results for presence of brook trout with current deposition and reduced deposition. Lakes were assigned to soil and river basin subregions. The number of lakes used was 1280 from the Adirondack Lakes Survey Corporation (ALSC) database. Cumulative frequencies are not shown for subregions with less than ten lakes.

assessment approach stresses quantification of effects and uncertainty; such quantification should be our ultimate goal because it will provide policy makers and the public with an objective way to make decisions when cumulative effects are involved.

It is always a challenge to present in an understandable manner the analyses and results of complex assessments to policy makers and the general public. This task is only exacerbated for regional and cumulative assessments where large amounts of data, large geographic areas, and more quantitative methods are the norm. I believe that dose-response curves, cumulative frequency distributions, and maps are very important tools for illustrating cumulative effects analyses. Tools that will improve our ability to perform regional and cumulative assessments include geographic information systems, improved application of remote sensing data, and landscape indices that capture landscape patterns relevant to ecological processes (O'Neill et al. 1988). The availability of integrated databases is one of the factors most hindering our ability to perform these assessments. A recent emphasis on ecological monitoring at the regional and national scales (Hunsaker and Carpenter 1990) and revisions to national monitoring programs (Hirsch, Alley, and Wilber 1988) are an encouraging sign that such databases may exist in the future. Both consistent and comprehensive long-term monitoring are needed at the correct scales for cumulative effects assessments.

As outlined in Figure 1, there is a logical spatial hierarchy that is sometimes neglected in the preparation of impact assessments. As Myslicki (this volume, Chap. 5) comments, "many times a programmatic EIS is the only place that impacts across diverse geographic areas have the opportunity to be considered." Effects assessments for programmatic and cumulative assessments addressing a large geographic area share some common goals and needs, and a regional ecological risk assessment approach is suitable for these assessments.

Control of impacts often occurs at the local scale. Thus, if cumulative impacts are best addressed at the regional scale, we must prevent the EIS process from becoming an analysis without context. This can be achieved by regional and programmatic planning as performed by associations of city and county governments, river basin commissions, state planning activities, national monitoring and assessments, and follow-up audits of NEPA documents.

ACKNOWLEDGMENTS

The author wishes to thank her many coworkers for their contributions to the projects that provided the basis and impetus for this paper. A special thanks goes to Robin Graham, Glenn Suter, Larry Barnthouse, and Glenn Cada of the Environmental Sciences Division at Oak Ridge National Laboratory. I acknowledge the work of the many scientists who contributed to the U.S. Critical Loads project and the National Acid Precipitation and Assessment project. I especially thank J. W. Elwood for runs of Henriksen's model, J. P. Baker for use of her brook trout model, and P. Ringold and G. R. Holdren for project management. The use of chemistry data is also appreciated: Adirondack Lake Survey Corporation and U.S. Environmental Protection Agency (EPA) Direct/Delayed Response Project and Eastern Lake Survey. Research was sponsored jointly by the EPA under Interagency Agreement 1824-B014-A7 and the Environmental Sciences Division, Office of Health and Environmental Research, U.S. Department of Energy, under contract DE-AC05-84OR21400 with Martin Marietta Energy Systems, Inc. Although this research was partially funded by EPA, it has not been subjected to EPA review and, therefore, does not necessarily reflect the views of EPA and no official endorsement should be inferred.

REFERENCES

Adirondack Lakes Survey Corporation (ALSC). 1989. Adirondack lakes study (1984–1987): An Evaluation of Fish Communities and Water Chemistry. ALSC, Albany, NY.

Baker, J. P., C. S. Creager, S. W. Christensen, and L. Godbout. 1988. Identification of

Critical Values for Effects of Acidification on Fish Populations. Report No. 1296-122-7/31/88-01F. U.S. Environmental Protection Agency, Washington, D.C.

Baker, J. P., D. P. Bernard, S. W. Christensen, M. J. Sale, J. Freda, K. Heltcher, D. Marmorek, L. Rowe, P. Scanlon, G. Suter, W. Warren-Hicks, and P. Welbourn. 1990. Biological Effects of Changes in Surface Water Acid-Base Chemistry. State-of-Science/Technology Report 13. National Acid Precipitation Assessment Program, Washington, D.C.

Cada, G. F., and C. T. Hunsaker. 1990. Cumulative impacts of hydropower development: Reaching a watershed in impact assessment. *Environ. Prof.* 12:2–8.

Canadian Environmental Assessment Research Council and the U.S. National Research Council (eds.). 1986. Cumulative Environmental Effects: A Binational Perspective. Canada Environmental Assessment Research Council, Hull, Quebec.

Council on Environmental Quality (CEQ). 1978. Regulations for Implementing the Procedural Provisions of the National Environmental Policy Act. 40 CFR 1500-08. CEQ, Washington, D.C.

Convention on Long-Range Transboundary Air Pollution, Task Force on Mapping Critical Levels/Loads (CLRTAP). 1989. Methodologies and Criteria for Mapping Critical Levels/Loads and Geographical Areas Where They Are Exceeded, Draft Manual. Workshop on Mapping, November 6–9, 1989. Bad Harzburg, Federal Republic of Germany.

de Blij, H. J. 1978. *Geography: Regions and Concepts. Introduction.* John Wiley & Sons, New York.

Federal Energy Regulatory Commission (FERC). 1985a. Procedures for assessing hydropower projects clusters in river basins. Request for comments. Docket No. EL85-19-000. *Fed. Regist.* 50(16):3385–3403.

Federal Energy Regulatory Commission (FERC). 1985b. Final Environmental Impact Analysis of Small-Scale Hydroelectric Development in Selected Watersheds in the Upper San Joaquin River Basin, California. FERC/EIA-0001. Office of Hydropower Licensing, Washington, D.C.

Federal Energy Regulatory Commission (FERC). 1986. Owens River Basin, Seven Hydroelectric Projects — California. Final Environmental Impact Statement. FERC/EIS-0041. Office of Hydropower Licensing, Washington, D.C.

Federal Energy Regulatory Commission (FERC). 1988a. Hydroelectric Development in the Upper Ohio River Basin. Final Environmental Impact Statement. FERC/EIS-0051. Office of Hydropower Licensing, Washington, D.C.

Federal Energy Regulatory Commission (FERC). 1988b. Public Utility Regulatory Policies Act Benefits at New Dams and Diversions. Final Staff Report Evaluating Environmental and Economic Effects. Docket No. EL87-9. Office of Hydropower Licensing, Washington, D.C.

Henriksen, A. 1984. Changes in base cation concentrations due to freshwater acidification. *Verh. Internat. Verein. Limnol.* 22:692–698.

Hirsch, R. M., W. M. Alley, and W. G. Wilber. 1988. Concepts for a National Water-Quality Assessment Program. U.S. Geological Survey Circular 1021, Washington, D.C.

Hunsaker, C. T., S. W. Christensen, J. J. Beauchamp, R. J. Olson, R. S. Turner, and J. L. Malanchuk. 1986. Empirical Relationship Between Watershed Attributes and Headwater Chemistry in the Adirondack Region. ORNL/TM-9838. Oak Ridge National Laboratory, Oak Ridge, TN.

Hunsaker, C. T., and D. E. Carpenter (eds.). 1990. Ecological Indicators, Environmental Monitoring and Assessment Program. EPA 600/3-90/060. U.S. Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC.

Hunsaker, C. T., R. L. Graham, G. W. Suter, R. V. O'Neill, B. L. Jackson, and L. W. Barnthouse. 1990. Regional ecological risk assessment. *Environ. Manage.* 14(3):325–332.

Linthurst, R. A., D. H. Landers, J. M. Eilers, D. F. Brakke, W. S. Overton, E. P. Meier, and R. E. Crowe. 1986. Characteristics of Lakes in the Eastern United States. Vol. 1: Population Descriptions and Physiochemical Relationships. EPA-600/4-86-007A. U.S. Environmental Protection Agency, Washington, D.C.

Myslicki, A. 1993. The use of programmatic EISs in support of cumulative impact assessment. In S. G. Hildebrand, and J.B. Cannon (eds.), *The Scientific Challenges of NEPA: Future Directions Based on 20 Years of Experience.* Chap. 5. Lewis Publishers, Inc., Chelsea, MI.

NAPAP (National Acid Precipitation and Assessment Program). 1990. 1989 Annual Report to the President and Congress and Findings Update. NAPAP, Washington, D.C.

O'Neill, R. V., J. R. Krummel, R. H. Gardner, G. Sugihara, B. Jackson, D. L. DeAngelis, B. Milne, M. G. Turner, B. Zygmutt, S. Christensen, V. H. Dale, and R. L. Graham. 1988. Indices of landscape pattern. *Landscape Ecol.* 1:153–162.

Sigal, L. L., and J. W. Webb. 1989. The programmatic environmental impact statement: Its purpose and use. *Environ. Prof.* 11:14–24.

Sullivan, T. J. 1990. Historical Changes in Surface Water Acid-Base Chemistry in Response to Acidic Deposition. State-of-Science Technology Report 11. National Acid Precipitation Assessment Program, Washington, D.C.

Thornton, K., D. Marmorek, and P. Ryan. 1990. Methods for Projecting Future Changes in Surface Water Acid-Base Chemistry. State-of-Science Technology Report 14. National Acid Precipitation Assessment Program, Washington, D.C.

Turner, M. G. 1989. Landscape ecology: The effect of pattern on process. *Annu. Rev. Ecol. Syst.* 20:171–197.

Turner, R. S., C. C. Brandt, D. D. Schmoyer, J. C. Goyert, K. D. Van Hoesen, L. J. Allison, G. R. Holdren, P. W. Shaffer, M. G. Johnson, D. A. Lammers, J. J. Lee, M. R. Church, M. L. Papp, and L. J. Blume. 1990a. Direct/Delayed Response Project Data Base Users' Guide. ORNL/TM-10369. Oak Ridge National Laboratory, Oak Ridge, TN.

Turner, R. S., R. B. Cook, H. V. Miegroet, D. W. Johnson, J. W. Elwood, O. P. Bricker, S. E. Lindberg, and G. M. Hornberger. 1990b. Watershed and Lake Processes Affecting Surface Water Acid-Base Chemistry. State-of-Science Technology Report 10. National Acid Precipitation Assessment Program, Washington, D.C.

Assessing Ecological Risk on a Regional Scale[1]

CAROLYN T. HUNSAKER
ROBIN L. GRAHAM
GLENN W. SUTER II
ROBERT V. O'NEILL
LAWRENCE W. BARNTHOUSE
ROBERT H. GARDNER
Environmental Sciences Division
Oak Ridge National Laboratory
P.O. Box 2008
Oak Ridge, Tennessee 37831-6038, USA

ABSTRACT / Society needs a quantitative and systematic way to estimate and compare the impacts of environmental problems that affect large geographic areas. This paper presents an approach for regional risk assessment that combines regional assessment methods and landscape ecology theory with an existing framework for ecological risk assessment. Risk assessment evaluates the effects of an environmental change on a valued natural resource and interprets the significance of those effects in light of the uncertainties identified in each component of the assessment process. Unique and important issues for regional risk assessment are emphasized; these include the definition of the disturbance scenario, the assessment boundary definition, and the spatial heterogeneity of the landscape.

The objective of risk-based ecological assessment is to provide: (1) a quantitative basis for balancing and comparing risks associated with environmental hazards, and (2) a systematic means of improving the estimation and understanding of those risks. In ecological risk assessment, uncertainties concerning potential environmental effects are explicitly recognized and, if possible, quantified. A better understanding of risks associated with an environmental hazard is achieved by comparing the magnitudes of uncertainties in different steps of the causal chain that links the initial event (e.g., release of a toxic chemical) and its ultimate consequence (e.g., alteration of an ecosystem). Ecological processes operate at a variety of scales in space and time. Many environmental hazards impact large geographic areas (e.g., acid deposition, nonpoint-source pollution, and increased global CO_2), yet traditional concepts and methods in ecology and risk assessment are relevant mainly to single sites or small geographic areas. Effective long-term management and protection of valuable natural resources require a better understanding of how the scale of the environmental hazard affects ecological processes and over what scales the effects should be monitored and examined.

KEY WORDS Regional risk, Landscape ecology, Impact analysis, Environmental assessment

Any risk assessment should be properly scaled for the environmental hazard being analyzed. Hazards and their associated risk assessments will exist along a continuum of spatial scales, but for ease of discussion, we will divide that continuum into two classes—local and regional. Our differentiation is best illustrated by example. Local hazards amenable to local risk assessments include: (1) the annual effects of a single industrial effluent on water quality in the mixing zone of the discharge, and (2) the effects of harvesting practices on the habitat of an endangered species in a tract of a national forest. Regional counterparts to these local hazards would be: (1) the impacts on water quality in a river basin that will result from proposed industrial and municipal discharges and projected land use in the next ten years, and (2) the effect of forest management practices on the survival of the spotted owl in the entire Pacific Northwest. In these latter two examples, both the cause and the consequence of the environmental hazard are regional. Regional hazards can also be caused by a local phenomenon that has a regional consequence (e.g., single-source pollutants that become widely dispersed, such as the radioactivity from Chernobyl). Alternatively, multiple local factors can combine to create a regional hazard to a population, species, or ecosystem type that is widely dispersed.

In this article, we propose a two-phase approach for doing regional ecological risk assessment and discuss the key components of that approach. Sources of uncertainty in regional ecological risk assessment are identified and evaluated.

[1]Although the research described in this article has been funded wholly or in part by the United States Environmental Protection Agency (EPA) through Interagency Agreement Number DW89932112-01-2 to the U.S. Department of Energy, it has not been subjected to EPA review and therefore does not necessarily reflect the views of EPA and no official endorsement should be inferred.

Table 1. Regional risk assessment terms

Term	Definition	Example
Hazard	Pollutant or activity and its disruptive influence on the ecosystem containing the endpoint	Forest cutting practices that eliminate critical habitat for an endangered species
Endpoint	Environmental entity of concern and the descriptor or quality of the entity	Extinction of an endangered species
Source terms	Qualitative and quantitative descriptions of the source of the hazard	Forest cutting practices and the laws and economic factors that influence them in the Piedmont
Reference environment	Geographic location and temporal period for the risk assessment	Piedmont of the United States in the next ten years
Exposure/habitat modification	Intensity of chemical and physical exposures of an endpoint to a hazard	Amount of habitat, for an endangered species, that is lost

Approach to Regional Ecological Risk Assessment

Our approach to regional ecological risk assessment is derived from the method described by Barnthouse and Suter (1986). The key steps include: (1) qualitative and quantitative description of the source terms of the hazard (e.g., locations and emission levels for pollutant sources), (2) identification and description of the reference environment within which effects are expected, (3) selection of endpoints, (4) estimation of spatiotemporal patterns of exposure by using appropriate environmental transport models or available data, and (5) quantification of the relationship between exposure in the modified environment (reference environment) and effects on biota. These five steps produce a risk assessment that describes the ultimate effects of the hazard on the endpoints in the reference environment and interprets the significance of those effects in light of the uncertainties identified in each step.

To express some of the concepts pertinent to regional risk assessment, we find it helpful to adopt the conventional terms developed for site-specific assessment of chemicals (Cohrssen and Covello 1989, Barnthouse and Suter 1984). Definitions of these terms are given in Table 1 along with a regional example. Although it is conventional to use only the term exposure, we believe it is clearer to use exposure/habitat modification to define both the chemical and the physical exposures that the entity (or the target organism) might experience in the modified environment.

Regional and local risk assessments have two distinct phases (Figure 1): (1) the definition phase, in which the endpoint, source terms, and reference environment are defined; and (2) the solution phase, in which exposure and effect are assessed and exposure levels are related to effects levels to determine risk (the probability of a certain event happening).

Local risk assessments and regional risk assessments differ significantly in both definition and solution phases. The definition phase of local assessments tends to address the selection of endpoints, the development of source terms, and the description of the reference environment as distinct activities; this phase is likely to be influenced or constrained by regulations or conventional practices. In regional risk assessment, however, the initial concept of the hazard usually is more nebulous, and the interactions between the components of the definition phase are often complex. We foresee that the definition phase of regional ecological assessments will be iterative because the source terms, endpoint, and reference environment are all interdependent. Any refinement in one of these components will affect the others. Furthermore, one must consider not only ecological processes but also pertinent social, economic, and institutional processes. Only then can endpoints properly be selected, source terms developed, and the reference environment (in this case the region) described. In the solution phase, regional assessments differ from local ones in two ways. First, the models used in the exposure and effects assessment must be regional; local models may have to be adapted to larger geographic regions (Dailey and Olson 1987) or very different models developed. Second, the exposure or effects assessment must account for uncertainty that may arise because of spatial heterogeneity, a feature that may not be significant in local assessments.

Developing source terms can be difficult for regional hazards because they often involve multiple sources that vary in both space and time. For example, an analysis of a regional ozone problem would need to consider industrial point source emissions of hydrocarbons, biogenic hydrocarbon emissions, as well as automobile emissions of nitrous oxides. In some instances an emission may become a source of impact only under a certain combination of factors. For example, sewage treatment effluents may induce an environmental problem such as a major fish kill only when the treatment plants are overloaded and river

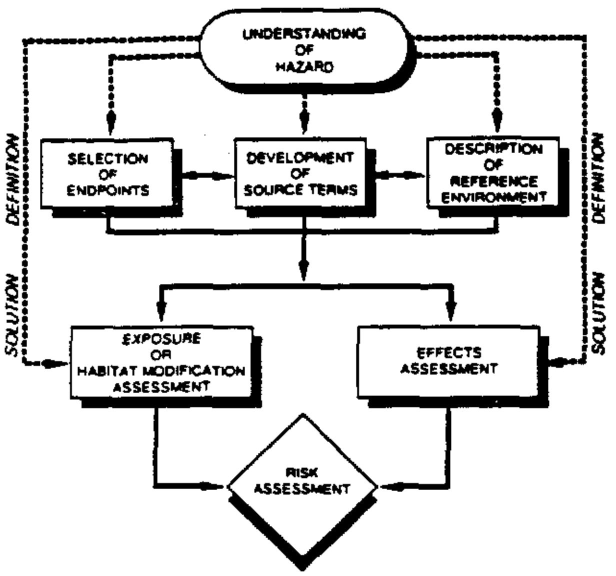


Figure 1. The two phases of regional risk assessment: the hazard definition and the problem solution.

flow is low. A long-term chronic problem can become an acute impact under certain conditions.

For a regional assessment to be effective, the spatial and temporal boundaries of the reference environment must be defined appropriately for the hazard and the endpoint. Furthermore, the region should be functionally defined: that is, its boundaries should be determined by physical or biological processes that affect the impact of the hazard such as the boundaries of watersheds, airsheds, and physiographic provinces. Unfortunately, however, assessment boundaries are often determined by nonecological factors, such as political boundaries, available data, or the influence of interest groups. Assessments that use functionally defined boundaries will have greater applicability to other regions with similar problems.

The geographic area in which an endpoint experiences the hazard might or might not be contained within the geographical area that produced the hazard (Bormann 1987). The latter possibility may occur when economic/political processes have an important affect on the hazard, as in the case of atmospheric deposition. For example, air masses cross political boundaries, thus risk assessment aimed at protecting the forests of Germany cannot be conducted without considering the emissions of neighboring countries. More research is needed on how to integrate ecological, social, and economic data in order to determine the boundaries of assessment regions. The appropriate spatial and temporal boundaries of the reference environment for a regional problem may not become apparent until an assessment is underway.

For any risk assessment, assessment endpoints must be potentially affected by the disturbance, be important to society, have an unambiguous and operational definition, and be accessible to prediction and measurement (Barnthouse and Suter 1984). In addition, endpoints for regional assessments must be representative of the regional reference environment. Regional assessments can use endpoints that occupy small areas if they are distributed across the reference environment (e.g., vernal pools in southern California coastal province).

Regional risk endpoints can be exposure-oriented or effects-oriented. Exposure-oriented endpoints include media or biota contaminated by pollutants. Effects-oriented endpoints include unacceptable changes in population or in system properties such as productivity or albedo. Endpoints can be defined by legislation (e.g., criteria or standards) or by ecological sensitivities to the hazard.

Our experience indicates that regional endpoints should be defined in terms of observations that can be made over large geographic areas and often long time periods. For terrestrial systems, endpoints might include the percent of cover of different vegetation types, the productivity or composition of the ecosystem, or the presence of an indicator species. The endpoints in aquatic assessments might be the frequency of lakes or *n*th-order streams in which an important species becomes extinct, the percent of areal reduction of *Spartina* in salt marshes, the species composition of an aquatic community, or the quality of water as indicated by a water-quality index. Long-term data are usually needed to identify significant trends in regional studies.

Integrated properties of regions or landscapes may also be appropriate endpoints in regional ecological risk assessments (Allen and others 1984). Examples of such properties include dominance (degree to which the landscape is dominated by a particular land type), contagion (degree to which the landscape is dissected into small patches or aggregated into large, continuous patches), fractal dimension (index of complexity of shapes on the landscape), and amount of edge (Krummel and others 1986, O'Neill and others 1988). Because such indices can be calculated from classifications developed from remotely sensed imagery, they might be especially useful in long-term monitoring of regional processes.

Sources of Uncertainty in Regional Assessments

Regional ecological risk assessment involves, for the most part, the same sources of uncertainty as local risk assessment. The relative importance of a given source of uncertainty depends on the hazard and the endpoint. Some components of uncertainty are relevant to both the description and the solution phases of a regional risk assessment, whereas others are important to only one phase. Uncertainties related to source terms and boundary definitions are relevant to the problem definition phase, whereas uncertainties related to model structure and model parameters are relevant to the problem solution phase. Uncertainties related to temporal scale and spatial heterogeneity are important to both phases of the assessment. All the uncertainties are combined in the final risk assessment.

The quantification of uncertainty for local ecological assessments has only recently received serious attention (Suter and others 1987), and quantification of uncertainty for regional assessments is just developing (Kamari and others 1986, Cosby and others 1987). Uncertainties may remain quite large in regional assessments, and there may be no practical way to reduce that uncertainty regardless of cost. Risk assessments centering on disturbances that are highly dependent on economic, social, and/or political factors are likely to fall into this category. If regional risk assessments are to be economical and useful, recognition of the importance of these factors early in the problem definition is critical.

Sometimes it is difficult to define source terms for a hazard, especially in predictions for the distant future. When some component of the hazard is highly uncertain, scenarios are a tool for bracketing the potential range of the hazard or some component of it. Typically, several possible sets of scenarios—that is source terms, reference environments, and endpoints—can be considered. Scenarios will likely be used in regional risk assessments when considerable uncertainty exists about the hazard (e.g., climate change or future mix of energy technologies). The results of such risk assessments are conditional on the events in the scenario. Thus it is important to try to select scenarios that take into account probable events. For regional studies, the absolute uncertainty predicted for a given scenario might not be very useful, but the comparisons between the relative uncertainty from the analysis of each scenario will be useful to the decision maker.

The least amount of uncertainty occurs when the "true" geographic boundary for the hazard is known (Allen and others 1984), as with a pollutant whose transport and fate are well defined. Boundary definitions become a problem when the functional region crosses political boundaries. Once a boundary is set and analysis proceeds, the ability to assess the uncertainty introduced by the choice of the boundary is lost. Boundary problems could especially add to the uncertainty of an assessment if there is an omission of an important source, a component of an endpoint, or a process that influences the relationship between a source and an endpoint. For some problems, the error associated with the definition of the spatial and temporal boundaries for a region should be evaluated by estimating the risks under several different boundary definitions.

Uncertainty will increase if the risk assessment does not encompass disturbance dynamics at the appropriate temporal scale. If exposure has considerable temporal variation within a year, mean annual values of exposure or monthly averages may not reflect the impact on the endpoint. For example, episodic events of low pH associated with snow melt are of very short duration but can nevertheless determine trout survival. In this case, the extremes for pH and aluminum, not the means, are of critical importance, and the use of monthly averages would result in a highly inaccurate or even meaningless estimate of effects on fish. Instead, hourly measurements for aquatic systems are needed. The appropriate temporal scale may vary with different aspects of the same disturbance. In the preceding trout example, knowing sulfur deposition on a daily, rather than on a monthly, basis would probably not reduce the uncertainty in the risk assessment since concentrations in snow are dependent on long-term not short-term deposition.

The availability of data bases and models is a critical factor in the quality of an assessment. Although uncertainties in models and data arise in local risk assessments, they may become more critical in regional ones. The ability of a model to represent environmental processes at the spatial and temporal scales of interest is a fundamental issue. Few regional-scale biological models exist. In most instances, either local models will have to be adapted to larger regions (Solomon 1986, Dale and Gardner 1987, Thornton and others 1987, Cosby and others 1987) or entirely new models will have to be developed (Emanuel and others 1985a,b, Hunsaker and others 1986).

Uncertainties associated with parameter values can be partially resolved through standard uncertainty test procedures (Gardner 1984, Hoffman and Gardner 1983). Parameter uncertainty includes both natural variability and uncertainty resulting from lack of knowledge. In regional ecological risk assessments, uncertainties that arise from the inherent variability and heterogeneity of natural populations and ecosystems are especially important. Population and ecosystem data contain inherent variability that no amount of monitoring will reduce.

The quality, acquisition, and use of data can dramatically affect the cost of an assessment and can contribute to uncertainty. Point data for large geographical regions are often uneven in quality and distribution. For example, one state may gather water-quality data with one technique, and another state may use another technique or a different sampling frequency. [Remote-sensing technology, however, offers a synoptic view of a region and holds promise for providing data that is truly regional (e.g., Greegor 1986, NASA 1987, Tucker and others 1986).] Data manipulation and extrapolation can also contribute to uncertainty because error may arise during the process of sampling at a particular spatial and temporal frequency (grain and extent), extrapolating from point data to contour data, and aggregating and disaggregating data. The classification of geographic areas according to the relative homogeneity of one or more environmental attributes can be extremely useful in reducing uncertainty if the classification scale is appropriate to the hazard. Ecoregions are examples of geographic classifications (Bailey 1983, 1987, Omernik 1987, Rohm and others 1987). However, the contribution to assessment uncertainty from such classification needs further investigation because classification or aggregation of data could mask spatial heterogeneity that is significant to a realistic evaluation of the hazard (McDaniel and others 1987).

Some regional-scale models may well be impossible to validate in the traditional sense. In such cases, quantification of the error associated with the model's structure will be difficult. Examples of such models include those that predict a modified environment as a result of events that have never occurred, such as a major transportation accident involving nerve gas, an extreme climate change, or any situation in the distant future. In such cases it is useful to compare models that purport to predict the same condition or effect (Thornton and others 1987, Turner 1987b); if the models give similar results, then confidence in their prediction is improved. But sometimes only one model is available. Another verification technique is to use, as the evaluation data set, the portions of a known data distribution that are representative of the conditions that the model needs to predict. For example, a model designed to predict the effects of climate change might be verified using data on the effects of observed and/or historic climatic extremes—the wettest/driest and warmest/coolest portions of meteorological records. Klemes (1985) discusses model transferability and presents a hierarchical scheme for systematic testing or verification of models.

Spatial heterogeneity can be a major source of uncertainty in regional ecological risk assessment. Most ecological modeling has not included spatial relationships, and there are no accepted measures of landscape pattern or heterogeneity that can be linked to processes occurring at a landscape scale (Bormann 1987). Although spatial heterogeneity is not necessarily a factor in all regional risk assessments, it can contribute to uncertainty in some situations. Thus, one must first ascertain if spatial heterogeneity is likely to influence the projected outcome of the hazard. If it is, then spatial heterogeneity must be accounted for in the assessment.

Some hazards can be viewed as an aggregation of local hazards—the situation where cumulative effects are linear or additive. In such cases, the regional risk is simply the sum of the local risks. For instance, estimation of the number of acidic lakes in the United States has been treated as an aggregate problem. Therefore, the United States was stratified into: (1) regions; (2) homogeneous subregions with respect to physiography, vegetation, climate, and soils; and (3) alkalinity classes. Then, a statistical sample of all the lakes in a stratum was used to predict a regional and, eventually, a national value for the number of acidic lakes (USEPA 1986). If, however, one or more properties associated with the hazard become apparent only on a regional scale, then treating the hazard as an aggregation of local effects is inappropriate. The impact of sewage on water quality, for example, is a function of not only the amount of sewage but also the quality of the water upstream of the discharge. Thus, when the connectedness of the hydrologic system is an important feature of the hazard, simply summing local risks is not an adequate assessment.

Aspects of spatial heterogeneity that might influence ecological risk include patch and population sizes, ratio of patch edge to interior, distance between patches, and appropriate spatial resolution. Because a better understanding of these aspects is essential for regional ecological assessment, they are discussed separately at greater length in the following paragraphs.

The size distribution of habitat patches or populations in a region may affect the impact of a disturbance (Turner 1987a, Sharpe and others 1987, Hayes and others 1987). For example, forest bird species richness in a temperate agricultural landscape is a linear function of the log of the size of remnant forest patches (Freemark and Merriam 1986). Thus a hazard that reduced forest patch size would affect species richness differently for different size patches. All species require habitat of some minimal area; certain populations are likely to disappear if that area becomes too small (Noss 1983, van Dorp and Opdam 1987). Furthermore, some ecosystem functions (e.g., wetland ability to remove pollutants) may disappear when the system is reduced beyond a certain point. Ignoring the size distribution of patches or populations may increase the uncertainty associated with the risk assessment when this type of spatial heterogeneity is important.

The ratio of edge to interior of landscape elements, such as lakes and forests, may be important in assessing the ecological risk of some hazards. For example, the ratio of forest edge to interior has a profound effect on the magnitude of blowdown experienced in the Pacific Northwest (Franklin and Forman 1987). Cutting patterns that increase that ratio will increase blowdown even though the total area of cut forest may remain the same.

The distance between similar units of land or phenomena may also affect the outcome of a regional ecological risk assessment. For instance, distance between similar habitats may affect the ability of a species to migrate, which, in turn, may affect its ability to maintain a stable regional population under a given level of disturbance. Corridors that facilitate movement or transport affect the maximum connecting distance between areas for some processes or activities.

For each hazard there is a particular spatial scale at which uncertainty is minimized or the hazard is most clearly seen (Allen and others 1984). Landscapes are analogous to pointillistic paintings. If the viewer is too close (at too fine a resolution), the objects of interest cannot be seen. If the viewer is too far away (at too coarse a resolution), again, the objects of interest cannot be seen. It will be important in regional risk assessment both to identify the optimal spatial scale for viewing and collecting data and also to understand how the scale at which the landscape is viewed affects uncertainty.

Conclusion

Although regional studies have been performed for many years (McHarg 1969, Levenson and Stearns 1980, USDOE 1981, Klopatek and others 1981, Westman 1985), the ecosystem properties that are important for regional scales are still poorly understood (Meentemeyer and Box 1987). The degree to which these properties are significant in regional risk assessment is even less understood. To define the uncertainty associated with ecological risk assessments, we need also to consider the possible implications of scale to the risk assessment.

Regional risk assessment has some attributes in common with local risk assessment but has others that are unique. The general theoretical framework for doing the two types of environmental risk assessment is the same. Both have two phases: first, the hazard definition, in which the endpoint, source terms, and reference environments are defined and described; and second, the problem solution, in which the exposure and effect on the endpoint are assessed by using models and the risk and its associated uncertainty are determined. Regional risk assessment differs in: (1) the extent of interaction between the source terms, endpoints, and reference environment; and (2) the degree to which boundary definition and spatial heterogeneity are significant in determining uncertainty. Although local risk assessments involve the development of data bases and the use of models, these steps may be more significant in regional risk assessments. Few regional-level data bases of biological variables exist; furthermore, unique problems arise in aggregating or integrating dissimilar local data into regional data bases. Regional models of ecological processes are much less common and can be difficult to validate.

Although most of the fundamentals are in place for doing regional risk assessment, research is still needed on both theoretical and applied issues. Little is known about the influence that data aggregation has on uncertainty in model parameters. Questions about this influence invariably arise in regional studies with large data bases. Ecological hierarchy theory, ecoregion definitions, and multivariate and spatial statistical techniques will be useful in assessing the significance of data aggregation. Research on the appropriate models for regional studies needs to continue. We need to know under what circumstances it is appropriate to adapt a local model to a region and how to do so. Our tools for describing landscape pattern are still experimental. The development of landscape pattern indices that capture important ecological processes at the landscape scale could significantly simplify regional monitoring. However, this development will require a more complete understanding of the interaction between landscape pattern and ecological processes. Some of the more recent technological tools—such as geographic information systems and satellite sensors that capture biologically significant spectral patterns (Tucker and Sellers 1986)—will be useful for addressing the theoretical and applied research issues that the regional scale poses. The simple lack of adequate spatial and temporal data for large geographic areas severely limits regional risk assessments. Many tools and ideas exist, but they need to be tested and refined before regional ecological risk assessment can become an effective tool for managing and protecting natural resources.

Acknowledgments

We thank J. L. Malanchuk for his foresight in funding this work. We also thank V. H. Dale, K. A. Rose, S. M. Bartell, and M. G. Turner for their helpful manuscript reviews. Research was sponsored by the Office of Ecological Processes and Effects Research, US Environmental Protection Agency, Washington, DC, under EPA Interagency Agreement DW89932112-01-2 with the US Department of Energy under contract DE-AC05-84OR21400 with Martin Marietta Energy Systems, Inc. Publication No. 3462, Environmental Sciences Division, ORNL.

Literature Cited

Allen, T. F. H., R. V. O'Neill, and T. W. Hoekstra. 1984. Interlevel relations in ecological research and management: some working principles from hierarchy theory. General Technical Report RM-110. Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Bailey, R. G. 1983. Delineation of ecosystem regions. *Environmental Management* 7:365–373.

Bailey, R. G. 1987. Suggested hierarchy criteria for multiscale ecosystem mapping. *Urban Land Planning* 14:313–319.

Barnthouse, L. W., and G. W. Suter II. 1984. Risk assessment ecology. *Mechanical Engineering* 106:36–39.

Barnthouse, L. W., and G. W. Suter II (eds.). 1986. User's manual for ecological risk assessment. ORNL-6251. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Bormann, F. H. 1987. Landscape ecology and air pollution. Pages 37–57 *in* M. G. Turner (ed.), Landscape heterogeneity and disturbance. Springer-Verlag, New York.

Cohrssen, J. J., and V. T. Covello. 1989. Risk analysis: a guide to principles and methods for analyzing health and environmental risks. Council on Environmental Quality, Executive Office of the President, Washington, DC.

Cosby, B. J., G. M. Hornberger, P. F. Ryan, and D. M. Wolock. 1987. Evaluating uncertainty in a regional application of an acidification model. *Eos* 68:1289.

Dailey, N. S., and R. J. Olson. 1987. Proceedings of the workshop on regionalization of aquatic impacts using the Adirondacks as a case study. ORNL/TM-10044. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Dale, V. H., and R. H. Gardner. 1987. Assessing regional impacts of growth declines using a forest-succession model. *Environmental Management* 24:83–93.

Emanuel, W. R., H. H. Shugart, and M. P. Stevenson. 1985a. Climatic change and the broad-scale distribution of terrestrial ecosystem complexes. *Climatic Change* 7:26–43.

Emanuel, W. R., H. H. Shugart, and M. P. Stevenson. 1985b. Response to comment: climatic change and the broad-scale distribution of terrestrial ecosystem complexes. *Climatic Change* 7:457–460.

Franklin, J. F., and R. T. T. Forman. 1987. Creating landscape patterns by forest cutting: Ecological consequences and principles. *Landscape Ecology* 1:5–18.

Freemark, K. G., and H. G. Merriam. 1986. Importance of area and habitat heterogeneity to bird assemblages in temperate forest fragments. *Biological Conservation* 37:115–141.

Gardner, R. H. 1984. An unified approach to sensitivity and uncertainty analysis. Pages 155–157 *in* M. H. Hamza (ed.), Applied simulation and modelling, Proceedings, IASTED International Symposium, 4–6 June 1984, San Francisco, California. ACTA Press, Anaheim, California.

Greegor, D. H. 1986. Ecology from space. *Bioscience* 36(7):429–438.

Hayes, T. D., D. H. Riskind, and W. L. Pace. 1987. Patch-within-patch restoration of man-modified landscapes within Texas state parks. Pages 173–198 *in* M. G. Turner (ed.), Landscape heterogeneity and disturbance. Springer-Verlag, New York.

Hoffman, F. O., and R. H. Gardner. 1983. Evaluation of uncertainties in environmental radiological assessment models. Pages 11-1–11-55 *in* J. E. Meyer and H. R. Meyer (eds.), Radiological assessment: a textbook on environmental dose assessment. NUREG/CR-3332, ORNL-5968. US Nuclear Regulatory Commission, Washington, DC.

Hunsaker, C. T., S. W. Christensen, J. J. Beauchamp, R. J. Olson, R. S. Turner, and J. L. Malanchuk. 1986. Empirical relationships between watershed attributes and headwater lake chemistry in the Adirondack region. ORNL/TM-9838. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Kamari, J., M. Posch, R. H. Gardner, and J. Hettelingh. 1986. A model for analyzing lake water acidification on a large regional scale, part 2: regional application. International Institute for Applied Systems Analysis, Laxenburg, Austria.

Klemes, V. 1985. Sensitivity of water resource systems to climate variations. WCP-98. World Meteorological Organization, Geneva, Switzerland.

Klopatek, J. M., J. T. Kitchings, R. J. Olson, K. D. Dumar, and L. K. Mann. 1981. A hierarchical system for evaluating regional ecological resources. *Biological Conservation* 20:271–290.

Krummel, J. R., R. H. Gardner, G. Sugihara, and R. V. O'Neill. 1986. Landscape patterns in a disturbed environment. *Oikos* 48:321–324.

Levenson, J. B., and F. W. Stearns. 1980. Application of diversity to regional ecological assessment: A review with recommendations. ANL/AA-21. Argonne National Laboratory, Argonne, Illinois.

McDaniel, T. W., C. T. Hunsaker, and J. J. Beauchamp. 1987. Determining regional water quality patterns and their ecological relationships. *Environmental Management* 11:507–518.

McHarg, I. 1969. Design with nature. Natural History Press, New York.

Meentemeyer, V., and E. O. Box. 1987. Scale effects in landscape studies. Pages 15–34 *in* M. G. Turner (ed.), Landscape heterogeneity and disturbance. Springer-Verlag, New York.

NASA. 1987. Linking remote-sensing technology and global needs: a strategic vision. NASA, Washington, DC.

Noss, R. F. 1983. A regional landscape approach to maintain diversity. *BioScience* 33:700–706.

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

O'Neill, R. V., J. R. Krummel, R. H. Gardner, G. Sugihara, B. Jackson, D. L. DeAngelis, B. Milne, M. G. Turner, B. Zygmunt, S. W. Christensen, V. H. Dale, and R. L. Graham. 1988. Indices of landscape pattern. *Landscape Ecology* 1:153–162.

Rohm, C. M., J. W. Giese, and C. C. Bennett. 1987. Evaluation of an aquatic ecoregion classification of streams in Arkansas. *Journal of Freshwater Ecology* 4:127–140.

Sharpe, D. M., G. R. Guntenspergen, C. P. Dunn, L. A. Leitner, and F. Sterns. 1987. Vegetation dynamics in a southern Wisconsin agricultural landscape. Pages 137–155 *in* M. G. Turner (ed.), Landscape heterogeneity and disturbance. Springer-Verlag, New York.

Solomon, A. M. 1986. Transient response of forests to CO_2-induced climate change: simulation modeling experiments in eastern North America. *Oecologia* 68:567–579.

Suter, G. W., L. W. Barnthouse, and R. V. O'Neill. 1987. Treatment of risk in environmental impact assessment. *Environmental Management* 11:295–303.

Thornton, K. W., M. R. Church, B. J. Cosby, S. Gherini, and J. S. Schnoor. 1987. Identifying factors controlling long-term acidification through the use of multiple models. *Eos* 68:1289.

Tucker, C. J., and P. J. Sellers. 1986. Satellite remote sensing of primary production. *International Journal of Remote Sensing* 4:1395–1416.

Tucker, C. J., J. R. G. Townshend, and T. Goff. 1986. Continental land cover classification using NOAA-7 AVHRR data. *Science* 227:369–375.

Turner, M. G. (ed.). 1987a. Landscape heterogeneity and disturbance. Springer-Verlag, New York.

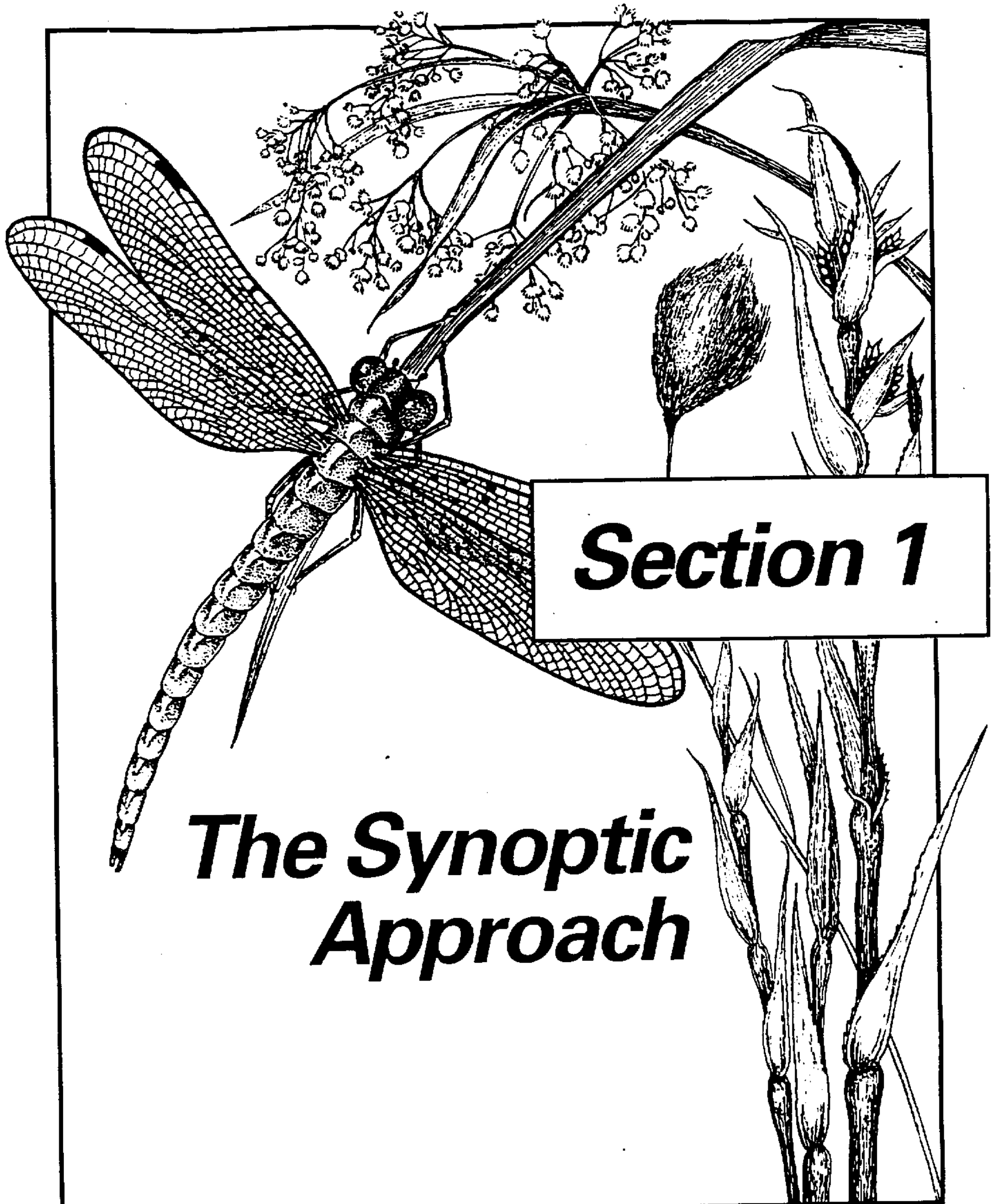
Turner, M. G. 1987b. Simulation of landscape changes in Georgia: a comparison of 3 transition models. *Landscape Ecology* 1:29–36.

USDOE (US Department of Energy). 1981. Regional issue identification and assessment. Washington, DC.

USEPA (US Environmental Protection Agency). 1986. Characteristics of lakes in the eastern United States. EPA/600/4-86/007a. Office of Research and Development, Washington, DC.

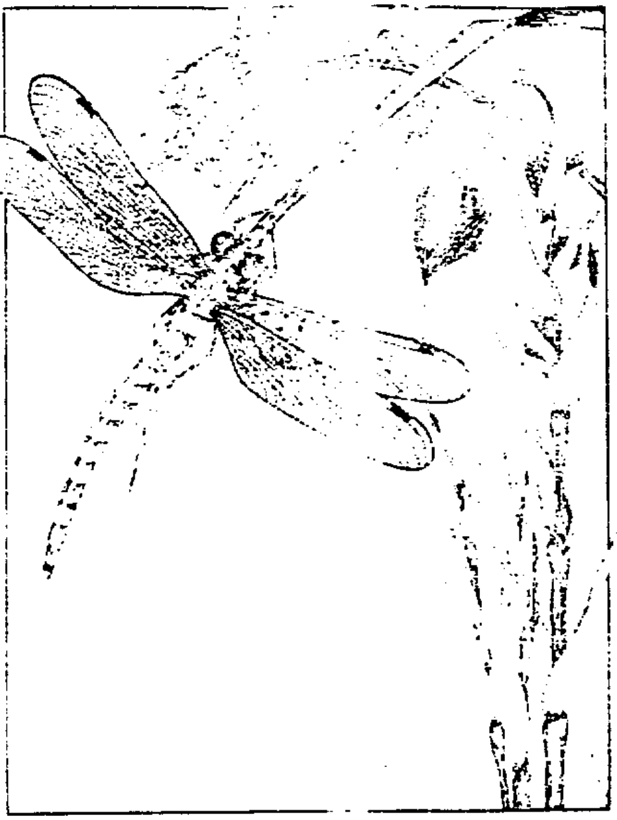
van Dorp, D., and P. F. M. Opdam. 1987. Effects of patch size, isolation and regional abundance on forest bird communities. *Landscape Ecology* 1:59–73.

Westman, W. E. 1985. Ecology, impact assessment, and environmental planning. Wiley-Interscience, New York.



Section 1

The Synoptic Approach



Chapter 1 Introduction

This report provides resource managers and technical staff with an approach for evaluating the cumulative environmental effects of individual human impacts on the environment, particularly with respect to wetlands. This document is intended to give the reader a general understanding of cumulative impacts and to describe how a synoptic assessment is produced. Although specifically designed for use in wetland permit evaluation under the Clean Water Act (CWA), this method can be applied to cumulative impact assessment in general[1]. A second objective of this report is to encourage resource managers responsible for wetland protection to consider and view wetlands within a landscape context.

The synoptic approach, so named because it provides a broad overview of the environment, was developed specifically for cases in which time, resources, and information are limited. The method is not intended to provide a precise, quantitative assessment of cumulative impacts *within* an area, nor can it be used to assess the cumulative effects of specific impacts. Rather, it provides a relative rating of cumulative impacts *between* areas. The approach is intended to be easily applied so it can augment the best professional judgment used daily by wetland managers and regulators.

This report is divided into two sections. Section 1 describes the method and illustrates its use. It defines cumulative impacts, reviews the regulatory basis for cumulative impact assessment, and introduces the Wetland Research Program's (WRP's) synoptic approach (Chapter 1). It also provides the ecological basis for the synoptic indices (Chapter 2), describes in detail how to conduct a synoptic assessment (Chapter 3), illustrates the method's use and several possible applications through four case studies (Chapter 4), and contains a summary that discusses future directions (Chapter 5). Section 2 contains detailed background material for readers interested in additional information. It includes a discussion of environmental stress (Chapter 6) and a review of wetland functions and the effects of impacts on these functions (Chapter 7).

Cumulative Impacts

Traditionally, impact assessment has evaluated the likely effects of a single action on the environment. There has been concern, however, that numerous activities considered insignificant by themselves could, when taken together, cause significant degradation and damage to

[1] Because of its general nature, the synoptic approach is not limited to legally defined (i.e., "jurisdictional") wetlands. We therefore define wetlands in the broadest sense, as those ecosystems that are characterized by: the presence of water; unique soils, compared to adjacent uplands; the presence of vegetation adapted to wet conditions; and an absence of flood-intolerant vegetation (Mitsch and Gosselink 1986).

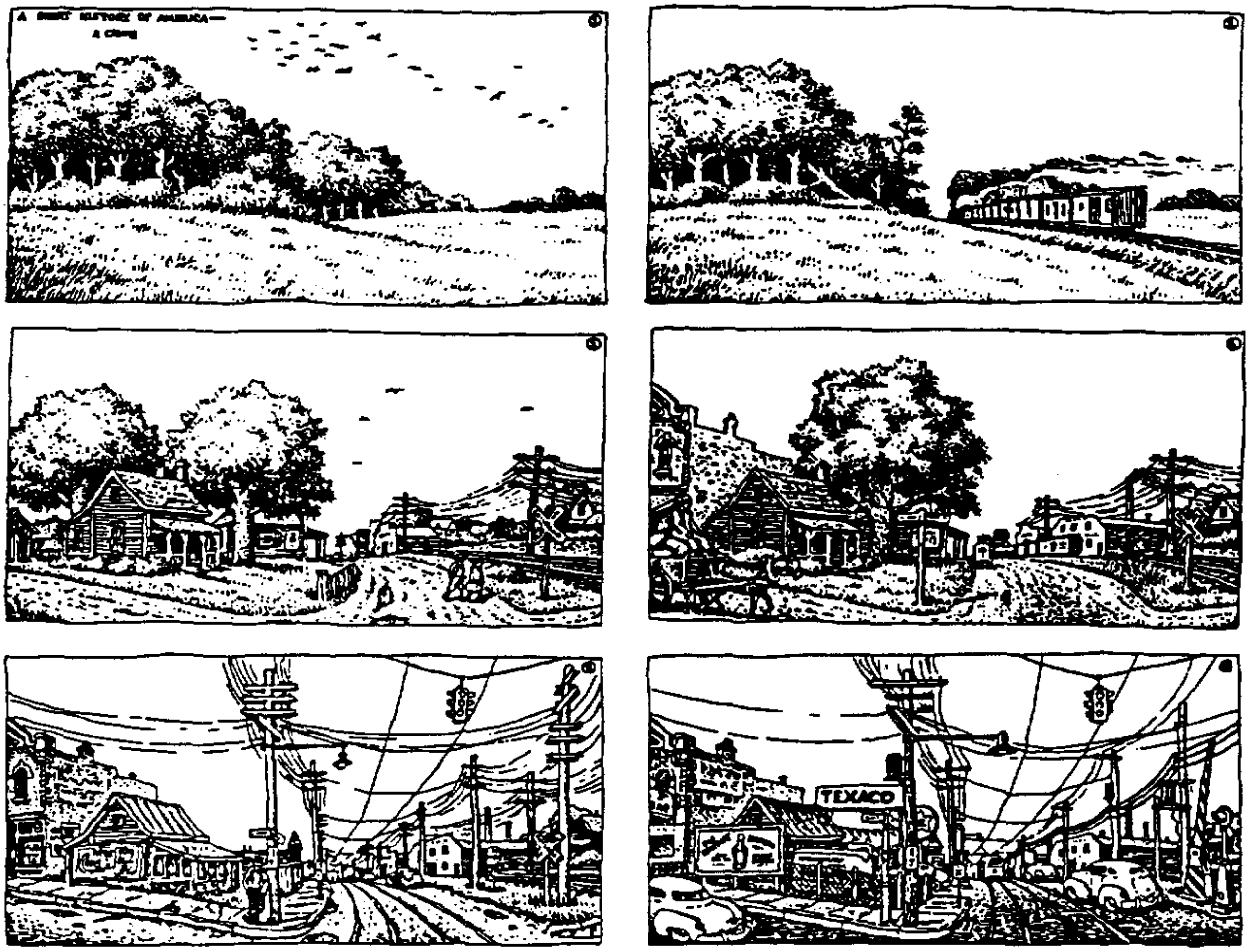
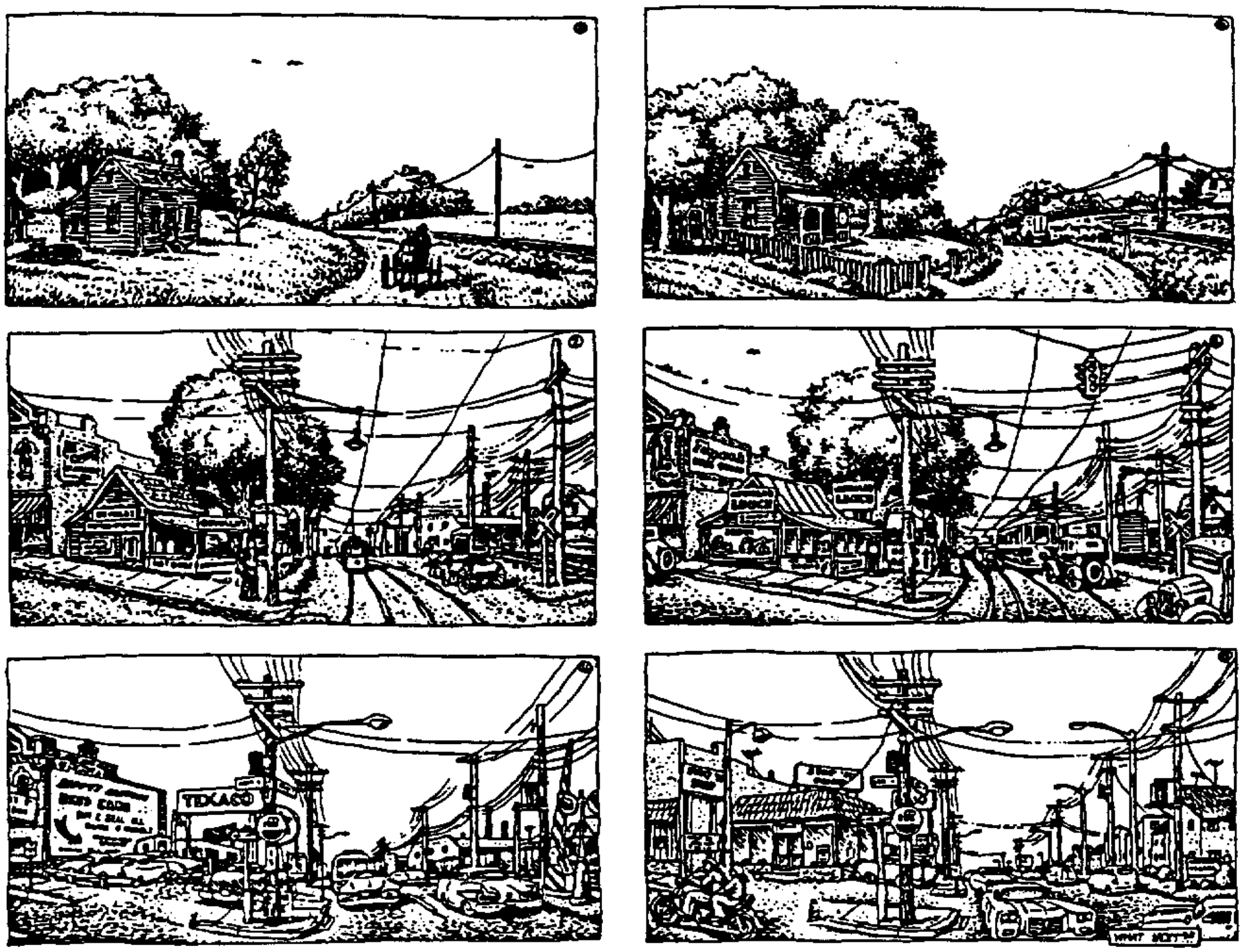


Figure 1.1. "A Short History of America," by the cartoonist R. Crumb, graphically illustrates cumulative impacts over time. Although none of the individual impacts would have been expected to significantly damage the environment, the cumulative result is a major loss of environmental functions (from CoEvolution Quarterly No. 23, Fall 1979, © R. Crumb 1992).

the environment (Kahn 1966; Odum 1982). An analogy provided by Ehrlich and Ehrlich (1981) illustrates this concept. If a single rivet pops out of a jet's wing, no serious threat exists, because no one rivet contributes significantly to the plane's airworthiness. But if enough rivets are lost, the integrity of the plane's structure gradually weakens until a failure occurs. In this analogy, the cumulative effect of the individually minor impacts would be catastrophic. In the same manner, a conventional impact analysis might conclude that a single discharge into a wetland would not amount to significant impact and would therefore be acceptable. However, an assessment that ignores the combined effect of these cumulative impacts could seriously underestimate the extent of environmental damage (Figure 1.1), thereby frustrating policy and management goals (Irwin and Rodes 1992).

A major difference between traditional impact assessment and cumulative impact assessment is that the former is performed with respect to the proposed *disturbance*. Cumulative impact assessment is performed with respect to valued environmental *functions* (Beanlands and Duinker 1983; Preston and Bedford 1988). Cumulative impact assessment must therefore take a holistic view of the environment. An excellent overview of cumulative impacts and wetlands is given in a special volume edited by Bedford and Preston (1988a) that includes a review of regulatory issues and the status of scientific understanding of cumulative impacts with respect to hydrology, water quality, and wildlife. This volume is highly recommended for readers interested in a more in-depth treatment of the subject.



Regulatory Mandate

Regulations prepared by the Council on Environmental Quality under the National Environmental Policy Act require environmental impact statements to "anticipate a cumulatively significant impact on the environment from Federal action" [2] (38 CFR Sect. 1500.6). A cumulative impact is 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 other actions. Cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time." (40 CFR Sect. 1508.7)

Under CWA Section 404, permits must be obtained to discharge dredged or fill material into waters of the United States, which include most wetlands. The CWA Section 404(b)(1) guidelines contain the criteria that are used in evaluating a permit for a proposed discharge. These regulations, promulgated by the Environmental Protection Agency (EPA) in conjunction with the Army Corps of Engineers, call for consideration of cumulative impacts (40 CFR 230.11):

> "[1] Cumulative impacts are the changes in an aquatic ecosystem that are attributable to the collective effect of a number of individual discharges of dredged or fill material. Although the impact of a particular discharge may constitute a minor change in itself, the

[2] "Federal action" has been interpreted to include any action regulated by the federal government.

cumulative effect of numerous such piecemeal changes can result in a major impairment of the water resources and interfere with the productivity and water quality of existing aquatic ecosystems.

[2] Cumulative effects attributable to the discharge of dredged or fill material in waters of the United States should be predicted to the extent reasonable and practical. The permitting authority shall collect information and solicit information from other sources about the cumulative impacts on the aquatic ecosystem. This information shall be documented and considered during the decision-making process concerning the evaluation of individual permit applications, the issuance of a General Permit, and monitoring and enforcement of existing permits."

Regulatory Context

If a proposed discharge involves a major or controversial action, permit evaluation requires extensive information and may include collection of field data and even an Environmental Impact Statement (Hirsch 1988). However, most of the permit requests received each year are for minor, routine actions. Because of the large number of requests and the limited amount of time and staff, a simpler environmental assessment must be conducted, based upon existing information.

There are a number of methods for evaluating cumulative impacts (Appendix A); however, none of these are practical within the regulatory constraints of Section 404. Although the concept of cumulative impacts is intuitive enough to have influenced the guidelines for permit evaluation, the lack of an easily applied method makes it difficult to consider cumulative impacts as part of routine permit decisions (Preston and Bedford 1988). Therefore, regulators must often rely on best professional judgment in order to comply with the 404(b)(1) guidelines. A major goal of EPA's Wetlands Research Program has been to provide permit reviewers with an easily applied technical approach for assessing cumulative impacts.

Our current understanding of the environment and our lack of data make it impossible to provide a precise, quantitative evaluation of the effects that cumulative wetland losses will have in a specific region or to predict how additional wetland losses will add to those effects. However, our understanding of ecological processes in general, and wetlands in particular, should be sufficient for us to make qualitative comparisons of these effects between different areas. For example, we may not be able to say that the cumulative loss of 100 hectares of wetland within a particular area caused a 10% reduction in water quality; however, we should be able to say that a 100 hectare loss of wetland in area "A" will more likely cause a reduction in water quality than a similar loss in area "B". The synoptic approach is a response to Hirsch's (1988) call for "simple protocols, analytical procedures, or logic flows, and some *do's and don'ts* or rules of thumb" that can augment the site-specific permit review process and improve in best professional judgment (Figure 1.2). Managers can use this approach to evaluate cumulative impacts until more rigorous research provides better alternatives.

The Synoptic Approach

The synoptic approach is an inexpensive, rapid assessment method that can assist managers and regulators in evaluating cumulative impacts within the regulatory constraints of tight schedules and budgets. Although research on the loss of wetland function is far from complete, the synoptic approach can support development of the best possible management strategies based on current knowledge.

Using the synoptic approach, wetland managers will be able to produce regional or statewide maps[3] that rank portions of the landscape according to synoptic indices. These maps and indices will enable permit reviewers to consider the landscape condition of the area in which a particular permit is proposed compared with other areas within their jurisdiction. By providing the environmental context in which wetlands occur, the maps also will allow wetland managers to examine wetland issues more comprehensively. Further, because the assessment is prepared at the same time for an entire state or region and not on a permit-by-permit basis, using this method will save time and money.

The synoptic approach consists of five steps (Table 1.1). Two major steps are definition of synoptic indices and selection of landscape indicators. The synoptic indices represent the actual functions and values within the particular environmental setting of interest. The landscape indicators are the actual data used to represent these indices. Choosing indicators often requires making simplifying assumptions because of limited information, time, and money. For example, agricultural area as measured from a land-use map could be a landscape indicator for agricultural nonpoint source nutrient loading, which would be the synoptic index for that particular management concern. The synoptic index and landscape indicator are defined separately to

[3] The end product of a synoptic assessment need not be a set of maps, but could consist solely of tabular data summaries. However, we believe that presentation as maps is more appropriate for the intended use, and gives a "big picture" overview that tables cannot provide.

keep them distinct, so we remember that agricultural area is not the management concern; it is only useful to the extent to which it represents nonpoint source nutrient loading.

The synoptic approach is flexible enough to cover a broad spectrum of management objectives and constraints. The specific synoptic indices and landscape indicators used in an application depend on the particular goals and constraints of the assessment. They also depend on the actual environmental setting. However, *this handbook does not provide a specific, detailed procedure for choosing the synoptic indices, nor does it supply a scientifically-tested list of landscape indicators having known confidence limits.* This is not possible, given our current state of knowledge and the strong dependency of the synoptic indices and landscape indicators on the particulars of the assessment. Instead, *the approach relies on the assessment team to make decisions, since they are best qualified to know their particular needs and constraints.* The synoptic approach provides the user with an ecologically-based framework in which local information and best professional judgment can be combined to address cumulative impacts and other landscape issues.

The synoptic approach is not a fixed procedure that always uses the same data sources and provides a standard end product. Rather, a synoptic assessment is a creative process that requires the manager to weigh the need for precision—as determined by management objectives—against the constraints: limited time, money, and information. An initial synoptic assessment could be conducted using the best available information and then updated as better data become available.

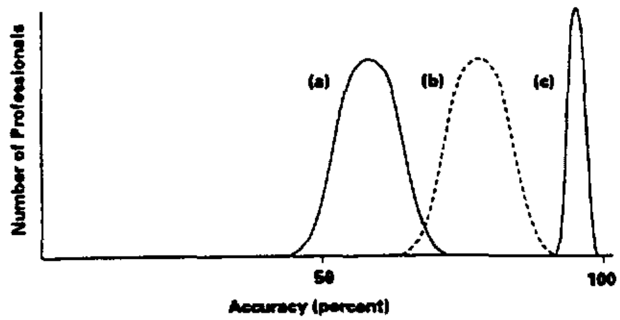
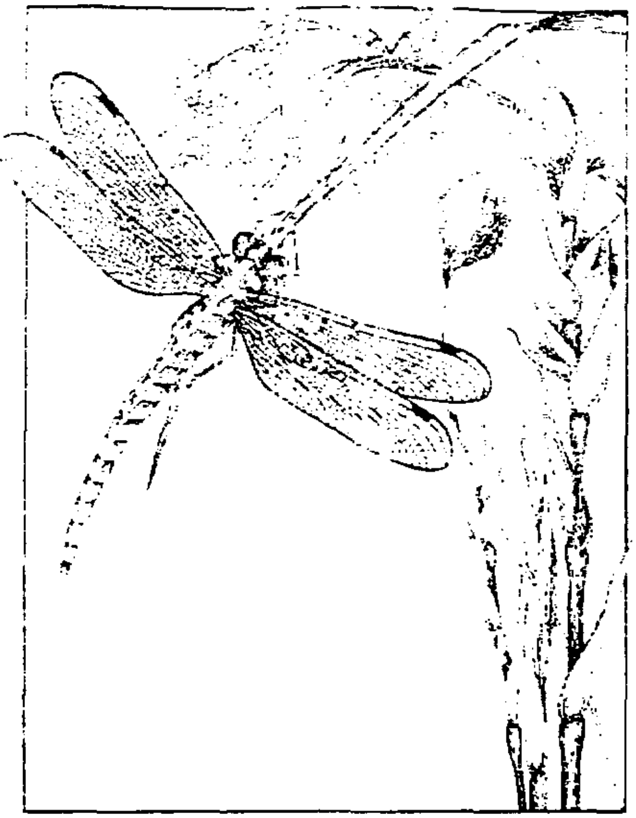


Figure 1.2. Improving best professional judgment (BPJ). "a" represents the hypothesized accuracy of BPJ under current conditions; most professionals probably give correct answers more than 50% of the time, and the most experienced professionals may be fairly accurate. However, the least experienced professionals may do worse than the flip of a coin, i.e., their answers may be wrong more often than right. A precise, quantitative assessment would greatly improve the accuracy of BPJ ("c") and reduce variability. However, such an assessment could be impractical within a regulatory context. The synoptic approach is a compromise that can be implemented within regulatory constraints and yet still improve the accuracy of BPJ ("b").

Table 1.1. Major steps in conducting a synoptic assessment.

Step 1.	Define Goals and Criteria
Step 2.	Define Synoptic Indices
Step 3.	Select Landscape Indicators
Step 4.	Conduct Assessment
Step 5.	Prepare Synoptic Reports



Chapter 2 Ecological Basis for the Synoptic Indices

The synoptic approach provides a framework for making comparisons between landscape subunits[1] so cumulative impacts can be considered in management decisions. Comparisons are made by evaluating one or more landscape variables, or ***synoptic indices***, for each subunit. Defining the proper synoptic indices for a particular assessment is a critical step and depends on the environmental setting and the specific goals of the assessment. In this chapter, we provide an overview and rationale for the synoptic indices, drawing on concepts from three disciplines: ***systems ecology***, or the study of ecological systems (ecosystems), including their response to stress; ***landscape ecology***, which examines the interactions between ecosystems; and ***risk assessment***, which evaluates environmental risks associated with human actions.

Rationale for a Landscape Approach

The purpose of a cumulative impact assessment is to evaluate the cumulative environmental response to various impacts. Because no standard usage exists for the term, we define ***impact*** as a human-generated action or activity that either by design or by oversight alters the characteristics of one or more ecosystems; ***cumulative impacts*** are the sum of all individual impacts occurring over time and space, including those of the foreseeable future. We define ***effects*** as the physical, chemical, and biological changes that result from an impact, including direct and indirect changes that can be removed in time and space. ***Cumulative effects***, then, are the sum of all these changes resulting from cumulative impacts.

In conducting a cumulative impact assessment, we are particularly concerned with the loss of valued functions. These ***ecological functions*** are aggregate behaviors that arise from the many physical, chemical, and biological processes that take place in the environment. For example, whether a wetland reduces flood peaks depends on the processes that determine the wetland's hydrologic budget, e.g., precipitation, evapotranspiration, surface and groundwater inflows and outflows, and tidal input (Mitsch and Gosselink 1986).

Because an impact can affect more than one ecosystem and because an ecosystem can be affected by activities outside its boundaries, an assessment of cumulative impacts cannot be limited to a single ecosystem. Also, many ecological functions valued by society depend on interactions between ecosystems; they are more properly viewed as landscape functions, rather than ecosystem functions. For example, the water quality of a river is not determined by any one ecosystem but by

[1] Examples of possible subunits are counties, watersheds, and ecoregions; selection of subunits as part of a synoptic assessment is discussed in Chapter 3.

the aggregate effect and interaction of all ecosystems within its drainage area. The landscape is an appropriate unit for considering cumulative impacts, especially since landscape factors partially determine an ecosystem's response to cumulative impacts. For example, the survival of organisms following disturbance can depend on landscape characteristics such as corridor quality (Henein and Merriam 1990) and the degree of habitat fragmentation (Merriam and Wegner 1992; Stacey and Taper 1992).

Synoptic indices allow us to evaluate overall wetland condition for a particular landscape subunit through comparison with other subunits. Because the approach is not intended to provide a detailed landscape assessment, we must simplify and generalize our view of the landscape to ensure that relevant factors are included. The synoptic indices are therefore based on a simple model that describes ecosystem functions within the landscape and includes the effect of impacts on these functions. Because the focus of an assessment is *valued* ecological functions, concepts of risk assessment are also incorporated.

Landscape Model of Ecosystem Function

Forman and Godron (1986) have defined a landscape as "a heterogeneous land area composed of a cluster of interacting ecosystems that is repeated in similar form throughout." Wetlands, forests, lakes, and streams are examples of such ecosystems. Interactions occur through transfers of energy and material — including nutrients, minerals, and organisms — between ecosystems. A landscape can be viewed as a portion of the environment composed of ecosystems within which materials and energy are transferred as a result of various ecological processes. To further simplify this view, we will consider these ecosystems only as they affect the transfer of materials within and through the landscape.

At any time, a landscape contains a pool of materials [2] and energy being transferred between component ecosystems (as opposed to being cycled or stored *within* individual ecosystems). This dynamic state can be described by the aggregate flow of these materials within and through the landscape; it also includes the processes that drive or are controlled by these flows. Landscape functions result from these interactions, as in the earlier discussion of the effect of drainage area on river water quality. Ecosystems contribute to landscape functions by affecting (1) the quantity of transferred material, i.e., either increasing or decreasing the active pool; (2) the quality of the material, i.e., transforming it into different forms; or (3) the timing of material transfers, e.g., introducing a temporal lag in transfers or altering transfer rates.

From the simplest perspective, each component ecosystem can be considered to function as either a source or a sink for a given material. An ecosystem is a *source* if it causes a net increase in the total amount of material being transferred within the landscape (i.e., exports from the ecosystem are greater than imports into it); it is considered a *sink* if it causes a net reduction in the material flux [3]. We define these terms in the broadest sense, without regard to the specific processes responsible for the functions. For example, an ecosystem could function as a sink through biochemical conversion, filtration (e.g., removal of suspended materials from water as it passes through clays), or trapping (e.g., settling out of particulates from water). In the case of biological materials, an ecosystem would be a sink if emigration were less than immigration, which could occur if the death rate exceeded the birth rate (MacArthur and Wilson 1967; Pulliam 1988).

Because our definition of a sink is independent of causative processes, an ecosystem that induces a net transfer of materials to on-site storage would also be considered a sink since this would lead to a net reduction in the pool of materials. Conversely, an ecosystem that removes material from storage and returns it to the pool acts as a source. For example, a riparian forest acts as a sink where stream velocities are low and sediment storage increases through deposition; however, it acts as a source if high current velocities cause bank erosion, thereby removing sediment from storage (Pinay et al. 1992).

A landscape model that describes an ecosystem as either a source or a sink can easily account for the effect ecosystems can have on the quantity of transferred materials. When the status of the ecosystem as source or sink is dynamic, the model can also account for qualitative and timing effects. For example, an ecosystem that converts nitrate to molecular nitrogen through denitrification (a qualitative effect) would be described as a sink for nitrate and a source for molecular nitrogen. An ecosystem that stores water below ground during spring runoff functions as a sink at that time of year, then as a source during summer and fall, when it slowly releases the water from storage.

The ability and degree to which an ecosystem functions as a source or a sink is controlled by on-site conditions, such as local hydrology and geomorphology, soil and vegetative characteristics, nutrient availability, and population densities. However, an ecosystem with the potential to reduce material flows could not function as a sink if the particular material was unavailable. In

[2] We define materials broadly to include biotic and abiotic materials.

[3] An ecosystem could be neither a source nor a sink if exports are equal to imports. Such an ecosystem would be neutral with respect to changes in the magnitude of landscape flows. However, such an ecosystem could still affect the distribution of materials; see Chapter 6.

other words, an ecosystem can reduce the pool of active landscape materials only if it is connected to at least one source. Thus the ability of an ecosystem to function as a sink depends on two factors: the ***assimilative capacity***, which is the amount of material the ecosystem could remove, assuming it was available; and ***landscape input***, which is the amount of material imported into the ecosystem from source ecosystems[4]. While capacity is controlled by characteristics *within* the ecosystem, landscape input is determined by interactions *between* ecosystems and depends on (1) the magnitude of the various sources, (2) where these sources are located relative to the target ecosystem, (3) the transport mechanism of the particular material (e.g., passive diffusion, wind-borne dispersion, gravity flow, or migratory movement in animals), and (4) the occurrence of any sinks along the transfer pathway.

Phosphorus retention by a wetland is one example of how capacity and landscape input control sink functions. A wetland's capacity to retain phosphorus depends on factors such as plant uptake; the concentrations of minerals that precipitate phosphorus (e.g., ferric iron and aluminum); soil pH, which affects phosphorus solubility; and adsorption to soil constituents such as clays and organic matter (Mitsch and Gosselink 1986). The landscape input of phosphorus into the wetland depends on the types of neighboring ecosystems, land-use practices outside the wetland (e.g., fertilizer application rates), and landscape characteristics that control sedimentation rates into the wetland, such as slope.

According to the model we have been describing, the landscape is a collection of source and sink ecosystems embedded within a matrix of neutral ecosystems. Although this is somewhat simplistic and ignores actual processes, simplifying the overwhelming complexity of a real landscape is necessary if overall function is to become understandable. This model allows us to visualize the landscape as a dynamic network of interacting ecosystems, each of which can affect the quantity, quality, and timing of the materials transferred within the landscape. It also provides a framework that allows us to consider the effect of impacts on landscape function.

Effect of Impacts on Landscape Function

It is important to differentiate between an activity (the impact) and the ecological response to it (the effect), because many environmental regulations target activities (e.g., discharge of dredge and fill materials under CWA Section 404). Numerous ecosystem characteristics could be altered by an impact. Lugo (1978) developed a generic model that described five ways in which an ecosystem could be stressed. We further aggregate these to define three general types of impact based on the type of characteristic being altered (Figure 2.1):

[4] As defined here, the capacity is the net amount of material that can be removed, after accounting for removal of on-site material. If gross capacity is preferred, landscape input would have to include on-site production.

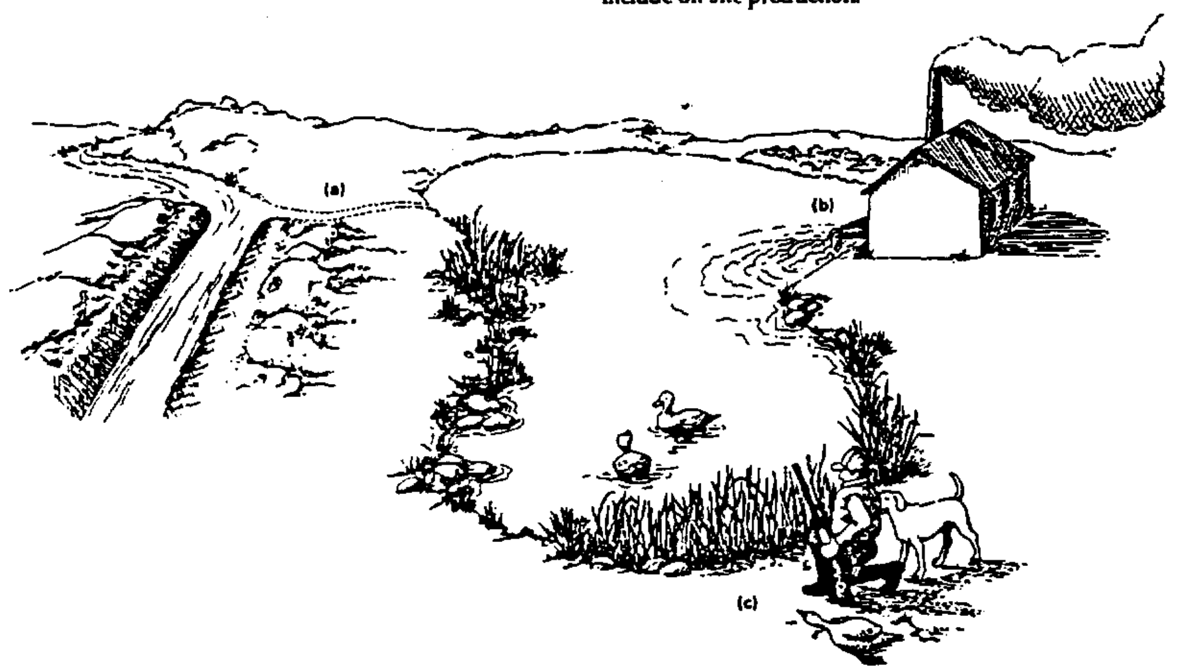


Figure 2.1 Generic model of ecosystem impacts. An impact can affect external driving factors (forcing functions) before they cross the ecosystem boundary, e.g., hydrologic diversion (a); an impact can affect ecosystem processes, e.g., discharge of industrial pollutants that alter productivity (b); and an impact can alter ecosystem structure, e.g., harvesting wildlife through hunting (c).

- Changes in forcing functions — Ecosystems are ultimately driven by material and energy flows that originate outside their boundaries. These driving factors are referred to as *forcing functions*. For example, sunlight is the ultimate forcing function for most ecosystems, and hydrologic input (in the form of surface water, groundwater, or tides) is an important driving factor for wetlands. Forcing functions can be diverted or reduced in magnitude, or the timing can be changed. New forcing functions to which the system is not adapted can be introduced, or the magnitude of an existing factor can be increased beyond its natural range.
- Changes in ecosystem process — Processes such as production or respiration can be stimulated or depressed, and material or energy distribution within the ecosystem can be altered.
- Changes in structure — *Structure*, built from energy and raw materials, is the collection of an ecosystem's physical, chemical, and biological characteristics. Biological examples of ecosystem structure include the various organisms, their complex behaviors, trophic relationships between organisms, seed banks that maintain biodiversity, and even dead matter. Physical structure includes concentrations of raw materials, such as lake water. Examples of structural impacts include harvesting of organisms by hunting or farming, introduction of domestic species not naturally present, reductions in water level through drainage, and destruction of soil structure by compaction.

In general, ecosystems affected by stress exhibit the following properties (Odum 1985): (1) internal material cycling is reduced, (2) the community reverts to earlier successional stages, (3) efficiency of resource use declines, and (4) parasitism increases. In stressed ecosystems, native species can be replaced by opportunistic species; this is especially significant in wetlands, where invasion by weedy species such as purple loosestrife can alter community structure (Wilcox 1989).

Not only does the environment respond to individual impacts, it also responds to them cumulatively. Examples of cumulative impacts and cumulative effects appear in Table 2.1. Bormann (1987) described seven stages of ecosystem stress, ranging from insignificant effects at low levels of pollution to complete ecosystem collapse under continued, severe pollution (Table 2.2). Although based on air pollution, these seven stages could represent a general model of ecosystem response to cumulative impacts. From a landscape perspective, the ultimate consequence of these changes is a loss of ecosystem function. This translates into a change in the ability of an ecosystem to act as a source or a sink either quantitatively (an increase or a decrease in the existing level of function) or qualitatively (e.g., a change from source to sink or vice versa).

The boundaries for cumulative impacts and cumulative effects need not coincide. Some cumulative effects could occur outside a cumulative impact boundary; conversely, cumulative effects within an area could partially result from impacts occurring outside the boundary. If the objective is to determine the cumulative effects within a specific area, a larger boundary must be defined that includes impacts to external forcing functions.

Synoptic Indices

Based on these principles, we define four synoptic indices for assessing cumulative impacts and relative risk: function, value, functional loss, and replacement potential. These indices are landscape-level measures, so each is evaluated for an entire landscape subunit, rather than for an individual component ecosystem. Although the indices are generic and could be applied to any ecosystem type, we discuss each as it applies specifically to wetlands. The hierarchical evaluation of these indices as part of a risk assessment can be found in Leibowitz et al. (1992).

Table 2.1. Typology of cumulative impacts and cumulative effects (after Beanlands et al. 1986).

Cumulative Impact	Description
Time-crowded Perturbations	Disturbances that are so frequent in time that the ecosystem does not have the chance to recover between disturbances
Space-crowded Perturbations	Disturbances that are so close in space that their effects overlap
Cumulative Effect	**Description**
Synergisms	Interaction of different types of disturbance to produce a response that is qualitatively and quantitatively different than the separate effects combined
Indirect Effects	Effects that are produced through a complex pathway and that are removed in time and/ or distance from the initial disturbance
Nibbling	Simple additive effects that result from cumulative impacts

Table 2.2. Model of ecosystem response to increasing stress (adapted from Bormann 1987).

Stress Level	Ecosystem Response
Insignificant	Insignificant
Low levels	Relatively unaffected; ecosystem may function as a sink
Levels inimical to some species	Changes in competitive ability of sensitive species; selection of resistant genotypes; little effect on biotic regulation
Increased stress	Resistant species substitute for sensitive ones; some niches opened for lack of substitutes; biotic regulation may be disrupted, but may return as system becomes wholly populated by resistant species
Severe levels	Large plants, trees, shrubs of all species die off; ecosystem converted to open-small shrubs, weedy herb system; biotic regulation severely diminished; increased runoff, erosion, nutrient loss
Continued severe stress	Ecosystem collapse; completely degraded ecosystem; ecosystem seeks lower level of stability with much less control over energy flow and little biotic regulation

Function

Wetlands are capable of performing various functions as a result of physical, chemical, and biological processes. These functions can be divided into three general categories:

- Habitat functions — Providing support for wetland-dependent species, including food, shelter, and breeding sites;
- Water quality functions — Water quality improvement, nutrient cycling and supply; and
- Hydrologic functions — Flood attenuation and moderation of hydrologic flow.

The *function* index refers to the total amount of a particular function a wetland provides within a landscape subunit *without consideration of benefits*. The index is the rate at which material or energy is added to or removed from the active landscape pool. In the case of a sink function, the index is separated into two components[5]: *capacity*, which is the maximum net amount of material that could be removed by a subunit's wetlands if the supply of material were unlimited; and *landscape input*, or the total amount of the material imported into wetlands from contributing sources.

Value

Environmental regulations such as the Clean Water Act consider both ecosystem functions and their impact on public welfare (Preston and Bedford 1988; Westman 1985); thus we identified *valued* ecological functions as the target of a cumulative impact assessment. Wetlands can be valued for the tangible benefits they provide, such as clean water or hunting, or for intangible benefits such as aesthetics. However, values are highly subjective, and a wetland characteristic valued by one individual could be perceived as a liability to another. Even when the wetland provides a service that benefits the individual (such as improved water quality), the service could be undervalued because of poor information or conflicting goals.

Whether a particular ecological function is considered valuable is not a technical issue, but must be determined by the policy maker initiating the synoptic assessment. Such a decision might be based on law or on agency mandate. For example, by enacting the Endangered Species Act, Congress has determined that endangered species are valuable; similarly, an agency mandated with protection of drinking water would value functions that improve water quality. Policy makers could determine values through public input, interagency consensus or both. Gosselink and Lee discuss policy considerations and the importance of goal-setting as part of a cumulative impact assessment (Gosselink and Lee 1989; Lee and Gosselink 1988). A framework for including the effects of cumulative impacts on programmatic decisions is given in Irwin and Rodes (1992).

Once it is decided that a particular function is important, the *value* index can be used to determine the relative value of that function within each landscape subunit. This ranking depends on two factors. First, value is related to overall level of function, although this need not be a linear relationship (e.g., there could be diminishing returns at higher functional levels). Second, a function may be considered valuable not because of its inherent value, but because it acts upon something else valued by society. In such instances, the overall value also depends on the occurrence of this valued object. For example, flood reduction has no inherent value; it is

[5] These two sub-components are similar to the terms "effectiveness" and "opportunity" used in the Wetland Evaluation Technique (Adamus 1983). However, the synoptic terms and their meaning are derived from the previously described landscape model.

valued because it reduces property damage and human injuries and deaths. Dams are not necessarily built where the largest floods occur, but where floods threaten human populations, valuable property, or both. Valued objects can also include plants and animals; the value of wetlands for habitat could increase with the number of rare and endangered species supported by that habitat. This index can also include future values by considering the future benefits of these functions. Finally, we note that this index does not represent economic value, since it does not consider market factors, etc. Instead, it provides an estimate of the value provided by a function within a landscape subunit, relative to other subunits.

Functional Loss

Functional loss represents the cumulative effects on a particular valued function that have occurred within a subunit. Functional loss caused by changes in forcing functions, processes, and structure should all be considered. The index should include complete loss of function from *conversion*, where the ecosystem is changed into a different ecosystem or land use (e.g., filling in a wetland to build a home), and partial loss through *degradation*, where the impact does not change the ecosystem type but alters function (e.g., reduced production through pesticide contamination). Future loss should also be considered as called for by Council on Environmental Quality regulations (40 CFR Sect. 1508.7).

Functional loss depends on the characteristics of the impact, including the type of impact, its magnitude, timing, and duration; and ecosystem *resistance*, or the relative sensitivity of the ecosystem to the impact, based on its robustness and overall health (see Chapter 6).

Replacement Potential

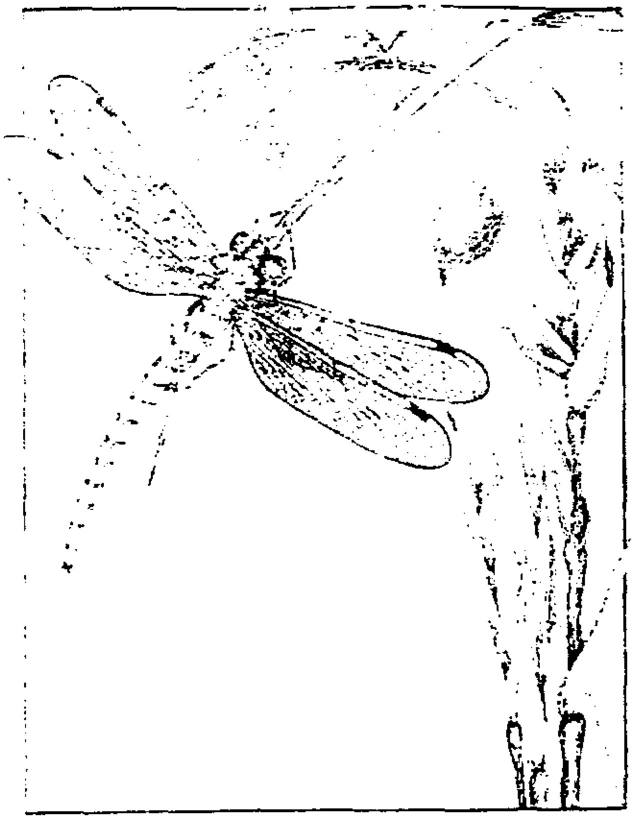
Replacement potential refers to the ability to replace a wetland and its valued functions. In this case, we are referring to functional replacement carried out by people; however, natural recovery could also be considered. Although not a component of a cumulative impact assessment *per se*, replacement potential is included as a synoptic index because it is a consideration within the 404 permit process and is also an important component of risk assessment (Leibowitz et al. 1992). The ability to offset the loss of valued functions and reduce ecological risk is greater if replacement potential is high; conversely, protection is more critical for risk reduction if replacement potential is low.

Replacement potential depends on many factors specific to the particular wetland, such as the type of wetland, the function to be restored, and, in the case of restoration, the kind of impact that altered the original wetland (Kentula et al. 1992; Kusler and Kentula 1990). In a synoptic assessment, however, we are more concerned with the landscape factors that contribute to replacement potential. Because it is more difficult to replace a wetland if critical driving factors have been disrupted, this index depends on the overall environmental condition of the subunit. For example, it would be difficult to restore a swamp within a historical flood plain if a levee had been constructed on the river. If restoration did take place, the wetland probably would not be sustainable because natural overbank flooding, which was a driving factor for the original swamp, would be disrupted.

Synoptic Index Evaluation

In conducting a synoptic assessment, it is necessary to refine the general synoptic indices into a specific set of indices that are most relevant to management concerns within a particular landscape setting. For example, in an application concerned with nonpoint source nitrogen pollution within an agricultural region, the specific indices for capacity and landscape input might be the maximum denitrification rate and the nitrate loading rate, respectively. However, quantifying the specific indices accurately for large landscape subunits would be difficult if not impossible. In order to evaluate the indices, the synoptic approach uses landscape *indicators* of actual functions, values, and effects. The indicators are first-order approximations that represent some particular index, given certain assumptions (see discussion in Chapter 3, Step 3.5). For example, data on agricultural nonpoint source nitrate loadings might not be available, in which case agricultural area could be used as a first-order landscape indicator.

In addition, we often take a risk-based approach to estimate specific indices. For example, we may not be able to quantify the actual loss of hydrologic function due to cumulative impacts, but we could assume that the risk of actual loss is greater in areas with high function and high cumulative impacts, compared with areas having low function and low impacts. Such an approach will undoubtedly make errors in assigning a relative ranking to each landscape subunit. However, a synoptic assessment need not provide a perfect evaluation of cumulative effects. The goal is to provide information that will improve permit evaluation and management decisions overall.



Chapter 3 Conducting a Synoptic Assessment

The process of producing a synoptic assesment involves five steps (Table 3.1). Although presented and discussed sequentially, it might be necessary in an actual application to follow these steps iteratively. We suggest that information resulting from this process not be viewed as the ultimate end product, but that synoptic assessments be updated periodically to reflect changing objectives and environmental conditions or to incorporate better data. Further, it may not be possible to achieve the desired management objectives in a one- or two-year period. By producing an initial assessment and improving it over time, an agency can obtain the desired results over the long run while gaining useful short-run results. A synoptic assessment should be an iterative process.

Preparation of a synoptic assessment requires the efforts of a team of individuals having different backgrounds and responsibilities (in an actual assessment, these roles need not literally be performed separately by three individuals):

- The manager, who is in charge of the resource management program and who makes the decision to conduct a synoptic assessment, is the individual with primary responsibility for defining the overall goals of the assessment.
- The resource specialist, who is the ultimate user of the final maps (e.g., a permit reviewer) and who is familiar with the area's wetland resources and their ecological functions, has the primary responsibility for defining the ecological relationships relevant to the particular management objectives.
- The technical analyst, who assembles the data, makes measurements, calculates the index values, and then maps them, should be familiar with database management and geographic information systems (GIS) or computerized mapping.

Step 1: Define Goals and Criteria

The purpose of this step is to identify explicitly the assessment objectives, intended use, required accuracy level, and the constraints within which the assessment will be conducted. Often the objectives call for more accuracy and detail than constraints allow. This step may require repetition until an acceptable combination of objectives, accuracy, and resource allocation is agreed upon.

Step 1.1 – Define Assessment Objectives

The general objectives of the assessment depend on the overall mission and goals of the particular agency or organization conducting it. If the manager works within a Department of Environmental Quality, the focus could be wetland water quality functions. A manager

Table 3.1. Steps in conducting a synoptic assessment.

Steps	Procedures
1. Define Goals and Criteria	1.1 Define Assessment Objectives
	1.2 Define Intended Use
	1.3 Assess Accuracy Needs
	1.4 Identify Assessment Constraints
2. Define Synoptic Indices	2.1 Identify Wetland Types
	2.2 Describe Natural Setting
	2.3 Define Landscape Boundary
	2.4 Define Wetland Functions
	2.5 Define Wetland Values
	2.6 Identify Significant Impacts
	2.7 Select Landscape Subunits
	2.8 Define Combination Rules
3. Select Landscape Indicators	3.1 Survey Data and Existing Methods
	3.2 Assess Data Adequacy
	3.3 Evaluate Costs of Better Data
	3.4 Compare and Select Indicators
	3.5 Describe Indicator Assumptions
	3.6 Finalize Subunit Selection
	3.7 Conduct Pre-Analysis Review
4. Conduct Assessment	4.1 Plan Quality Assurance/Quality Control
	4.2 Perform Map Measurements
	4.3 Analyze Data
	4.4 Produce Maps
	4.5 Assess Accuracy
	4.6 Conduct Post-Analysis Review
5. Prepare Synoptic Reports	5.1 Prepare User's Guide
	5.2 Prepare Assessment Documentation

for the Fish and Game Division might be particularly interested in wetland habitat functions. A manager of a wetland protection program, however, might be interested in not just one particular function but in several functions or in wetland restoration. The management objectives could be very specific, e.g., determination of wetland degradation caused by superfund sites, protection of wetland habitat for sport fish, protection of floodplain wetlands, etc.

During this step, the boundary for the study unit needs to be defined explicitly. This would typically be either a political boundary, based on the agency's jurisdiction (a state or multi-county region) or a natural boundary, e.g., a natural watershed or geomorphological province. The study area could be of special interest to management (one for which a special area management plan is being developed). It may be necessary to get input from other agencies or interested parties before finalizing the boundary.

Step 1.2 – Define Intended Use

The manager should define how assessment results will be applied. The assessment could be used to support very specific decisions, e.g., to support cumulative impact assessment as part of Section 404 permit review, or it could be used for general planning, e.g., to help identify areas sensitive to future impacts as part of a State Wetland Conservation Plan. The particular use affects the level of accuracy required and the degree of review the final products must undergo. In addition, an assessment used as part of a regulatory program might need to meet specific legal tests or require public comment or interagency consensus. The manager should also determine whether the assessment is to be purely technical or whether political considerations need to be included.

Step 1.3 – Assess Accuracy Needs

The overall management objectives and intended use of the information determine the level of uncertainty the manager is willing to accept in decisions that make use of a synoptic assessment. EPA guidelines on data quality assurance refer to the process of selecting the level of accuracy needed as defining the data quality objectives. This process includes five steps (EPA 1989):

- Define the decision;
- Describe the information needed for the decision;
- Define the use of environmental data;
- Define the consequences of an incorrect decision attributable to inadequate environmental data; and
- Estimate available resources.

The previous sections covered the first three steps of this process. Since any analysis has a level of uncertainty, and thus the chance of erroneous conclusions, the manager must consider the repercussions of incorrect decisions based on the level of uncertainty. If it could lead to litigation, for example, an assessment developed for regulatory applications might require a high confidence level. If the assessment is being conducted for broad-scale planning using best professional judgment, results might be sufficient as long as they are "more right than wrong." In other words, results need not be completely accurate; rather, the data must be adequate for the stated purposes of the assessment. The manager, in consultation with other team members, must define the level of accuracy needed for an assessment so the benefits outweigh the liabilities. Estimating available resources is discussed in the following section.

Step 1.4 – Identify Assessment Constraints

The manager must estimate the amount of time, money, and personnel hours that can be committed to the project. Regardless of the objectives and needs for accuracy, the effort will be limited by available resources.

As an example of possible assessment costs, the Louisiana and Washington pilot projects that are discussed in Chapter 4 each took a year and a half for completion and required a half-time senior scientist and both a full-time and half-time technical analyst (i.e., two full-time equivalents per year for each project). Much of the technical analysts' time was spent collecting data from various agencies, conducting quality control checks, performing map calculations, digitizing, and creating various databases. Other costs included approximately $20,000 for supplies and materials (excluding data, which mostly were obtained from cooperating agencies), plus access to a GIS. Although the purpose of the pilots was methods and development, and not an actual application, costs for a similar statewide analysis should be comparable. At the opposite extreme, an application requiring high precision and field verification could easily require several years of effort and cost hundreds of thousands of dollars for data collection, analysis, and labor. Project costs depend on study area extent and whether adequate data already exist (Steps 3.1-3.3).

The team should also consider other constraints that influence the outcome of an assessment, such as legal requirements, agency mandates, institutional constraints, and the need for public comment or interagency coordination.

If the resources available for an assessment are much less than what is deemed necessary based on best professional judgment (Steps 1.1-1.3), then management can change the objectives (e.g., assess a smaller area or accept less accurate results), relax the constraints (find a source of extra funding), or conclude that the assessment is not feasible at that time.

Step 2: Define Synoptic Indices

Once the objectives have been determined, the resource specialist must define a specific set of synoptic indices that will meet the objectives and intended use of the assessment. This involves replacing the four generic indices (function, value, functional loss, and replacement potential) with a set of indices specific to the objectives.

Defining the specific indices and the factors they include requires an understanding of the interactions between wetlands and regional landscapes. To summarize this understanding, the resource specialist can provide a landscape description that includes wetland types, functions and related societal values, natural factors sustaining the wetlands and major impacts (Table 3.2).

The resource specialist can consult with regional experts for assistance in determining these interactions, for example:

- University or state Soil Conservation Service (SCS) soil scientists are familiar with regional factors affecting denitrification capacity and adsorption potential (e.g., percent of organic matter);

Table 3.2. Examples of landscape descriptions that can be used in selecting indices.

Category	Example 1
Management Objective	Develop risk assessment guidance for county planners to protect sparse wetland populations of central Washington for waterfowl and other wildlife habitat.
Wetland Type	Palustrine (emergent, scrub-scrub and forested) on floodplains; saline (scrub-scrub) in playas and wind created depressions (Canning and Stevens 1989).
Natural Setting	Basin, characterized by loess deposits and deep dry channels cut into basalt, surrounded by mountain ranges which provide hydrologic inputs; arid climate (23-64 cm average annual precipitation); streams predominantly influent, many go dry in dry years (Omernik and Gallant 1988).
Landscape Boundary	Columbia Basin in Central Washington.
Significant Impacts	Water withdrawal for irrigation; altered water quality and stream morphology from grazing; high nutrient and suspended sediments from agriculture and mining.
Specific Indices	Habitat support, low stream flow and hydrologic modification (water withdrawal); non-point source pollution.
Landscape Subunits	Subwatersheds and county boundaries.
Category	**Example 2**
Management Objective	Include cumulative impacts as part of 404 permit review in Southern California.
Wetland Type	Intertidal salt marshes.
Natural Setting	Mediterranean climate, accretion and erosion of sediments, warm ocean current from Mexico, tidal flushing. Natural perturbations include storm events and catastrophic sedimentation; drought; lagoon closure (Zedler 1982).
Landscape Boundary	Southern California coast including intertidal slopes in river valleys, from Point Conception to the international border with Mexico.
Significant Impacts	Urban development (dredge and fill disposal); reduced circulation from anthropogenic sedimentation; altered watershed hydrology (Zedler 1982).
Specific Indices	Cumulative wetland loss, suspended sediment loading, peak discharge, hydrologic modification.
Landscape Subunits	Coastal watersheds.

- Hydrologists with universities or the state office of the U.S. Geological Survey (USGS) can provide insight into the hydrologic factors that form wetlands, and can also provide information on hydrologic modifications that may affect wetland functions;
- Biologists with the U.S. Fish and Wildlife Service (USFWS), state agencies, or the Nature Conservancy/Natural Heritage Program can provide expertise on wetland habitat and wetland-dependent species; and
- Biologists with the SCS and other agencies will be familiar with wetlands in agricultural settings, as well as with opportunities for restoration.

Other valuable resources are USFWS "Community Profile" reports. Each of these reports provides a wealth of information on a regional wetland type and often includes discussions of geological/climatic setting, natural forcing functions, ecological functions, ecosystem structure, and degradation by human impacts.

Step 2.1 – Identify Wetland Types

The first step in developing synoptic indices is to compile a list of the major wetland types found in the assessment area, e.g., specific wetland communities. This list can be limited to a particular type of wetland if management objectives are narrow, or it can include all of the area's wetlands if objectives are broad. The identification of these wetland types can be based on popular classifications (e.g., marsh, bog, or pothole), a functional classification (e.g., Novitzki 1979; O'Brien and Motts 1980), or the more detailed system developed by USFWS (Cowardin et al. 1979). The choice of classification should match the assessment objectives and constraints. For example, if protection of wetlands for flood control is the primary objective, the analyst could focus on palustrine or floodplain wetlands as defined by the Cowardin system or floodplain/river lower perennial wetlands as defined by a hydrogeomorphic classification (personal communication, M. Brinson, East Carolina University, Greenville, North Carolina). If, however, the objective is protection of wetlands for environmental education, then unique or rare wetlands near urban areas could be classified using a popular system or one defined by the State Heritage Program. Where the objective is to assess cumulative impacts, it will be important to select a classification that is broad and synthetic.

Selection of a particular wetland classification scheme also depends upon the availability of information. For example, if National Wetland Inventory (NWI) maps are available for the region, the Cowardin classification is a logical choice. At the minimum, the classification should include or be cross-referenced with information on geomorphic setting and source of water because both are important components of the natural setting (Step 2.2) and are useful for identifying significant impacts (Step 2.6).

Step 2.2 – Describe Natural Setting

The analyst should understand the landscape driving factors or forcing functions responsible for the formation and maintenance of wetlands because this information is important for defining landscape boundaries (Step 2.3) and for evaluating the significance of impacts (Step 2.6). The natural factors include natural stresses, such as drought, and structural components, such as soil and seed banks (see Chapter 6). The classification used to identify wetland types (Step 2.1) should provide relevant information. A broad-scale or detailed description of natural factors can be developed around a series of questions such as those listed in Table 3.3.

Step 2.3 – Define Landscape Boundary

In Chapter 2 we noted that the boundaries for cumulative impacts and cumulative effects need not be the same; the cumulative effects occurring within a given area could result partially from impacts that take place outside the boundary. The resource specialist must define the landscape boundary to include the appropriate natural setting (Step 2.2) and impacts (Step 2.6) that could be operating outside the study area. Even if the actual analysis ignores this larger boundary, the boundary must be defined so the resource specialist can determine the degree to which the assessment might be ignoring important factors.

Because hydrology is the single most important determinant of wetland type and function, the landscape boundary should include at least the entire drainage area in which the study is located. For example, an assessment of the state of Louisiana cannot stop at the state boundary but must consider hydrologic input from upstream segments of the Mississippi, Red, Sabine, Ouachita, and Pearl rivers. The landscape boundary for groundwater discharge wetlands might include recharge areas hundreds of miles outside the study area; likewise, the boundary for coastal wetlands will probably include estuarine, nearshore, and even offshore waters. These hydrologic boundaries also delimit many water quality processes, such as transport of nutrients, sediments, and pollutants.

Defining the boundary for habitat processes is more problematic than for the other functions. Biotic factors operate on scales defined by the ranges of wetland-dependent species. Given the diversity of species, no single spatial unit can encompass all species' ranges for a particular study area. Many times, ecoregions provide useful landscape units for habitat support (Omernik 1987); research by Inkley and Anderson (1982) and Larsen et al. (1986) demonstrates a correspondence between ecoregions and wildlife and fish communities, respectively. If habitat of wide-ranging migratory species is an important element of the assessment, a broader landscape boundary must be defined.

Table 3.3. Examples of technical questions that could be used to describe the natural factors determining wetland function.

	Technical Questions
Describing natural wetland setting related to forcing functions, ecosystem processes, and structure:	What are the geological processes responsible for the wetlands' formation, e.g., deposition of marine or riverine sediments, glaciation?
	What are the physiographic characteristics associated with the wetlands, e.g., large depressions, river valleys, karst topography?
	What are the hydrologic influences, e.g., tidal, riverine or lacustrine energy, or groundwater influence?
	What are the climatic influences, e.g., timing, type and amount of precipitation, length of growing season?
	What are the chemical characteristics and fluxes of the wetlands, e.g., salinity, organic content, nutrient and mineral availability?
	What are the natural perturbations that wetlands are either adapted to or dependent on, e.g., fire dependent species, periodic inundation, seasonal drought?

Step 2.4 – Define Wetland Functions

The resource specialist next defines the particular wetland functions to be addressed. Depending on management objectives, the functions of interest could be either specific or broad. Because it is impossible to assess all functions, even when the objectives are general, the specialist must determine a subset of functions that best represents the broader class. For example, consideration of hydrologic function in regions where small, non-tidal wetlands prevail might include wetland influence on peak flow but not on storm surges, which occur mainly in larger, tidal wetlands.

Habitat functions can be defined by determining the various species (including birds, fish, and mammals) that are dependent on or utilize the wetland communities identified in Step 2.1. For hydrologic and water quality functions, wetlands often function as sinks. Therefore it is useful to consider the hydrologic and water quality sources that are found within the particular landscape setting, since the source is a component of sink functions (Chapter 2). Natural and anthropogenic sources should both be included. Chapter 7 provides a detailed discussion of wetland functions that have been reported in the literature and can serve as a source of candidate functions that should be considered during this step.

Step 2.5 – Define Wetland Values

As discussed in Chapter 2, whether a function is valued is a policy decision rather than a technical consideration. These valued functions could be a given, based on the objectives. However, the manager might choose to map the relative magnitude of many functions first, then use this information to determine which wetland functions are most valuable. If so, the manager has deferred the valuation until after analysis. In either instance, the value may also depend on the co-occurrence of the function and "valued objects" such as property.

To define a synoptic index for value, the team must determine who ultimately benefits from the various wetland functions and whether other valued objects are involved (see discussion on value, Chapter 2). For example, they might decide that the value of flood protection is low if it occurs mostly in uninhabited regions or that the value of water quality improvement is very high if it occurs in areas that supply drinking water to large urban centers.

Functions and values are kept distinct by defining them in separate steps. This allows the team to consider whether important ecological functions, based on technical information, are being undervalued in terms of social perceptions.

Step 2.6 – Identify Significant Impacts

In this step, the resource specialist determines the most significant impacts on the functions of interest. If the proportion of recent wetland conversion within a particular region is high, it may be the dominant cause of functional loss, in which case other factors may be assigned lower priority. In this case, the index for functional loss would be loss of wetland area.

If conversion in the region is insignificant or if the specialist thinks conversion is not the dominant cause of functional loss, then the impacts most likely to cause wetland degradation must be identified. Tables 3.4 and 3.5 are examples of how best professional judgment could be organized to guide this process. Table 3.4 contains a list of impacts associated with agriculture along with the type of degradation each is expected to produce. Similar tables for other major classes of wetland impacts (resource extraction, urbanization, and water management) appear in Appendix B. Using Table 3.4 or a modification, the specialist can identify significant types of degradation that would result from commonly occurring impacts. Then the specialist could use Table 3.5 to determine which hydrologic functions would most likely be affected by these impacts (similar tables for water quality and habitat functions appear in Appendix C). The tables can be used in reverse order to determine which impacts would most likely degrade a given function.

As an example, in a state where livestock ranching is a major agricultural activity, possible impacts include fertilizers, harvesting, pesticides, species introduction, trampling, and water consumption (Table 3.4). Based on familiarity with the region, the specialist might decide that harvesting and trampling are the two most common impacts. Both have a high likelihood of causing degradation through changes in behavior or habits of wetland animals resulting from habitat alteration, and both have a medium likelihood of causing denudation (Table 3.4). If the overall function of interest is hydrology, Table 3.5 indicates that functional loss from changes in animal behavior is not likely.

These tables represent hypotheses about the mechanistic linkages between impacts, degradation, and functions; they are an example of how best professional judgment could be used to guide the selection process. The resource specialist should consult regional experts to ascertain whether these relationships hold true in the specific study area.

Step 2.7 – Select Landscape Subunits

At this time the resource specialist defines the landscape subunits that will be the basis for making relative comparisons and reporting results. For now, the decision

Table 3.4. Typical relationships expected between agricultural impacts and wetland degradation based on best professional judgment. Letter indicates degree of expected association and not the intensity or duration of impact (H = high, M = medium, L = low).

Impact	Acidification	Altered Animal Behavior	Compaction	Contamination/Toxicity	Denudation
Channelization[3]					H
Drainage[3,4]	L	H	L	M	
Fertilizers[1–5]	L			M	M
Fill[2,3]	L	H	H	L	H
Harvesting or Burning[1–5]	M	H[1–3]			M[2]
Impoundment[1]		H		M	
Irrigation/Flooding[3]	L	M		M	
Pesticides[1–5]				H	M
Species Introduction[1–5]		H			
Tillage[3]	L	L			H
Trampling[1–5]		H	L		M
Vehicles/Boats/Planes[1–4]		M	M	L	L
Water Consumption[1–5]				M	

Impact	Dehydration	Eutrophication/Enrichment	Erosion	Inundation	Light Reduction
Channelization[3]	M	M	M		L
Drainage[3,4]	H	M	M		M
Fertilizers[1-5]		H			L
Fill[2,3]	H	M	L		H
Harvesting and Burning[1-5]				M[2]	
Impoundment[1]		M	L	H	M
Irrigation/Flooding[3]		M	M	H	M
Pesticides[1-5]					
Species Introduction[1-5]	L				L
Tillage[3]		M	H		
Trampling[1-5]			L		L
Vehicles/Boats/Planes[1-4]			M		L
Water Consumption[1-5]	H	M			

Impact	Salinization	Sedimentation	Surface Runoff Timing	Thermal Warming
Channelization[3]	L	L	H	M
Drainage[3,4]	L	M	H	
Fertilizers[1-5]	M			
Fill[2,3]	L	H	M	
Harvesting and Burning[1-5]		M[2]	M[2]	H[2]
Impoundment[1]	M	M	H	L
Irrigation/Flooding[3]	H	M	M	
Pesticides[1-5]				
Species Introduction[1-5]				
Tillage[3]	L	H	M	
Trampling[1-5]				
Vehicles/Boats/Planes[1-4]				
Water Consumption[1-5]	M		H	L

[1] Aquaculture (e.g., cranberries, rice, crayfish)
[2] Crops - No Till
[3] Crops - Till
[4] Forestry
[5] Livestock

Table 3.5. Effect of wetland degradation on hydrologic functions and degree of expected association based on best professional judgment (H = high, M = medium, L = low).

Type of Degradation	Peak Flow Reduction	Storm Surge Reduction	Water Conservation	Groundwater Exchange	Hydrologic Input
Acidification					
Animal Behavior					
Compaction	L	L		M	M
Contamination/Toxicity					
Denudation	M	M	M	H	M
Dehydration	H	H	H	H	H
Eutrophication/Enrichment			L	L	
Erosion		M			
Habitat Fragmentation				M	
Inundation	H	H	H	H	H
Light Reduction			L	L	
Salinization					
Sedimentation	M	L	M	M	
Surface Runoff	H	H	H	H	H
Thermal Warming			L	L	

should be based on management objectives and ecological considerations; data availability will be considered in Step 3. For assessments at the state or regional level, the USGS cataloging unit or a similar state unit might be most appropriate because it functions as a natural drainage area. Ecoregion subunits (see the previous section) or finer-resolution subunits, e.g., soil-vegetation associations, may also be useful. Selection of landscape subunits might also be based on political criteria, e.g., county boundaries.

Step 2.8 – Define Combination Rules

A specific synoptic index is typically a mathematical expression that includes several factors. Factors that may be combined in an index include components of an index (for example, capacity and landscape input could be components of function, and degradation and conversion could be components of functional loss) or other indices (e.g., an index of value would include function). Although a separate index could be defined for each of these factors (e.g., separate indices of functional loss through stormwater runoff and agricultural conversion), it is often desirable to mathematically combine them into a single index, in which case a set of combination rules needs to be defined. These combination rules must address the following questions:

- Will the factors be combined by addition, multiplication, or some other operation?
- Will the data be normalized, that is, adjusted to a common ordinal scale, prior to combination? If so, by what procedure?
- Will all factors be considered to contribute equally, or should weighting factors be applied to some?
- Will the same combination rules apply to all wetland types and across the entire range of conditions within the study area?

Decisions concerning combination rules are difficult and often subjective, but deserve careful attention to reduce error. Mathematical relationships between factors may be available from the literature or regional models. It is often necessary, however, to assume that factors have equal weight (i.e., are added without weighting factors) or that there is a first-order proportionality between factors, i.e., that the factors are multiplicatively combined. At the minimum, the resource specialist should explicitly describe the combination rules and any assumptions as part of the review (Step 3.7) and documentation (Step 5.2). Combination rules are further discussed in Hopkins (1977), O'Banion (1980), Skutch and Flowerdew (1976), Smith and Theberge (1987), and USFWS (1981).

Step 3: Select Landscape Indicators

Landscape indicators are the actual measures used to estimate the synoptic indices; either a single indicator or combination of indicators can be used. Selecting indicators requires balance between accuracy and cost. Major considerations are discussed below.

Selection of landscape indicators, which depends on data availability, should not begin until goals are defined (Step 1) and the relevant environmental variables are identified (Step 2). In order to evaluate the adequacy of an assessment (Step 4.5), it is important to keep the goals and environmental variables distinct from the trade-offs that occur because of data limitations. If data availability is considered too early on, real-world limitations begin to dominate the process before the goals and environmental variables are articulated. Goal setting, defining synoptic indices, and selecting landscape indicators should occur iteratively and not simultaneously.

Step 3.1 – Survey Data and Existing Methods

Contact various federal and state agencies having jurisdiction over the study area to determine what kind of environmental data are available; for smaller study areas, include county agencies. Other sources could be university experts and state and university libraries. The survey should include both mapped and tabular information available for the entire assessment area. (Examples of data that can be used for the various synoptic indices appear in Appendix D; sources for the data appear in Appendix E). As part of the survey, the technical analyst should also note the following types of information, which will be necessary for assessing data adequacy (Step 3.2):

- The purpose of the database and the type of information it contains;
- The methods used in collecting, measuring, and analyzing the data;
- Examples of how the data have been used, especially if reported as case studies;
- Known problems or limitations;
- Data format, e.g., hard copy or computer compatible;
- Availability of documentation, both for data collection and quality assurance procedures and, if appropriate, file formats for computerized databases;
- Procedure needed to acquire data, including cost.

The survey need not be limited to databases. Various existing methods and techniques can also be used to estimate indices. For example, the USGS collects discharge data at various sampling locations on many streams and rivers. Annual water resources data reports for each state provide summaries of these data; they are also entered into the WATSTORE database (see Appendix E). Unfortunately, monitoring stations are not typically at the locations needed for the synoptic assessment, e.g., at the lowest downstream point of the subunit. The technical analyst would have to select an indicator appropriate for estimating discharge at that location.

One possibility is to use regression equations published by most state USGS offices for estimating discharge using watershed characteristics. For example, variables for regression equations developed for eastern Mississippi include watershed area, channel slope, and mainstem channel length (Landers and Wilson 1991). Alternatively, mathematical models can estimate many variables; e.g., SCS's TR-55 (SCS 1986) and the USDA Agricultural Research Service's AGNPS model (Young et al. 1987) estimate peak discharge and agricultural nonpoint source pollution, respectively, from factors such as topography, precipitation, land use, and soils. The technical analyst can determine whether appropriate methods are available through a literature review, by conferring with regional experts, or both.

Step 3.2 – Assess Data Adequacy

Adequacy of existing data depends on several factors, including the degree to which an indicator based on the data represents the index and the quality of the data relative to the management objectives (Table 3.6). The following example illustrates the difference between these factors: For a synoptic index of peak discharge, two possible indicators are runoff volume as calculated by the "curve number" technique (SCS 1986) and discharge estimates produced by the USGS regression methods, discussed above. For the former, the physical quantity being estimated (volume) is different from the variable of interest (peak rate of discharge or volume/time). There is a relationship between runoff and peak discharge, but the two variables are not identical. However, the estimate of runoff could be accurate if based on high quality data. Conversely, an indicator based on the USGS regression represents the same physical quantity defined by the index, yet it could be unacceptable if calculated using poor quality data. Both of these issues must be taken into account. If an indicator that is physically different from the index is being considered, the resource specialist or technical analyst must determine whether the indicator represents a reasonable first-order approximation to the actual index and whether the use of that indicator is contingent upon any unreasonable assumptions (Step 3.5).

Potential indicator data should be evaluated according to a set of criteria (e.g., Table 3.6). The technical analyst must also consider extra effort required to translate the data into the format needed for the assessment. For example, data found in reports might require entry into a database. It is especially important to consider the extra effort required for processing mapped data. Do not assume that more detail is better until you consider the additional cost. For example, the use of 1:250,000 scale STATSGO soil maps, if available, may be much more appropriate for statewide synoptic assessments than 1:20,000 scale county soil survey maps because greater effort would be required to analyze the more detailed maps.

Step 3.3 – Evaluate Costs of Better Data

The technical analyst should assess the time and cost of obtaining better data. Identifying the types of data needed and the associated costs for producing results of various confidence levels is useful. For example, how much would the highest quality, most up-to-date information cost? What would be the gain in accuracy if the budget were increased by $10,000 or if two extra months were available for the assessment? These considerations would allow existing information to be compared.

Table 3.6. Example of objectives and related questions for defining landscape indicators for synoptic indices.

Objectives	*Technical Questions*
Determine how well the indicator *represents the index:*	Do comparable data exist for the entire study area or are there gaps that would limit intraregional comparison?
	Do standardized data exist for the appropriate time period, e.g., the past ten years, the entire year, or by season?
	Are data at the appropriate spatial scale or are there major scale differences between data sources?
	Are the classification systems used for wetlands and other landscape variables compatible? For example, the USFWS National Wetland Inventory maps, SCS soils maps and USGS Land Use/Land Cover maps classify wetlands according to different criteria.
Assess the quality of existing data:	What is the source of the data, e.g., agency or university?
	Can the originator (person or agency responsible for data collection) be contacted?
	When, where and how often were the data collected?
	What methods were used for the data collection?
	Was the data collection associated with a Quality Assurance program? If so, what information is available on the precision, accuracy, representativeness, comparability and *completeness* of the data?
	Are there assumptions, limitations or caveats to consider in using the database?
	What are the time, personnel and cost constraints of obtaining better data?
Determine level of confidence in the data:	What are the common assumptions between indicators and indices?
	What evidence would violate these assumptions?
	How should the weighing of variables be adjusted to compensate?

Step 3.4 – Compare and Select Indicators

Given the adequacy of available data (Step 3.2) and the cost of obtaining better information (Step 3.3), the resource specialist and technical analyst can select a suite of indicators that best balances the level of accuracy needed to satisfy management objectives (Step 1.3) within existing constraints (Step 1.4). These choices are an optimal solution, given the existing opportunities and constraints.

Step 3.5 – Describe Indicator Assumptions

Once indicators have been selected, the resource specialist and the technical analyst should carefully determine which assumptions must hold if the indicator is to represent the synoptic index adequately (in this case, "adequately" is defined relative to the need for accuracy, as stated in Step 1.3). It is important for these assumptions to be stated explicitly, so they can be revisited later in the assessment to determine whether the assumptions were violated (Step 4.5). This information will also be included as part of the assessment documentation (Step 5.2). Examples of assumptions that can affect the outcome of an analysis are:

- The USGS regression estimates for peak discharge are often developed using data from watersheds that are not heavily urbanized, channelized, or dammed (e.g., Landers and Wilson 1991); in other words, these regressions are meant to represent "pristine" conditions. Use of regressions developed in this manner would include the implicit assumption that none of the watersheds has undergone significant hydrologic modification.
- Use of area as an indicator for wetland function assumes that function or capacity per unit area is similar for all wetlands or, if it varies, that wetlands having different unit area responses are similarly distributed between landscape subunits. The use of area as an indicator of a sink function further assumes that all wetlands receive import from a source or, if not, that the spatial relationship between wetlands and sources is similar between landscape subunits.
- The use of hydric soil area as an indicator of historical wetland area assumes that (a) wetland soil retains its hydric characteristics after drainage or conversion, (b) hydric soils are properly mapped, and (c) more permanently flooded wetlands, which could appear on SCS maps as water and not hydric soils, are either insignificant in an area or are distributed in such a way that bias is uniform across all subunits.

Step 3.6 – Finalize Subunit Selection

After selecting the final indicators, the resource analyst should reconsider subunits in light of the type of data available. For example, at first the analyst may select watersheds for subunits in Step 2.7 but later find that most data were based on county units. The analyst must then decide whether to prorate the county data to watershed units (see Appendix F) or to use counties as landscape subunits. This will depend on overall project goals and on whether the assumptions necessary for prorating hold true.

Step 3.7 – Conduct Pre-Analysis Review

Before conducting the assessment, the analyst should ask management and technical experts to review the overall management objectives, the synoptic indices that were defined, and the selected landscape indicators. The experts should, in particular, consider the appropriateness of the indicators with respect to objectives and constraints, and also review indicator assumptions for any evidence of violations. If violations are found, data may need to be adjusted or discarded, and alternate indicators considered.

Step 4: Conduct Assessment

Once landscape indicators have been defined and assumptions have been explicitly identified, maps and data can be obtained from the appropriate sources. The technical analyst can begin the process of producing the synoptic maps.

Step 4.1 – Plan Quality Assurance/Quality Control

Data for a synoptic assessment typically come from multiple sources (e.g., state and federal agencies, universities, and non-profit organizations) and come in a variety of formats, including mapped data, tabular data from reports, and computerized databases. Because reliability of the final product depends on quality control of data processing, a set of protocols should be developed for determining and maintaining data quality. The technical analyst should begin this step even before data are received, using information obtained during the data survey phase (Step 3.1).

Protocols should be developed for designing the database and for screening, archiving, and documenting the data. For example, protocols developed for data screening should identify questionable data based on an understanding of expected values and obvious outliers: A value of 100 centimeters per year for average precipitation would be questionable for a state in the arid southwest, and a peak discharge of only 100 cubic meters per second would obviously be too low for a major river. Percentages should add up to 100, and areas for component land uses should add up to total area. Protocols should also be developed for any variables to be measured, e.g., map measurements, and should include criteria for assessing accuracy, precision, completeness, representativeness, and comparability (EPA 1989).

In addition to the initial information collected during the data survey (Step 3.1), data documentation should include descriptions of the protocols, database design, and archiving formats. This information should be included as part of the assessment documentation (Step 5.2).

Step 4.2 – Perform Map Measurements

Much of the information used in a synoptic assessment is derived from maps. Examples of information and sources include: wetland area and number of wetland types from NWI maps, hydric soil area from county soil surveys, elevations and stream channel lengths from USGS topographic maps, and non-wetland land use from USGS Land Use/Land Cover (LULC) maps.

Two types of measurements are often made from maps: area and length. If the map is in digital format, a GIS can be used to generate these measurements. If a GIS is not available, the features can be planimetered or estimated using a dot grid. These three techniques are discussed in Appendix G.

If data reported for one type of spatial unit are to be prorated to another type of unit, joint areas must be calculated to serve as weighting factors. For example, if population data reported by county need to be adjusted to watershed subunits, the percent of the county lying in a particular watershed must be determined from an overlay of the two different areas (see Appendix F).

Error or bias can be introduced in map measurement through inadequate technician training, differences in accuracy between analysts, and defects or improper calibration of equipment. If maps are digitized for analysis in a GIS, compare hard copies of the digitized maps to the originals for accuracy. Also perform a quality control check for all map measurements by having a different analyst repeat 5% to 10% of the measurements to establish an error level. A discrepancy of more than 5% between analysts might be considered unacceptable. If the target is not met, a more comprehensive check is necessary.

The technical analyst must keep in mind the difference between *accuracy of map measurement* and *overall map accuracy*. A map can be measured very accurately, but still have unacceptable overall accuracy if the map itself contains errors. For example, a map produced through photo-interpretation of aerial photography might contain significant classification errors if the photo-interpreter is inexperienced. A good discussion of data quality and errors in mapping is found in Burrough (1986).

Step 4.3 – Analyze Data

A number of calculations could be required to produce an index value for each landscape subunit from the various data sources. Common analyses might include:

- *Calculating Channel Slope* – USGS discharge regressions often include channel slope as a variable. This slope is defined as the difference between the elevation of points located at 85% and 10% of the mainstream channel length. This difference is divided by the channel distance between the two points, i.e., 75% of the channel length (Appendix H).
- *Prorating Areas* – As discussed in Step 4.3, data must be prorated if an indicator is to be calculated for one type of unit based on data reported for a different type of subunit. Many types of data are typically reported by county, e.g., population statistics, agricultural data, soil characteristics data, and endangered species statistics; if the synoptic subunits are not counties, these data must be prorated using the weightings generated in Step 4.2.
- *305b Water Quality Summaries* – Under Section 305b of the Clean Water Act, states are required to report the extent to which their waters are meeting water quality standards. These 305b reports list, by stream segment or type of water body, whether a sampled segment fully supports, partially supports, or does not support (non-supporting) the "designated use" of that segment (for example, a stream can be designated as swimmable or fishable). If the segment is not fully supporting, the report lists the category of pollutant impacting the waters, e.g., point or nonpoint. The percentage of assessed streams that fully support state designated uses could be employed as an indicator of overall water quality. To produce such an indicator, the stream segments within each subunit must be identified and the relevant data summarized for that subunit. Note that the quality of state 305b reports varies by state. The analyst should also be aware of how the data were collected.

Final index estimates are produced by completing any other necessary calculations and converting to standard units, e.g., from English to metric. However, caution must be exercised when using regression equations. For example, the USGS regression equations for Mississippi (Landers and Wilson 1991) estimate peak discharge in ft^3/sec, using area (mi^2), channel length (mi), and slope (ft/mi); using metric units for area, channel length, and slope would be incorrect, since the regression equation was based on those English units. If metric units were desired, discharge should first be calculated in ft^3/sec using the English units, and then converted to m^3/sec. This indicator of hydrologic input could then be combined with an indicator of capacity to produce an estimate of hydrologic function. Additional examples of index estimation are provided in the case studies (Chapter 4).

After index values are calculated for each subunit, the subunits can be ranked by numerical values. For example, in an assessment of 50 subunits, the subunit with the highest value could be given a rank of 1 for that index, and the subunit with the lowest value given a rank of 50. Statistical packages such as SAS® (SAS Institute, Inc. 1988) can perform these calculations automatically. Rankings for each index should be included as part of the database.

The last step in analyzing the data is to perform a complete data quality check on the final database. For any calculations performed by computer, the analyst should recalculate a sample by hand to assure that the algorithms were programmed properly and that the output is accurate.

Step 4.4 – Produce Maps

The final synoptic maps can be produced by a computer mapping package, such as a GIS, or manually if resources are extremely limited or if no automated system is available. A GIS is recommended because it offers easy storage and manipulation of data and allows interim products to be used in later analyses. A GIS also gives the technical analyst greater flexibility to experiment with different display formats.

If a GIS is used, two different databases are typically required: one of the digital boundaries of the study area and its subunits and one of the index values that will be assigned to the subunits. Boundaries for all U.S. states, counties, and USGS accounting units have been digitized and are available at low cost in various formats (see LULC entry, Appendix E). If digital boundary data are not available, hand digitization may be necessary. This could be cost prohibitive if the study area includes a large number of highly detailed polygons, but the benefits of producing computer-generated maps often outweigh the digitizing costs. In some instances, sufficient accuracy may be achieved at even lower cost by using electronic scanners that digitize maps automatically.

The index values and rankings for each subunit must also be entered into the GIS. The method of accomplishing this and the amount of effort required will depend on the particular database-GIS combination. Many GIS packages provide routines for loading information from commonly used commercial databases.

Once the data are in the GIS, map production can begin. We recommend that the technical analyst produce component maps for each index if the index represents a combination of data sources. For example, if the

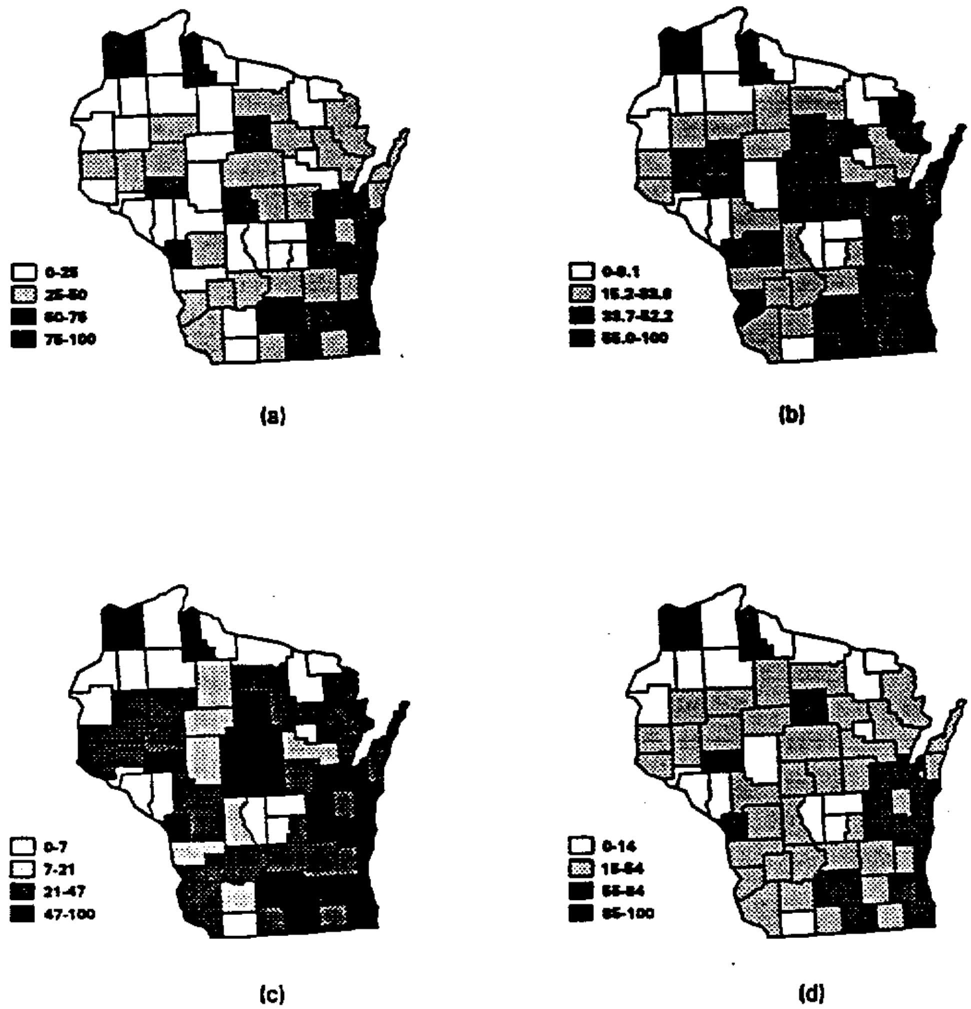


Figure 3.1. Illustration of maps using different class intervals to represent the same data: (a) equal intervals based on the data range; (b) intervals based on quartiles; (c) intervals increasing at constant rate; and (d) intervals based on the frequency distribution (adapted from Robinson et al. 1984).

USGS regressions are being used in Mississippi for peak discharge, then component maps of area, channel length, and slope should also be produced. This would allow the technical analyst and resource specialist to examine the data and determine whether the resulting spatial relationships are reasonable.

One of the most important decisions in the map production phase is how to display the data. At a minimum, the map should include the index value for each subunit. However, to promote interpretation, the data are typically aggregated into classes, or intervals. Ideally, class boundaries should reflect actual thresholds of function or value, e.g., patch sizes below which wildlife use drops precipitously or stream size above which local urban flooding is known to occur. Because such technically specific information is often unavailable, common alternatives are to divide the range of numeric values into equal intervals, or assign an equal number of subunits to each interval based on rankings (e.g., quartiles). The visual appearance of a given set of results can vary greatly, depending on how intervals are selected (Figure 3.1). The choice of class intervals is one of the more important decisions in the entire process because the synoptic maps will be the assessment's most visible outcome. People can easily reach erroneous conclusions if the map they are examining contains improperly displayed data. Perhaps the best way to design the intervals for map display is to first create a histogram or frequency curve showing the distribution of the numerical data (Figure 3.2). This will allow the analyst to detect any natural clumpings and also reveal

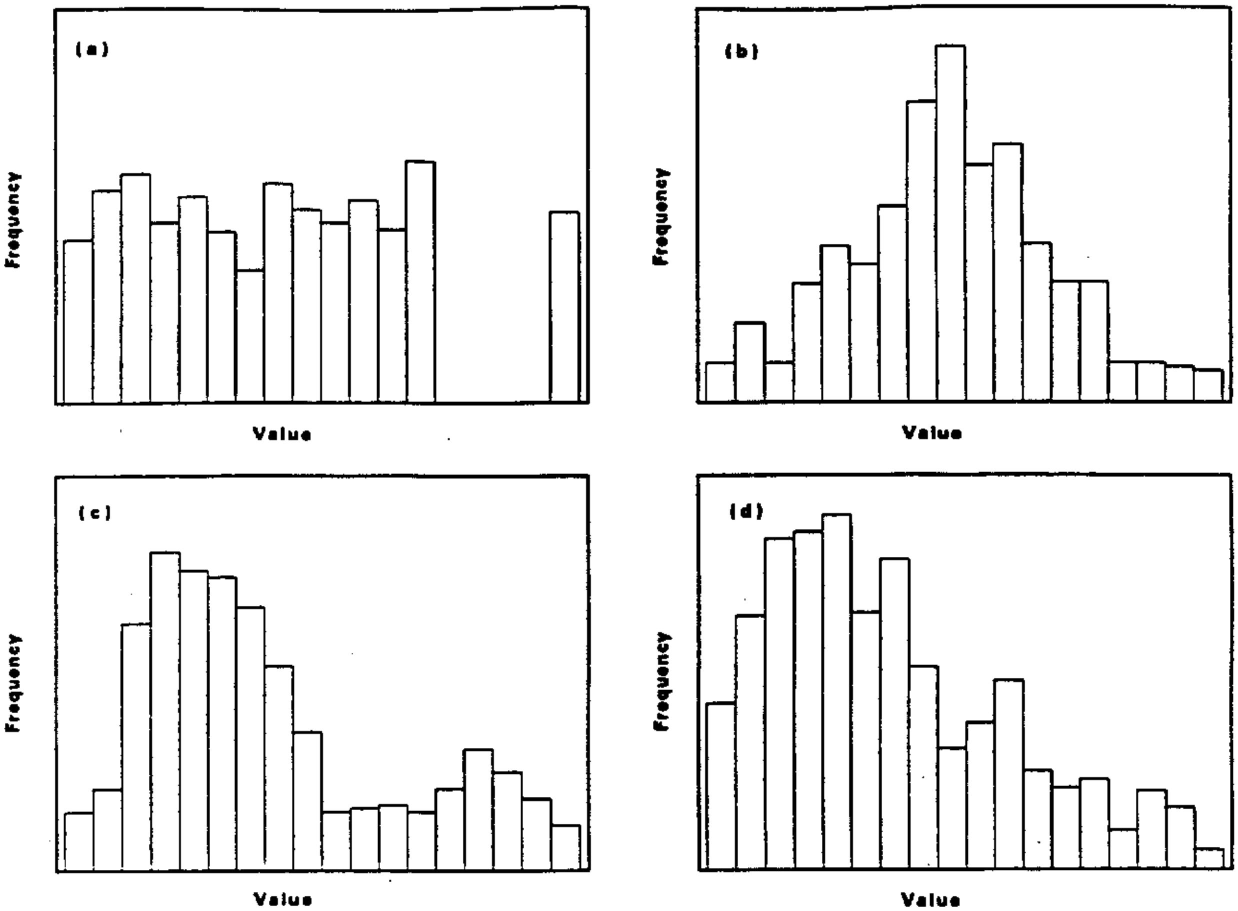


Figure 3.2. Different possible data distributions: (a) uniform with outlier, (b) normal, (c) bimodal, and (d) negative binomial.

common patterns such as normal or logarithmic distributions. Many standard texts on cartography, such as Robinson et al. (1984), include discussions on display of mapped data.

Once the appropriate intervals have been selected, the technical analyst considers options for displaying the range of values, e.g., color, shading, or hatching. Color, although more expensive, gives the greatest contrast and flexibility and should be considered if slide presentations will be made. Document production is less expensive if gray shadings are used; however, the analyst should select shades that provide enough contrast to be distinguished after photocopying.

Step 4.5 – Assess Accuracy

Throughout the course of the assessment, the technical analyst and resource specialist should look for evidence that any of the assumptions stated in Step 3.5 have been violated and consider the effects this would have on the assessment's accuracy. If the assumptions were violated for some units, it might be possible to adjust the index values. For example:

- Selection of an indicator for peak discharge could have been based on the assumption that subunits were not significantly regulated by dams. If a subunit is found to have a large dam or other major regulation, peak discharge would be significantly lower than the discharge that would occur naturally. The index value for that subunit could then be reassigned to the lowest category.
- To calculate wetland loss, the indicator for current wetland area could have been derived from USGS LULC maps if digital NWI wetland maps were not available. In cross-checking the classification, the analyst might have found that some areas classified as seasonally flooded riverine wetlands by NWI are classified on the LULC maps as deciduous forest, i.e., non-wetland. This underestimate of wetland area would cause an overestimate of historic wetland loss. These data may be adequate for relative comparisons of wetland loss if the proportion of deciduous forest

is similar in all subunits. Even if some subunits are much more dominated by deciduous forest than others, the analyst might be able to derive a correction factor to adjust the subunits, based on the percent of riparian land cover.

If the indices cannot be adjusted in such a fashion, the analyst may need to discard the data for the landscape subunits in which violations occurred. In some cases, the analyst might determine that the indicator is unsuitable for the required level of accuracy.

Throughout the entire assessment process, the technical analyst must consider the quality and accuracy of data sources to determine the overall quality of the final products. Unfortunately, no formal process for weighing the various factors exists. Ultimately, the technical analyst and resource specialist must use their own judgment and familiarity with the data to determine whether the synoptic results meet the stated needs (Step 1.3).

Step 4.6 – Conduct Post-Analysis Review

The assessment team should again seek technical experts' review comments following completion of the data analysis and synthesis. This information will assist the team in deriving conclusions and suggesting ways the results can be used. Because there is no method for quantitatively assessing the accuracy of results, this step and the pre-analysis review (Step 3.7) are essential to assure that results are adequate for the intended use.

Step 5: Prepare Synoptic Reports

The last step in the assessment is to report how the information was derived and how it can be used. Two different documents are appropriate: a report for the manager and resource specialist (a user's guide) and a detailed reporting of procedures to serve as a record of the complete assessment process (assessment documentation). Draft versions of these documents could also be included as part of the post-analysis review (Step 4.6).

Step 5.1 – Prepare User's Guide

This report should focus on the results of the assessment and how the results can be used to meet the original objectives. It might include protocols and illustrations of how the synoptic maps can be used in 404 permit reviews and should include any important caveats and assumptions as well as the overall level of accuracy. In particular, the user's guide should make clear that final numeric values are relative rankings, and should be treated as such. For example, if a subunit is ranked lowest of six for habitat functions, this does not necessarily mean the subunit lacks habitat or that its habitat is insignificant. It means it has lower habitat function, relative to the other subunits. Similarly, a relatively high subunit ranking for wetland replacement potential does not necessarily mean all wetland losses in that subunit can be easily replaced.

The intended audience for this report includes resource specialists who are involved in decision-making or planning, as well as resource agencies, scientists, and the public.

Step 5.2 – Prepare Assessment Documentation

Each synoptic assessment should include, for internal use or distribution to interested parties, complete documentation of how the assessment was conducted, including the objectives, constraints, rationale for index definition and indicator selection, assumptions related to the indicators, and detailed descriptions of the procedures used in measuring and analyzing the data. Any problems encountered should also be described. The report should carefully document the sources and quality of the various data sets and describe where and how the data are archived. It also should include an overall assessment of data quality and recommendations on how the assessment could be improved in the future. This document is a detailed record of the assessment process, and could be valuable if procedures are forgotten, challenged (e.g., through litigation), or if the assessment is updated.

2/12/92

CUMULATIVE IMPACTS ASSESSMENT AND MANAGEMENT PLANNING: LESSONS LEARNED TO DATE

by

Samuel C. Williamson
U.S. Fish and Wildlife Service
National Ecology Research Center
4512 McMurray Avenue
Fort Collins, CO 80525-3400

ABSTRACT

Cumulative impacts assessment should be bound closely with management planning for an ecosystem of concern and should consist of scoping and analysis of impacts from the past to the present. Cumulative impacts management planning for an ecosystem of concern should consist of interpretation and direction of impacts of ongoing and near future actions. When dealing with many problems in a complex situation, the recommended cumulative impacts assessment course of action is to: emphasize scientific, cause-effect understanding and communication; stress measurable overall action toward progressive goals; use a generation-long, ecosystem-level, problem-solving and solution-achieving process; and ratify an interagency collaborative drive toward cumulative improvement of the situation.

Selection of a strategy for dealing with each priority cumulative impacts problem should based on the mitigation options of restoration, impact avoidance, or impact minimization. The major objectives of cumulative impacts assessment and management planning should be to: generate logical, scientific, and timely problem analyses; bring agencies together collaboratively to develop an overall management strategy, plan, and specific, measurable resource goals; and meld those results into comprehensive species and habitat maintenance and enhancement blueprints for the ecosystem of concern. Natural resource agencies can soon anticipate a shift from scrutinizing individual permits, licenses, and assessments within an ecosystem of concern to a new capability of providing ecosystem-level guidance. The public can expect an active increase in positive ecological impacts and reduction in negative impacts as a result of cumulative impacts assessment and management.

INTRODUCTION

The growing awareness of cumulative impacts is accompanied by some puzzlement as to how they should be addressed. There is general agreement, however, that cumulative impacts (also known as cumulative effects) are a serious ecological challenge, as typified by: a) significant deterioration of major ecosystems (estuaries, lakes, and rivers); b) fragmentation and loss of critical habitats (wetland complexes); and c) long-term population declines (anadromous fish and migratory waterfowl).

The foremost cumulative impacts concern of natural resource agencies has been the negative effects of multiple human actions (interacting with each other and with natural events) within a major and highly valued ecosystem. As an example of an ecosystem of concern, San Francisco Bay is considered the major estuary in the United States that has been most modified by human activity (Nichols et al. 1986). Of the original 140,000 ha of freshwater marsh and 80,000 ha of saltwater marsh, only 12,500 ha (6%) remain. Sediment attributed to hydraulic mining debris has been deposited in central San Francisco Bay to a depth of 25 cm. Of the historic flow of the river system, 40% has been removed annually for local consumption upstream and within the delta while another 24% has been exported annually for municipal and agricultural consumption. Maximum annual concentration of sulfate and nitrate in the San Joaquin River have increased threefold and fivefold since 1950. Approximately 100 invertebrate species have been introduced; nearly all macroinvertebrates on the inner shallows of the bay are introduced species. Only commercial fisheries for herring and anchovy still exist; the former commercial fisheries for salmon, sturgeon, introduced striped bass (*Morone saxatilis*), and Dungeness crab (*Cancer magister*) have halted (Nichols et al. 1986).

Forman and Godron (1986) described a progression of landscape ecology degradation (beginning with the most sensitive) : 1) relative species abundance changes; 2) sensitive species disappear and native species diversity decreases; 3) nonnative species colonize; 4) biomass and cover decrease; 5) production decreases; and 6) erosion increases. In San Francisco Bay, all of these changes have been observed. Progress has been made on water quality attributes such as dissolved oxygen and enteric bacteria concentrations. Unfortunately, the major changes in the estuary (sediment deposition, loss of wetlands habitat, population declines of many fishes, and introduction of exotic species) occurred decades ago and former high quality conditions have been forgotten. Nichols et al. (1986) concluded that further improvement in water quality alone is not likely to have a significant positive effect on these major changes.

For the past decade, the term "cumulative impact" has been used merely in conjunction with assessment (i.e., scoping and analysis). The process described in this document is intended to commit assessment to management planning needs (i.e., interpretation and direction) of total cumulative impacts in an affected ecosystem. Solitary cumulative impacts assessment may be a decreed, one-time assignment; cumulative impacts assessment in combination with management planning should be a proactive, long-term process. Assessing one cumulative impact (the incremental impact only, without the rest of the cumulative impacts to date in the affected ecosystem) has been relatively unsuccessful. Assessing cumulative actions (again without the rest of the cumulative impacts to date in the affected ecosystem) likewise has been relatively unsuccessful. Assessing cumulative impacts (the total impacts to date of past actions and natural events) can and has been accomplished with various levels of success. We have learned to recommend cumulative impacts assessment and management planning, because of its greater potential for achieving long-term goals.

DEFINITION

When determining the scope of an environmental impact study, regulations of the Council on Environmental Quality require Federal agencies to consider three types of actions and three types of impacts (40 Code of Federal Regulations, Section 1508, 1987). The three types of actions to be considered are: cumulative actions (when viewed with other proposed actions have *cumulatively significant impacts*), connected actions (closely related actions that may be triggering or interdependent), and similar actions (have similarities such as common timing or geography). The three impacts that should be discussed in the same impact statement are direct, indirect, and cumulative impacts.

This paper is specifically concerned with ecological cumulative impacts assessment (concentrating on impacts up to the present). The Council's regulations, first published in 1978, provide definitions that can be summarized as follows:

- <u>Cumulative impact</u> is the "impact on the environment" of the "incremental impact of the action".
- <u>Cumulative impacts</u> are the total of the incremental impacts of past actions and present actions on the environment.
- <u>Environment</u> means the "effects on natural resources and on the components, structures, and functioning of affected ecosystems".
- <u>Effects</u> are synonymous with impacts, and the total effect of an action may, on balance, be either beneficial or detrimental.

TRANSLATION

Most of the terminology of cumulative impacts assessment is relatively new and subject to various interpretations. In numerous workshops, the specific wording of definitions and distinctions has proven to be necessary, but not universally acceptable. In this paper, the following distinctions and definitions are used:

- Cumulative impacts are the <u>combined effects of all human actions and natural events on the ecological environment</u> (Salwasser and Samson 1985).
- Cumulative actions (plural) assessment is scoping and analysis of the <u>total impacts of multiple proposed actions on the affected ecosystem</u>, and is not the subject of this paper.
- Cumulative impact assessment is scoping and analysis of the <u>incremental impact of one past action on the affected ecosystem</u>, and also is not the subject of this paper.
- Cumulative impacts (plural) assessment is scoping and analysis of the <u>total impacts of all past actions and natural events on the affected ecosystem</u>.
- Cumulative impacts management planning is interpretation and direction of the <u>total impacts of present actions and multiple proposed actions</u> on the affected ecosystem.

TYPOLOGY

We have found that typologies of cumulative impacts create a good deal of research interest but, like definitions, the pursuit of a definitive typology may turn into a tangent from cumulative impacts assessment. The typology of cumulative impacts presented by the National Research Council's Committee on the Applications of Ecological Theory to Environmental Problems (1986) is adopted for this paper. That typology recognized the following cumulative impacts: time-crowded perturbations, space-crowded perturbations, synergisms, indirect effects, nibbling, threshold developments, and lag effects.

An ecosystem of concern is usually characterized by substantial reductions in populations and lower or discontinued harvest of several important fish and wildlife species, substantial declines in the quality and quantity of several critical habitats, several human actions are causing the declines, and the declines are probably irreparable in the near future without society's corrective actions. Multiple causes of these declines is one of the major difficulties with cumulative impacts assessment and management planning projects. Cumulative impacts assessment within an ecosystem of concern should first connect multiple ecological causes (due to human actions and interrelated natural events) to historic and current state of the affected ecosystem (represented by habitat components, structures, and functioning) and then to numerous observed effects on natural resources (particularly fish and wildlife).

CONVENTIONAL ENVIRONMENTAL IMPACT ASSESSMENT

A classical environmental impact assessment is motivated by a proposed project; it focuses on and describes many site-specific environmental effects due to the project (an individual development action or one interrelated set of development actions) (Truett et al. 1992). A cumulative impacts assessment is generally driven by resource declines or concern over possible declines; it should focus on an ecosystem of concern and provide an overview of major species and habitat problems and the causes of the problems. The Council on Environmental Quality directed that environmental impact assessment consider cumulative impacts. According to Granholm et al. (1987), numerous institutional difficulties have been found with the practice of including cumulative impacts assessment as part of the environmental impact assessment process including: a) determining appropriate timing, costs and level of effort; b) apportioning the cost and responsibility for the assessment and mitigation among participants; c) coordinating assessment of different types of projects that cross agency jurisdictions; d) selecting appropriate methods and development scenarios for a particular assessment; e) limited history of application of most of the appropriate methods in a regulatory context; and f) identifying specific roles for project proponents and other interested parties. Making ecological cumulative impacts assessment part of an environmental impact assessment has been difficult and ineffective; the best use for a cumulative impacts assessment has been in management planning for an ecosystem of concern.

Environmental impact assessment focuses on inventorying and analyzing individual project effects; cumulative impacts assessment for management planning should focus on understanding of the ecosystem involved and formulating management programs to solve ecological problems. Recognizing that no agency has the overall authority to regulate, design, or plan for all the aspects of cumulative impacts, a cumulative impacts assessment should look at a much larger geographic area than typically used for evaluating an individual development action. Cumulative impacts are a pervasive problem that requires a different way of doing business from just the review of individual Federal projects, permits, or licenses (Muir et al. 1990).

In environmental impact assessment, decreasing the negative effects of individual development actions (minimizing impacts to no net loss when possible) is a desirable near-term strategy; in cumulative impacts assessment, more can be accomplished through striving to increase the positive effects of total development actions (improving the ecosystem when the opportunity presents itself). The individual elements of cumulative impacts cannot be regulated well on a project basis, but overall impacts can be assessed and managed (Burns 1991). Cumulative impacts assessment, as described here, can lead to comprehensive ecosystem guidance with information feedback from implementation, monitoring, and evaluation.

WHY DO CUMULATIVE IMPACTS ASSESSMENT?

Cumulative impacts assessment is most appropriate when dealing with many ecological causes and effects. Cumulative impacts assessment and management planning should be used in the most difficult ecological situations that encompass cumulative causality (started by multiple human actions and natural events), cumulative system effects (followed by decline of multiple habitats), cumulative fish and wildlife population effects (resulting in declines of multiple species), and cumulative restoration (rejuvenated by multiple human actions).

To be truly effective, cumulative impacts assessment and management planning should investigate and decrease the ongoing negative effects of human actions, but should concentrate on exploring and obtaining a more positive overall impact. The ecological challenge of cumulative impacts assessment and management planning in the future is to identify what should be done in terms of ecological changes, rather than merely what should not be done.

DESIGN OF SUCCESSFUL ASSESSMENT

Advocates of traditional methods in educational and governmental institutions have created unfocused, time-consuming, misguided, and narrowly defined assessments (National Research Council of the United States 1986; Canadian Environmental Assessment Research Council and United States National Research Council 1986). The evolution of cumulative impacts assessment methods has been constrained by reluctance to accept responsibility for cumulative impacts assessment and management planning. Under these conditions, jurisdictional problems have overwhelmed the process and the broad spatial and temporal bounds necessary for managing cumulative impacts are not incorporated into the assessments. It is important for effective cumulative impacts assessment and management planning to emphasize not numerous small assessments or a single, final plan, but an ongoing, regional, long-term strategy and planning process (National Research Council of the United States 1986; Canadian Environmental Assessment Research Council and United States National Research Council 1986).

Numerous theoretical, analytical, and institutional impediments hinder cumulative impacts assessment (Dickert and Tuttle 1985; Meehan and Webber 1985). Gosselink et al. (1990) pointed out that regulatory agencies have difficulty in dealing with cumulative impacts because the environment in which impacts interact is complex, changes may not be measurable individually, site-specific reviews do not represent a large enough geographical area (i.e., entire watershed or river basin) and ignore the time line, and regulatory agencies find effective, concerted action difficult. The recommended ways to deal with these four difficulties are based on a dozen case studies, the results of which are presented in this paper:

- Emphasize scientific, cause-effect understanding and communication of the overall situation, each problem, and problem interactions.
- Stress measurable overall action toward progressive goals for each problem.
- Use a generation-long, ecosystem-level, problem-solving and solution-generating process.
- Ratify an interagency collaborative drive toward cumulative improvement of the overall situation.

COLLABORATIVE

One essential point in conducting a cumulative impacts assessment and management planning project is gaining early consensus among the concerned natural resource agencies and institutions, particularly on whether to conduct such an assessment and on a strategy for addressing the ecosystem of concern. Collaboration with other regulatory agencies is essential to a successful cumulative impacts assessment and management planning project because the responsibility for natural resources typically rests with many local, State, and Federal agencies.

At least one subject matter expert from each of the concerned natural resource agencies should be involved in the scoping and analysis phases of a cumulative impacts assessment and management planning project. Agency differences can be minimized and support gained from sharing information and understanding of technical issues. Management users from the concerned natural resource management agencies should be involved in the early design of the assessment and again later in the interpretation and direction phases. This creates a sense of ownership, commitment, and responsibility in the participants and their agency and promotes greater coordination, cooperation, and consensus among the natural resource agencies.

GOAL ORIENTED

Goal orientation forces a cumulative impacts assessment and management planning project to be purposeful and focused. Setting quantitative, measurable, time-dependent goals implies that society has deemed particular ecological resources or conditions as desirable, and that management agencies are committed to conserving, protecting, or enhancing those resources and conditions. For a cumulative impacts assessment and management planning project to be purposeful, it should be directed toward increasing some of the resources above current status, not just maintaining status quo or avoiding deterioration thresholds. In particular, deterioration threshold evasion (impact minimization) is not a desirable way to deal with cumulative impacts in an ecosystem of concern given the opportunity of stabilizing (impact avoidance) or managing upward (restoration).

The Canadian Environmental Assessment Research Council and United States National Research Council (1986) questioned whether cumulative effects could be managed without a comprehensive set of societal goals. To improve cumulative impacts assessments and make them useful in regulatory decisionmaking, explicit societal goals should be defined and made part of a comprehensive, future-oriented planning process (Horak et al. 1983; Stakhiv 1986; Canadian Environmental Assessment Research Council and United States National Research Council 1986). A key point of cumulative impacts management planning is that strategic policy decisions should be made and goals for resources of concern set in the assessment, before management planning takes place.

PROBLEM SOLVING PROCESS

Because each situation differs in complexity, amount of available and usable data, and degree of understanding of the problems and ecological processes, an extensive education with ecological problem solving is needed for agencies to gain the skill, knowledge, and technology required to successfully assess and manage cumulative impacts. It is important to start cumulative impacts assessment from the effects (species and habitat problems) side instead of the causes (development actions and natural events) side and take a problem-solving and solution-generating, total ecosystem view. The advantages of a problem-solving approach are that it encourages concentration of effort, a thorough search for an unbiased statement of the situation and specific problems, an incremental and sequential analysis, and identification and selection of realistic, feasible, and economical solutions. Ecological problem solving is a key element in a successful cumulative impacts assessment (Salwasser and Samson 1985; National Research Council of the United States 1986).

The complexity of many cumulative impacts problems corroborates the assertion that cumulative impacts assessment cannot be accomplished by a method or technique developed to apply in all cases. According to the Canadian Environmental Assessment Research Council (1988), cumulative impacts assessment underlines the need for a long-term, well-organized approach leading to resolution of the problem's scientific and institutional aspects. A successful cumulative impacts assessment should employ a problem solving process that can be applied intensively to a wide range of situations and utilizes adaptively the most appropriate methods and techniques.

SCIENTIFIC CAUSE AND EFFECT

Cumulative impacts assessment requires a high order of analysis and interpretation of cause-effect linkages; new concepts and alternative thinking processes to restructure the problem; new techniques to aggregate diverse impacts, and a holistic, integrative perspective (Horak et al. 1983). Granholm et al. (1987) concluded that new methods are needed to deal with the complexities of multidisciplinary systems and that available techniques such as group problem-solving, area assessment, and simulation modeling are either used ineffectively or not at all. Because cumulative impacts assessment, unlike traditional environmental impact assessment, is a form of pattern analysis and must detect and analyze trends, cumulative impacts assessment needs scientific understanding of cause and effect (Canadian Environmental Assessment Research Council and the United States National Research Council 1986).

Many cumulative impacts assessment efforts, no matter how potentially sound analytically, degenerate before they begin because of the lack of four prerequisites for successful management. To be effective in cumulative impacts assessment, use both a problem-solving process and scientific cause and effect; to be effective in cumulative impacts management planning, use both goal setting and collaboration. The major ecosystem-level success stories (e.g., Lake Washington, Lake Erie, Lake Michigan, Potomac River) have had those necessary ingredients for success.

RECOMMENDED PROCESS

The recommended cumulative impacts assessment and management planning process should follow the steps: 1) in the scoping phase, define the ecological situation in specific terms of individual problem statements and select one strategy for each problem; 2) in the analysis phase, investigate and document the problems and their causes in detail using the best available data and analytical tools and then set several goals; 3) in the interpretation phase, develop and document options, estimate changes using mathematical models, and develop a plan; and 4) in the direction phase, implement and incrementally improve the management plan and systematically evaluate, improve and update the problem statements, data, analytical tools, and mathematical models.

It has been useful to distinguish cumulative impacts assessment (Steps 1 and 2 above) as the portion of the time horizon from the past to the present and cumulative impacts management planning (Steps 3 and 4 above) as the portion of the time horizon from the present to the future. Step 1 focuses on qualitative problem descriptions and is intended to accomplish problem identification, clarification, and expression. Establishing appropriate temporal, spatial, and political boundaries is difficult, but critical to the success of a cumulative impacts assessment (Lee and Gosselink 1988). Concern about cumulative impacts by Federal natural resource regulatory agencies has been pronounced in areas that are moderately large and complex (entire ecosystems with a focus on aquatic and wetland habitat). Generally, a multiagency group of natural resource management experts should be gathered to work collaboratively in a workshop setting. The group identifies important ecological problems contributing to the overall situation, agrees on problem statements, and documents those problems using the relevant scientific literature. Careful statement of each problem goes a long way toward stimulating action on its solution.

Step 2 provides quantitative problem analyses and goal statements that are technically and scientifically credible. The status and historic trends of the priority resources are documented, graphed, and mapped. Based on an evaluation of the best data, literature, and scientific judgment available, early problem statements are accepted, modified, or rejected. The importance of causal factors is evaluated. Data gaps, research needs, and preferred predictive mathematical models are identified. Specific management goals are generated and supported, both scientifically and institutionally. For example, in an early restoration planning workshop for Commencement Bay, Washington the natural resource trust agencies developed the following problem and goal statement: "Virtually none (less than 1%) of the original 10 square km (2,470 acres) of subaerial wetlands in the Commencement Bay-lower Puyallup River ecosystem remain. By 2005, restore at least x-y acres (some numbers between 10% and 50%) of these wetlands in that ecosystem."

In Step 3, the focus is on defining management opportunities. The quantitative analyses from Step 2 should be used to identify the most important causal factors in each problem. Effective alternative actions that may achieve the goals are identified and evaluated. Determine which of the actions identified above are ecologically, politically, institutionally, economically, and legally feasible, and identify the mechanisms through which effective actions can be implemented. Each individual agency's responsibilities should be identified, and the ability of agencies to have a significant positive effect should be evaluated. Several alternative management plans should be evaluated with the mathematical model for achieving the resource goals. The recommended plan should contain the set of effective actions that optimally achieve the multiple goals for the priority resources.

At several points during the assessment process, subjective value judgments must be made with reference to some framework of social values. It has proven essential to deliberate collaboratively on the ramifications of each possible strategy and gain interagency consensus early in the scoping of the problem. Strategy selection should be based on the Council on Environmental Quality's five options for mitigation and depends on society's "acceptable standards" for ecological resources: a) where the current ecological condition is below acceptable standards, a restoration strategy is appropriate; b) where the current condition is about equal to acceptable standards, a strategy of impact avoidance (no net loss of habitat) is usually chosen; and c) where the current condition is above acceptable standards, a strategy of allowing some decline from current conditions by impact minimization will work. Impact minimization is generally the current strategy of the natural resource agencies concerned about cumulative impacts assessment. Just an agreement on the most desirable strategy for each problem is frequently a major advancement for the agencies involved.

RECENT HISTORY

Considering the number of articles being published, interest in cumulative impacts is increasing. The first two articles with the term were published in 1975. In the period from 1975 to 1980, the publication rate was between one and four papers per year. In the period 1981 to 1984, the publication rate was between 6 and 13 papers per year. In the period from 1985 to 1988, the publication rate was between 11 and 37 papers per year (Williamson and Hamilton 1989).

PROJECT BACKGROUND

A cumulative impacts assessment project was initiated by the U.S. Fish and Wildlife Service in 1984 at the Western Energy and Land Use Team (now the National Ecology Research Center). The project's systems analysis approach involved the stages of: 1) understanding Ecological Services' user needs (Williamson et al. 1986), 2) conducting real world analyses of cumulative impacts problems using prototype trials (e.g., Williamson et al. 1987), and 3) developing and refining an assessment process. Collaboration with other cumulative impacts assessment researchers was emphasized in interagency conferences to advance cumulative impacts assessment (see Williamson and Hamilton 1989).

To develop and improve a Fish and Wildlife Service approach, we undertook several cumulative impacts assessment case studies (by observing the work of other agencies) and prototype trials (by conducting them jointly with U.S. Fish and Wildlife Service's Ecological Services field offices across the country). Some of these are described below. The process developed through these stages has been characterized as a classical planning process for the purpose of ecological problem solving.

UNDERSTANDING BARGE NAVIGATION EFFECTS ON RIVERS

The field office in Cookeville, Tennessee used a scientific, cause-effect network diagram to prepare comments on barge traffic permit applications in the Ohio, Tennessee, and Cumberland Rivers. The diagram met the needs of the Corps of Engineers and the field office in determining principal resources of concern, problems, and causal pathways. The major contribution of this project was the successful use of cause-effect network analysis. The diagram did not provide a quantitative analysis, but it did provide a mechanism for understanding and communication between agencies about important factors and a framework for tracking potential effects of barge traffic. The Corps' office asked that future efforts also provide such a cause-effect network diagram. The cause-effect network diagram was later used by the Annapolis field office and the National Fisheries Center-Great Lakes to specifically describe several major omissions in a barge traffic simulation model prepared under contract for the Corps of Engineers.

DESIGNING BETTER OIL FIELD DEVELOPMENT IN ALASKA

The Fish and Wildlife Service's regional and field offices developed planning aid documents to minimize the cumulative impacts of oil and gas development on the wet tundra of the North Slope coastal plain of Alaska (Meehan and Webber 1985; Walker et al. 1987). The regional office applied the cumulative impacts assessment and management planning process to the Colville Delta oil field with the intention of extending the lessons learned at the Prudhoe Bay oil field. For example, in the wettest parts of the oil field, resultant flooding and thermokarst were found over more than twice the area covered by roads and other construction (Walker et al. 1987). Alaskan oil development provided an opportunity to study cumulative impacts on a well-defined terrestrial scale and in a relatively pristine habitat resource.

PLANNING RESTORATION OF CHESAPEAKE BAY

The Annapolis field office conducted a cumulative impacts assessment and management planning project in accord with the Chesapeake Bay Restoration Plan. The Environmental Protection Agency has the lead role in the bay restoration program and has emphasized restoration of water quality (e.g., nitrogen loading, dissolved oxygen concentration) in the bay. One of the documents (Flemer et al. 1983) prepared as part of the bay restoration program is so good that it can serve as a template for the report for a cumulative impacts assessment. In conducting a prototype cumulative impacts assessment, workshop participants defined problems, identified important cause-effect relationships, and developed preliminary remedial action plans (Williamson et al. 1987). As problems were examined through cause-effect network analysis, there was a clear movement away from problem statements focusing on development actions (near the start of causal chains) and fish and wildlife species (near the end of effect chains); when problem identification was based on habitats (the hub of causes and effects), the assessment focused clearly on ecological goal attainment and remedial action management planning. As a consequence of the assessment, a 70% decline in distribution of native submerged aquatic vegetation across the bay proper and increased amounts of suspended particulates in historically productive watersheds were identified as keystone problems to be dealt with. The Fish and Wildlife Service has chosen to pursue an emphasis on living resources (e.g., indigenous species of submerged aquatic vegetation) as opposed to an emphasis on water quality.

GUIDING GROWTH IN AN URBANIZED ESTUARY

The Daphne, Alabama field office conducted a Mobile Bay cumulative impacts assessment and management planning project. The assessment and planning project contains four major elements: 1) a cause-effect network analysis; 2) a status and trends analysis; 3) goal-setting for bay resources by the natural resource management agencies; and 4) development of a coordinated action agenda. In goal-setting work for eight problems, each of them had some combination of the following: current action goals (things that can be done immediately or that should continue), management-related information goals (things that need to be done to improve and maintain our understanding of the bay system), and 20-year attainment goals (what would we like the bay to look like in 20 years). The State of Alabama was concurrently conducting a cumulative impacts assessment for Mobile Bay. The field office's work (with the other natural resource agencies) was adopted and carried forward as the State's recommended approach and results for Mobile Bay. The advisory group for the State's project has come to the conclusion that project-by-project cumulative impact assessments (see earlier definition) are ineffectual and that an ecosystem-level cumulative impacts assessment (leading to comprehensive ecosystem-level guidance) is advantageous.

ANALYZING A GREAT LAKES CONNECTING WATERWAYS ECOSYSTEM

The ecosystem of concern is the connecting waterways between Lake Huron and Lake Erie consisting of Lake St. Clair and the St. Clair and Detroit Rivers. In a multi-agency (Federal, Michigan, Ontario, and Great Lakes Fishery Commission) workshop sponsored by the Fish and Wildlife Service's National Fisheries Center-Great Lakes and the East Lansing, Michigan field office, cause-effect network diagrams were constructed for four ecological problems in each of the waterways. Using a mathematical matrix method, relative importance values for cause-effect relationships were assigned by subject matter experts at the workshop. With institutional mechanisms for making decisions and implementing actions already in place (National Research Council of the United States and The Royal Society of Canada 1985), this cumulative impacts assessment and management planning project emphasized technical enhancement of assessment methods, particularly matrices and simulation modeling.

CLUSTER IMPACT ASSESSMENT PROCEDURE

In 1985, the Federal Energy Regulatory Commission contracted with Argonne National Laboratory to develop the Cluster Impact Assessment Procedure to evaluate the potential cumulative impacts of multiple small scale hydroelectric projects (Bain et al. 1986). The Cluster Impact Assessment Procedure was used to identify geographic areas of concern in the Snohomish River (Washington) and Salmon River (Idaho) watersheds, determine projects that could have adverse effects on target resources, and conduct a multiple project (cluster) impact analysis. The cumulative actions assessment (see earlier definition) was intractable for cumulative impacts assessment (U.S. General Accounting Office 1988). The natural resource agencies found the mathematical matrix approach unduly complex and biologically unacceptable. In both river basins, the Commission issued preliminary permits without requiring site-specific information for assessment of cumulative impacts or a comprehensive plan (Feldman 1988). The U.S. General Accounting Office (1988) found that preparation of a comprehensive river basin plan could have been a major help in resolving disagreements between the Federal Energy Regulatory Commission and the natural resource agencies over the way to carry out a cumulative impacts assessment.

AVOIDING IMPACTS ON SALMON HABITAT

The Snohomish Guidelines for the Evaluation of Hydropower Projects were designed to avoid further loss of salmonid habitats and populations due to 600 proposed (mostly small-scale) hydroelectric development projects in Washington's Snohomish River basin. This cumulative impacts management planning project (see earlier definition) addressed each project through individual project siting, design, operating, and timing specifications for the project applications. The Snohomish guidelines (Stout 1988) were used in a situation where no further deterioration or loss of habitat has been accepted (and legislated) as society's intention. The Guidelines were developed by a coalition of concerned Indian tribes, Washington State agencies, and Federal natural resource agencies.

BOTTOMLAND HARDWOODS

The Fish and Wildlife Service conducted three workshops for the Environmental Protection Agency on the ecological impacts of bottomland hardwood deforestation. Cumulative impacts analysis at large watershed or river basin levels became an important part of the project. The participants came to the conclusions that ecological goals are essential, that goals frequently are not available at scales that match cumulative impacts problems, and that a means of nonincremental analysis is needed for cumulative impacts assessment (Gosselink et al. 1990). Setting goals other than no change or no further loss was difficult, because it involved society's preferences as well as practical aspects of regulation. Gosselink and Lee (1989) described a landscape ecology approach and its use in a cumulative impacts assessment that involves habitat fragmentation and loss. They hypothesized that individual features are not as important as the pattern, and it is the key features of the pattern which must be identified to conserve biotic diversity and the broad functional values associated with these ecosystems.

FUTURE NEEDS

The general problem facing the U.S. Fish and Wildlife Service's field offices in trying to assess cumulative impacts in 1985 was that cumulative impacts assessments were not happening (Williamson et al. 1986). That problem could best be redressed by convincing the responsible entities, both inside and outside the Fish and Wildlife Service, to conduct those assessments. In addition, a process and methods were needed that provide the technical capability to conduct a cumulative impacts assessment based on the resources, not just regulations. Two causes of the identified problems were technical shortcomings of individual assessment tools and institutional hesitancy to try something new.

From a tentative schedule for the ten highest priority actions for addressing cumulative impacts (Williamson et al. 1986), the first six actions have been effected with various levels of effort and success. The other four actions have the potential of moving Federal agencies from attempted cumulative impact assessment to the more productive area of cumulative impacts assessment and management planning. In their order of importance, the suggested actions are: a) review and enlarge agency goals and policies for habitats and species; b) conduct cumulative impacts assessments for fish, wildlife, and habitat resources of national and regional concern; c) develop a Federal interagency council to foster better cumulative impacts assessments; and d) make ecological monitoring and project follow-up functions a natural resource agency responsibility.

Institutional reluctance to conduct cumulative impacts assessment and management planning projects has decreased considerably but needs to be reduced further (Muir et al. 1990). An interagency analysis of the differences between an incremental impact viewpoint (i.e., environmental impact assessment) and a total impacts viewpoint (i.e., cumulative impacts assessment) would help. Undertaking sound environmental impact assessment without the regional context and cumulative changes is difficult (Canadian Environmental Assessment Research Council 1988; U.S. General Accounting Office 1988). Federal agencies could improve individual analyses of permits and licenses by providing a framework within which to evaluate them; this can be done by highlighting ecosystem-based, collaborative cumulative impacts assessment and management planning (Stakhiv 1988; U.S. General Accounting Office 1988; Gosselink et al. 1990).

An example of such an approach is the Environmental Protection Agency's National Estuary Program. The program employs collaborative problem solving to balance conflicting uses while restoring or maintaining the estuary's environmental quality. Because of their early entry into cumulative impacts work, the Chesapeake Bay and Great Lakes programs are frequently used as models for cumulative impacts assessment and management planning. Using knowledge gained from the Great Lakes and Chesapeake Bay as its foundations, the participating natural resource agencies have learned how to get the desired results in less time and with less money. The National Estuary Program stresses focusing on the most significant problems, using existing data, emphasizing applied research, funding specifically targeted basic research, and employing demonstrated management strategies. Nichols et al. (1986) observed that the future well-being of the urbanized estuaries depends on achieving an increased understanding of each one's physical, chemical and biological processes and how specific human activities affect those processes; meanwhile, economically important actions are considered without sufficient quantitative understanding of an action's effects on the estuary.

Through coalition and commitment of the responsible natural resource agencies, those agencies can jointly do a more effective and efficient job of comprehensive ecological planning and management. So far in cumulative impacts assessment and management planning projects, the responsible natural resource agencies have included State departments of fish and wildlife, State and local departments of natural resources, ecological research institutes, Indian tribal councils, the Corps of Engineers, Environmental Protection Agency, Fish and Wildlife Service, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, and Canadian federal and provincial natural resource agencies. Cumulative impacts assessment and management planning can shed light on these agencies' unified activities that will attract funding for ecological programs.

The real contribution of mitigation and reclamation actions to achieving society's ecological goals could be improved by using cumulative impacts assessment and management. When the natural resource conservation, regulatory, and land management agencies ratify an interagency collaborative drive toward cumulative improvement of the overall situation, they should be able to move toward several management goals simultaneously. With cumulative impacts assessment and management planning, the Fish and Wildlife Service has been able to promote such positive aspects of management, mitigation and reclamation and avoid negative, adversarial, and confrontational situations.

We have progressed considerably from the baseline study approach that emerged shortly after passage of the National Environmental Policy Act as the primary response of ecologists to multiple-species concerns and as the major supplier of information for the environmental impact assessment process (Truett et al. 1992). Twenty years later, we have learned that you cannot effectively regulate individual minor contributions to cumulative impacts but that you can plan for them in the aggregate. It is no longer a question whether we should conduct cumulative impacts assessment and management planning projects. We should! The question is also not how can we best conduct such a project. That depends on the situation, as described here. The question now is "How can we acquire the support for conducting cumulative impact assessment and management planning projects?" Instead of continuing to rely on the ability of American citizens and institutions to respond to individual ecological crises as they are recognized and popularized, a technological capability for coordinated, effective action through cumulative impacts assessment and management planning should be developed before ecological problems reach crisis proportions.

ACKNOWLEDGMENTS

Special thanks are extended to all of the U.S. Fish and Wildlife Service personnel who participated in developing the information presented in this document.

LITERATURE CITED

Bain, M.B., J.S. Irving, R.D. Olsen, E.A. Stull, and G.W. Witmer. 1986. Cumulative impact assessment: evaluating the environmental effects of multiple human developments. Argonne Nat. Lab. ANL/EES-TM-309. Argonne, IL. 71 pp.

Burns, D.C. 1991. Cumulative effects of small modifications to habitat. Fisheries 16:12-17.

Canadian Environmental Assessment Research Council. 1988. The assessment of cumulative effects: a research prospectus. Minister of Supply and Services Canada. Cat. No. En 106-10/88. 11 pp.

Canadian Environmental Assessment Research Council and United States National Research Council. 1986. Cumulative environmental effects: a binational perspective. Minister of Supply and Services Canada. Cat. No. En 106-2/85. 175 pp.

Dickert, T.G., and A.E. Tuttle. 1985. Cumulative impact assessment in environmental planning: a coastal wetland watershed example. Environ. Impact Assess. Rev. 5:37-64.

Feldman, M.D. 1988. National Wildlife Federation v. FERC and Washington State Department of Fisheries v. FERC: Federal Energy Regulatory Commission ignores Ninth Circuit rebuke on hydropower permitting. Ecology Law Quarterly 15:3-360.

Flemer, D.A., G.B. Mackiernan, W. Nehlsen, V.K. Tippie, R.B. Biggs, D. Blaylock, N.H. Burger, L.C. Davidson, D. Haberman, K.S. Price, and J.L. Taft. 1983. Chesapeake Bay: A profile of environmental change. U.S. Environmental Protection Agency, Philadelphia, PA. 200 pp.

Forman, R.T.T., and M. Godron. 1986. Landscape ecology. John Wiley & Sons, York, N.Y. 196 pp.

Gosselink, J.G., and L.C. Lee. 1989. Cumulative impact assessment in bottomland hardwood forests. Wetlands 9:89-174.
DELETE Gosselink, J.G., and L.C. Lee. 1987.

Gosselink, J.G., L.C. Lee, and T.A. Muir, eds. 1990. Ecological processes and cumulative impacts: illustrated by bottomland hardwood wetland ecosystems. Lewis Publishers, Inc. 708 pp.
DELETE Gosselink, J.G., and L.C. Lee. 1988.

Granholm, S.L., E. Gerstler, R.R. Everitt, D.P. Bernard, and E.C. Vlachos. 1987. Issues, methods, and institutional processes for assessing cumulative biological impacts. Prepared for Pacific Gas and Electric Company, San Ramon, CA. v.p.

Horak, G.C., E.C. Vlachos, and E.W. Cline. 1983. Methodological guidance for assessing cumulative impacts on fish and wildlife. Prepared for U.S. Fish and Wildlife Service, Eastern Energy and Land Use Team, Kearneysville, WV. Contract No. 14-16-0009-81-058. 99 pp.

Lee, L.C., and J.G. Gosselink. 1988. Cumulative impacts on wetlands: linking scientific assessments and regulatory alternatives. Environ. Manage. 12:591-602.

Meehan, R., and P.J. Webber. 1985. Towards an understanding and assessment of the cumulative impacts of Alaskan North Slope oil and gas development. U.S. Environmental Protection Agency under U.S. Department of Energy Interagency Agreement No. DE-A106-84RL10584.

Muir, T.A., C. Rhodes, and J.G. Gosselink. 1990. Federal statutes and programs relating to cumulative impacts in wetlands. Pages 223-236 in Gosselink, J.G., L.C. Lee, and T.A. Muir, eds. Ecological processes and cumulative impacts. Lewis Publishers, Inc.

National Research Council of the United States, Committee on the Applications of Ecological Theory to Environmental Problems. 1986. Ecological knowledge and environmental problem-solving: concepts and case studies. National Academy Press, Washington, DC. 388 pp.

National Research Council of the United States and The Royal Society of Canada. 1985. The Great Lakes Water Quality Agreement: an evolving instrument for ecosystem management. National Academy Press, Washington, D.C. 224 pp.

Nichols, F.H., J.E. Cloern, S.N. Luoma, and D.H. Peterson. 1986. The modification of an estuary. Science 231:567-573.

DELETE Roelle, J.E., G.T. Auble, D.B. Hamilton, R.L. Johnson, and C.A. Segelquist, eds. 1987.

Salwasser, H., and F.B. Samson. 1985. Cumulative effects analysis: an advance in wildlife planning and management. Trans. N. Amer. Wildl. Nat. Resour. Conf. 50:313-321.

Stakhiv, E.Z. 1986. Cumulative impact analysis for regulating decisionmaking. Pages 213-222 in J.A. Kusler and P. Riexinger, eds. Proceedings of the national wetland assessment symposium. ASWM Tech. Rep. 1. Association of State Wetland Managers, Chester, VT.

Stakhiv, E.Z. 1988. An evaluation paradigm for cumulative impact analysis. Environ. Manage. 12:725-748.

Stout, D.J. 1988. Preventing cumulative impacts: the Washington experience. Pages 204-206 in J.A. Kusler, M.L. Quammen, and G. Brooks, eds. Proceedings of the national wetland symposium: mitigation of impacts and losses. Association of State Wetland Managers, Tech. Rep. 3.

Truett, J.C., H.L. Short, and S.C. Williamson. 1992 (in press). Ecological impact assessment. Pages ___-___ in Wildlife Management Techniques. T. Bookhout, ed. The Wildlife Society, Inc., Washington, D.C. 5th edition.

U.S. General Accounting Office. 1988. Energy regulation: opportunities for strengthening hydropower cumulative impact assessments. Report to the Chairman, Committee on Energy and Commerce, House of Representatives, GAO-RCED-88-82. 20 pp.

Walker, D.A., P.J. Webber, E.F. Binnian, K.R. Everitt, N.D. Lederer, E.A. Nordstrand, and M.D. Walker. 1987. Cumulative impacts of oil fields on northern Alaskan landscapes. Science 238:757-761.

Williamson, S.C., and K. Hamilton. 1989. Annotated bibliography of ecological cumulative impacts assessment. U.S. Fish and Wildlife Serv. Biol. Rep. 89(11). 80 pp.

Williamson, S.C., C.L. Armour, and R.L. Johnson. 1986. Preparing a FWS cumulative impacts program: January 1985 workshop proceedings. U.S. Fish Wildl. Serv. Biol. Rep. 85(11.2). 37 pp.

Williamson, S.C., C.L. Armour, G.W. Kinser, S.L. Funderburk, and T.N. Hall. 1987. Cumulative impacts assessment: an application to Chesapeake Bay. Trans. N. Amer. Wildl. Nat. Resour. Conf. 52:377-388.

Cumulative Impacts Assessment: An Application to Chesapeake Bay

Samuel C. Williamson and **Carl L. Armour**
U.S. Fish and Wildlife Service
Ft. Collins, Colorado

Glenn W. Kinser, Steven L. Funderburk and **Timothy N. Hall**
U.S. Fish and Wildlife Service
Annapolis, Maryland

Introduction

This paper describes a cumulative impacts assessment process and the results to date from its prototype application to Chesapeake Bay. Cumulative impacts, as used here, are the accumulation of all effects of human actions and natural events on the ecological environment (Salwasser and Samson 1985). Odum (1982) called this situation "the tyranny of small decisions" because no decision was ever consciously made to allow cumulative impacts. Causes of eventual cumulative impacts problems are usually separated in space or time and frequently differ in degree; therefore, the reduction in environmental quality is gradual and often goes unnoticed (Williamson et al. 1986).

The Chesapeake Bay ecosystem (Figure 1) has been subjected to numerous alterations, which often interact in complex and poorly understood ways. Human actions in relation to housing, industry, agriculture, transportation and navigation have resulted in disposal and dispersal of solid wastes, heavy metals, petroleum hydrocarbons, biocides, synthetic organics, heated water, nutrients, and acidic atmospheric emissions (Capper et al. 1983, Flemer et al. 1983). The water chemistry and physics of the Bay have deteriorated, as indicated by increases in biochemical oxygen demand, water temperature, suspended sediment load, sediment deposition and turbidity, by reduction in stream flows and dissolved oxygen, and by modifications of salinity and alkalinity. The biological composition of the Chesapeake Bay ecosystem has been altered, as indicated by changes in algae and submerged plant species (more nuisance species), decreases of most species of endemic submerged aquatic vegetation, and increases of epiphytes on endemic submerged plant species and predators on sedentary shellfish. Populations of fish and wildlife species—such as the American oyster (*Crassostrea virginica*), striped bass (*Morone saxatilis*) and canvasback duck (*Aythya valisineria*)—have declined substantially (Lippson 1985).

The U.S. Fish and Wildlife Service (USFWS) has a legal responsibility for maintenance and enhancement of biological resources in the Bay. The USFWS can assist in restoring Bay water quality, biological productivity, and fish and wildlife populations to their former levels by: (1) participating in the current revision of Section 208 watershed management plans through the Federal Clean Water Act; (2) reviewing federally permitted activities within the Bay's watersheds, as mandated by the Fish and Wildlife Coordination Act; and (3) reviewing environmental impact statements for major federal actions as mandated by the National Environmental Policy Act.

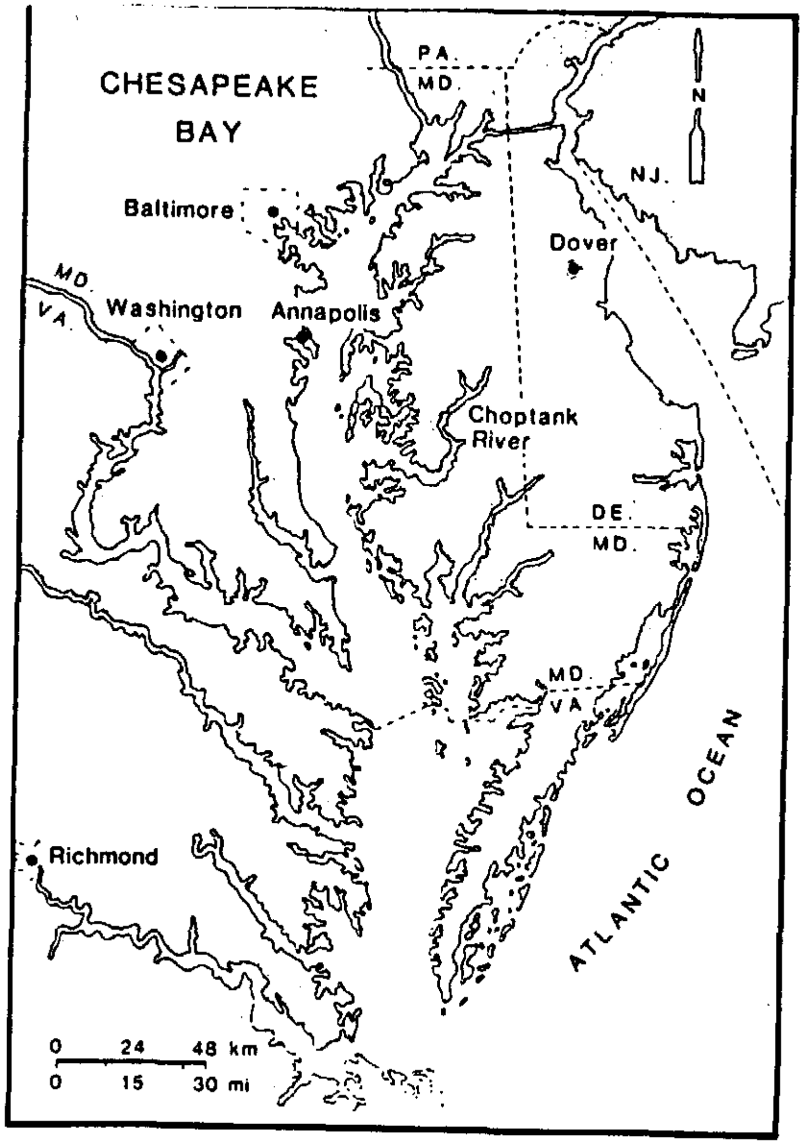


Figure 1. The Chesapeake Bay ecosystem.

The USFWS's Annapolis Field Office recently became involved in a multiagency program to restore Chesapeake Bay and, therefore, needed to understand clearly each problem that influences environmental quality and how that problem is related to *other problems in the Bay*. Once problems are arrayed, the USFWS can then: (1) analyze each problem or set of problems; (2) prioritize them in terms of relative importance; (3) set management goals for affected resources; (4) design economically feasible remedial measures; and (5) measure progress toward management goals.

Methods

In cumulative impacts assessment, it is tempting to evaluate what is readily quantifiable yet not meaningful. Usually, the difficulty is to evaluate what is meaningful but not readily quantifiable. One category of problems can be formulated in a relatively easy manner by use of mathematical models, suitable algorithms and a computer's data-processing capabilities (Schenk 1986). Another category of problems can be formulated only with much difficulty, or not at all, and is addressed only by a specialist whose problem-solving ability is potentially far superior to computer methods. The process used in this study relies on the problem-solving abilities of a group of resource management experts working cooperatively and collaboratively in a workshop setting.

The important problems affecting Chesapeake Bay were identified in two workshops held at the USFWS's Annapolis Field Office. Causes, effects and trends of those problems were modeled, and needed actions were identified. Three criteria were used to screen problems before complete analysis in the second workshop: (1) which elements of the Chesapeake Bay restoration are the Field Office's responsibility or concern; (2) whether and when the Field Office would be able to measure success in solving a problem; and (3) which resources or ecological parameters could be managed effectively.

We used an ecological problem-solving approach (Salwasser and Samson 1985, National Research Council 1986) to cumulative impacts assessment for Chesapeake Bay. The complex environmental situation and major efforts being expended on restoration of Chesapeake Bay (Flemer et al. 1983) indicated the focus should be on a basic step—understanding the situation. The process involved identifying the problems contributing to the situation, agreeing on keystone problems, analyzing and documenting keystone problems, and planning corrective actions for keystone problems. The resulting information is then used to convince decision makers and managers of the need for action. The early steps in the problem-solving process are the equivalent of the hypothesis generation and experimental design phases of a scientific study. The process we used is analogous to the diagnostic procedure called "SOAP" used by the medical profession.

S: *Subjective description* of the problems as stated by the patient (we applied the nominal group technique in the first workshop.

O: *Objective description* of the problems based on the medical team's examinations and laboratory test results (we used a resource management team's consensus and an examination of the scientific literature).

A: *Assessment* of the case by itemizing and ranking the major problems as determined by the medical team (we used cause-effect diagramming in the first and second workshops).

P: *Plan* of specific corrective actions for each identified problem (we applied the functional analysis system technique in the second workshop).

The highest priority problems were identified using the nominal group technique (Bakus et al. 1982), which limits verbal interactions to maximize independent thought and input. We started with the premise, "Cumulative impacts are not being adequately addressed in Chesapeake Bay," and then asked "What are the ecological and environmental problems associated with the Bay?" The participants independently identified numerous problems and then ranked the five most important problems. At the second workshop, we re-examined the most important problems based on our cause-effect analyses from the first workshop. As Erickson (1981) found often happens, the concluding problem statements differed substantially from the initial problem statements.

Cause-effect diagramming (Riggs and Inoue 1975) was used to relate the causes and effects of the important problems. One of the major obstacles in dealing with a complex environmental situation is the difficulty in discerning the individual pathways in a larger causal network. Using the problem statement as the focus, for each step to the left of the problem statement in a cause-effect diagram, the question is asked "What are the causes?" And for each step to the right of the problem statement, the question "What are the effects?" is asked. This technique presents information in a logical, understandable and technically defensible format. For presentation purposes, all of the analyses were summarized and abbreviated to their essential points.

In functional analysis system technique (FAST) diagramming, logical, needed actions are identified based on the causal pathways in the cause-effect diagram (Erickson 1981). Discrete tasks are then identified to address each cause nearest to the problem. First, the refined, quantified problem statement is recast as an objective for recovery. Then, at each step to the right the question "How?" is asked and compared to the question "Why?" for each step to the left. Only the causes of the problem are analyzed for remedies; thus, solutions that treat symptoms rather than causes can be avoided.

Results and Discussion

Problem Description

The 13 USFWS participants in the first workshop (November 1985) created a list of 33 problems associated with Chesapeake Bay. The 10 *problems* listed most frequently were: (1) degraded water quality; (2) loss of marsh and wetland habitat; (3) intensive shoreline alteration; (4) loss of submerged aquatic vegetation; (5) excessive nutrients; (6) oversedimentation; (7) poor farming techniques; (8) problems caused by expanding human populations; (9) decline in anadromous fish populations; and (10) overharvest. The first two problems were analyzed at the first workshop.

Analysis of the water quality degradation problem produced the following causal categories: increased water acidity; excessive nutrients; low dissolved oxygen concentration; toxic wastes; sedimentation; increased water temperature; salinity modifications; greater erosion; more algae blooms; and increased amounts of bacteria and viruses. Some of the other *problems* identified by the participants also were listed among the causes of degraded water quality.

Analysis of loss of wetland habitat produced the following causal categories: pollution (point source and nonpoint source); wetland drainage; fill operations; dredging; freshwater impoundment; wave and current scour action; wildlife grazing damage to submerged aquatic vegetation; and rising sea level. Here, also, a number of the *problems* identified by the participants were listed as causes of loss of wetland habitat. Problem statements of water quality degradation and wetland habitat loss proved to be too broad for productive cause-effect analysis.

As a result of the first workshop, we learned that we should specify how to measure a success, focus on the responsibilities and concerns of the USFWS, and then rank-order the important problems facing the Annapolis Field Office. The essential points were "How would we measure success?" and "Do we have enough control to be successful?" and "Is that measurement of success a clear responsibility of our organization?" At the second workshop (January 1986), five participants (the authors) analyzed three of the high priority problems facing the Field Office: (1) the decline of overwintering populations of canvasback ducks; (2) the decline of submerged aquatic vegetation; and (3) the increase of suspended particulates in a sample watershed.

Problem Analysis

Canvasback ducks. The effects of the decline in overwintering populations of canvasback ducks have been generally negative, mostly in terms of decreased viewing and hunting success (Table 1). Because of the simultaneous but unrelated increase in the Canada goose population, however, the level of popular concern is not as high as it might be (Meanley 1982). The USFWS has significant control over legal and illegal harvest (through hunting regulations and law enforcement), lead shot poisoning (through lead shot restrictions) and avian disease (through breakup of duck concentrations). However, these are not perceived as important causes of the canvasback duck decline. The decline of the canvasback's preferred food, submerged aquatic vegetation, is the most important cause of the overwintering population decline.

Of some 20 species of submerged aquatic plants in Chesapeake Bay, 7 of the 10 predominant species are regularly used by waterfowl (Meanley 1982)—wild celery (*Vallisneria americana*), southern naiad (*Najas guadalupensis*), muskgrass (*Chara* spp.), redhead grass (*Potamogeton perfoliatus*), sago pondweed (*Potamogeton pectinatus*), widgeon grass (*Ruppia maritima*) and eelgrass (*Zostera marina*). The abundance of a major alternative food source for canvasbacks, the Baltic clam (*Macoma baltica*), has also declined. Canvasbacks now use wintering grounds in the North Carolina sounds, where submerged aquatic vegetation and the Baltic clam are still abundant. Restoration of submerged aquatic vegetation would appear to offer the greatest hope of increasing the winter canvasback duck population.

For the canvasback duck decline, we can measure management success (by aerial surveys of duck populations), and the USFWS has a legal responsibility for main-

Table 1. Causes and effects of canvasback duck decline on Chesapeake Bay (the causes and effects perceived as most important by the U.S. Fish and Wildlife Service's Annapolis Field Office are capitalized).

Causes	Problem statement	Effects
•Increased boat traffic •Increased shoreline development •Increased human presence on shoreline •Excessive legal harvest and crippling loss •Illegal harvest •Lead shot poisoning •DECLINE OF THE PREFERRED FOOD SOURCE (SUBMERGED AQUATIC VEGETATION) •REDUCED PRODUCTION OR SURVIVAL ON THE BREEDING GROUNDS •RELOCATION TO NEW WINTERING GROUND IN THE CAROLINA SOUNDS •Competition from other species •Catastrophic events, such as freezes •Chronic pollution effects on an alternate food source, the Baltic clam •Waterfowl diseases such as avian cholera	THE OVERWINTERING POPULATION OF CANVASBACK DUCKS HAS DECLINED ON CHESAPEAKE BAY BY 70% FROM 1958 LEVELS AND HAS FAILED TO RECOVER	♦ Increased hunting pressure on other species such as Canada geese ♦ Potentially less species diversity and a less stable ecosystem ♦ DECREASED OPPORTUNITIES AND SUCCESS IN OBSERVING AND HUNTING CANVASBACK DUCKS ♦ Increased costs for canvasback duck research and restoration ♦ Shift in the cultural values, aesthetics and ethics of waterfowl and waterfowl hunting ♦ Reduced recreational demand and negative impact on the local economy

tenance of canvasback duck populations. The USFWS, however, does not have control over the major factors limiting overwintering canvasback duck populations. Thus, efforts expended on direct restoration of overwintering populations of canvasback ducks on Chesapeake Bay may be wasteful and perhaps futile. This same conclusion may well apply to other declining migratory waterfowl species—redhead (*Aythya americana*), American wigeon (*Anas americana*) and pintail (*A. acuta*).

Submerged aquatic vegetation. The most important effects of the decline of submerged aquatic vegetation are reduced food and cover for shellfish, finfish and waterfowl, which lead to reduced abundance of those animals and, consequently, to reduced hunting, fishing and observation opportunities (Table 2). The major decline of submerged aquatic vegetation in Chesapeake Bay has occurred since Hurricane Agnes in June 1972. Although submerged aquatic vegetation beds recovered within two or three years from damage by a more severe hurricane in August 1933, the expected recovery following Agnes has not occurred (Stevenson et al. 1979). With

Table 2. Simplified causes and effects analysis of submerged aquatic vegetation decline in Chesapeake Bay (the causes and effects perceived as most important by the U.S. Fish and Wildlife Service's Annapolis Field Office are capitalized).

Causes	Problem statement	Effects
•Deteriorated substrate from sediment deposition and fill operations •Reduced substrate availability due to dredging operations and scouring •Water temperature warming •INCREASED SUSPENDED PARTICULATES FROM ALGAE BLOOMS AND SEDIMENT •More biocides in the form of herbicides, pesticides and chlorine •More toxic wastes in the form of heavy metals and petroleum hydrocarbons •Increased water acidity from acid precipitation and inflow •Salinity modification from tropical storms, inflow reduction and channelization •COMPETITION FOR LIGHT AND SPACE FROM ALGAE AND EPIPHYTES •Overgrazing and disturbance by fish and wildlife •Invasion and competition by exotic vegetation species •Invasion and infestation by diseases and parasites	SUBMERGED AQUATIC VEGETATION DISTRIBUTION HAS DECLINED IN CHESAPEAKE BAY BY 70% FROM 1969 LEVELS AND HAS FAILED TO RECOVER	•Increased scour from winds, currents and boat traffic ♦ Increased rate of sediment resuspension •REDUCED FOOD AND COVER RESOURCES FOR SHELLFISH, FINFISH AND WATERFOWL ♦ REDUCED ABUNDANCE OF SHELLFISH, FINFISH AND WATERFOWL ♦ REDUCED SPORT AND COMMERCIAL FISHING SUCCESS ♦ Reduced buffering from the effects of acid precipitation and toxic wastes ♦ Reduced dissolved oxygen available to aquatic fauna ♦ Reduced photosynthesis and biomass production

the exception of the invasion of the introduced Eurasian watermilfoil (*Myriophyllum spicatum*), almost all submerged aquatic vegetation species have declined simultaneously.

Hurricane Agnes caused prolonged freshening of Bay waters, which did extensive damage to submerged aquatic vegetation beds (Stevenson et al. 1979). Regrowth to former abundance has not occurred, and it is generally believed that eutrophication of the Bay, not salinity modification, is limiting the reestablishment of submerged aquatic vegetation. Net primary production of submerged aquatic vegetation was 40 percent of the total for submerged aquatic vegetation and algae in 1963; in 1975, it was 6 percent of the total. Excessive amounts of nutrients in agricultural runoff waters are stimulating green and blue-green algae blooms. These blooms increase turbidity and increase biochemical oxygen demand. Suspended sediment from erosion is believed to be preventing reestablishment of many submerged aquatic vegetation beds. Numerous other causes may contribute to the depression of submerged aquatic vegetation regrowth, but are considered to be relatively minor.

Management success could be readily measured in area and biomass of submerged aquatic vegetation beds. The USFWS has a responsibility to safeguard wetlands, but no direct regulatory authority over any of the important causes of submerged aquatic vegetation decline. The agencies that the USFWS works with on the Chesapeake Bay restoration, however, have management control (e.g., Maryland Department of Natural Resources) and regulatory control (e.g., U.S. Army Corps of Engineers). By cooperating with these agencies, the USFWS can work on the submerged aquatic vegetation decline problem in a significant way.

Suspended particulates. The negative effects of excessive suspended particulates are pervasive across physical habitat, submerged aquatic vegetation, benthic invertebrates, shellfish and finfish (Table 3). The Choptank River (Figure 1) was chosen as an example watershed because it has a high historic and existing (but degrading) fish and wildlife resource value, and it has a high ranking by the Maryland Department of Agriculture in the Chesapeake Bay nonpoint source pollution program. The problem statement was restricted to the upper portion of the watershed, which is primarily influenced by intrawatershed inputs. There, as in Chesapeake Bay in general, the current major concerns are nonpoint source loadings of agricultural fertilizers, sediment, animal wastes and pesticides.

The majority of the Choptank River watershed is in agricultural land use, and point source discharges of industrial effluent and municipal sewage are not significant concerns. The major chemical cause of suspended particulates in the entire watershed is high nutrient concentrations. Approximately 94 percent of the total nitrogen and 63 percent of the total phosphorus in the aquatic system are due to nonpoint sources (T.R. Fisher personal communication; 1986). Increased nutrient loading from the upper watershed and the nuisance algae blooms in the lower watershed are largely caused by excess agricultural fertilizers and increased drainage rates. The major physical cause is increased sediment loading due largely to land-clearing activities, agricultural practices and natural erosion.

Management success with the suspended particulates problem could be measured using on-site samples and turbidity measurements or remotely sensed reflectances. The USFWS has only indirect responsibility for suspended particulates (as they affect fish and wildlife) and no regulatory authority. The agencies that the USFWS works with on the Chesapeake Bay restoration have regulatory authority over only part of the problem (i.e., federal permits for channelization of riverine and palustrine wetlands). By working cooperatively on the current revision of the Section 208 watershed management plans through the Federal Clean Water Act, however, the USFWS can influence the suspended particulates problem, but only on a watershed-by-watershed basis. In total, the suspended particulates problem is measurable, not readily manageable and an indirect responsibility of the Service as it relates to fish, wildlife and their habitat.

Plan of Corrective Action

A major step in the cumulative impacts assessment process is the synthesis of a plan for correcting each problem. The USFWS has the best opportunity for success

Table 3. Simplified causes and effects analysis of the increase of suspended particulates in the upper Choptank River watershed (the causes and effects perceived as most important by the U.S. Fish and Wildlife Service's Annapolis Field Office are capitalized).

Causes	Problem statement	Effects
Algae Blooms ●Nutrient loading from municipal and industrial sources ●Nutrient loading from manure ●NUTRIENT LOADING FROM EXCESS AGRICULTURAL FERTILIZERS ●NUTRIENT TRANSPORT FROM INCREASED AGRICULTURAL DRAINAGE **Sediment** ●LAND-CLEARING ACTIVITIES AND DRAINAGE CONSTRUCTION ●AGRICULTURAL PRACTICES ●NATURAL EROSIVE PROCESSES ●Natural catastrophic events	THE AMOUNT OF SUSPENDED PARTICULATES HAS INCREASED IN THE WATERS OF THE UPPER CHOPTANK RIVER WATERSHED	♦ Altered timing and location of fish spawning ♦ INTERFERENCE WITH RESPIRATORY AND FILTER FEEDING MECHANISMS OF FINFISH, SHELLFISH AND INVERTEBRATES ♦ FILL INTERSTITIAL SPACES IN STREAM SUBSTRATE ♦ REDUCTION OR ELIMINATION OF SPAWNING HABITAT ♦ COVER AND REDUCE BIOMASS OF BENTHIC INVERTEBRATES ♦ REDUCTION OF DISSOLVED OXYGEN AND INCREASED ANOXIC CONDITIONS ♦ INCREASED EPIPHYTIC GROWTH ON SUBMERGED AQUATIC VEGETATION (SAV) ♦ REDUCED PHOTOSYNTHESIS AND GROWTH OF SAV ♦ Reduction or elimination of susceptible finfish eggs and larvae

with the three problems here analyzed by influencing submerged aquatic vegetation restoration. The problem statement was changed into an objective of restoration of submerged aquatic vegetation. An analysis of a problem's causes should lead to identification of the elements of a solution. We planned a set of tasks to address the most important problem causes (Table 4). Most of the corrective efforts were directed toward decreasing nutrient and sediment loading in Bay waters. Additional tasks were directed toward special management and damage reduction for existing and historic submerged aquatic vegetation beds; research on the effects on submerged aquatic vegetation beds of navigation channelization, exotic species invasion and aquatic herbivores; and measurement of restoration effort success by aerial photography missions, submerged aquatic vegetation bed mapping and regular vegetation biomass sampling.

Table 1. Task planning to achieve the objective of restoring submerged aquatic vegetation in Chesapeake Bay (the actions and tasks perceived as most important by the U.S. Fish and Wildlife Service's Annapolis Field Office are capitalized)

Problem statement	Objective	Actions	Tasks
		INCREASE LIGHT TRANSMISSION BY DECREASING NUTRIENT LOAD	•Identify agricultural best management practices (BMPs) for fish, wildlife and their habitat ○ASSESS EXCESS NUTRIENT IMPACTS ON VALUABLE WATERSHEDS ○REVIEW STATE WATER QUALITY PLANS TO PROMOTE NUTRIENT REDUCTIONS ○INFLUENCE FEDERAL COST SHARING PROGRAMS FOR AGRICULTURE TO PROMOTE NUTRIENT REDUCTIONS
SUBMERGED AQUATIC VEGETATION (SAV) DISTRIBUTION HAS DECLINED IN CHESAPEAKE BAY BY 70% FROM 1969 LEVELS AND HAS FAILED TO RECOVER	INCREASE THE DISTRIBUTION OF SUBMERGED AQUATIC VEGETATION IN CHESAPEAKE BAY TO 1969 LEVELS (200,000 ACRES)	INCREASE LIGHT TRANSMISSION BY DECREASING SUSPENDED SEDIMENT	•PROMOTE RETENTION AND USE OF RIPARIAN FOREST BUFFER STRIPS AND SHELTERBELTS ○Promote BMPs for dredging and channelization ○Review Section 10/404 permits and environmental impact statements to promote erosion control ○Influence federal cost sharing programs to maximize soil retention ○IDENTIFY BMPs TO REDUCE SOIL LOSS FROM AGRICULTURAL LANDS ○Identify BMPs to control erosion in problem areas and situations
		Provide special management for critical areas	•SURVEY SAV DISTRIBUTION ANNUALLY ○Sample SAV biomass and species composition annually ○Map existing and historic SAV beds ○Assess the value of SAV transplants into suitable areas
		Reduce damage to existing beds	•Review Section 10/404 permits to reduce dredge/fill effects ○Remove or reduce destructive concentrations of aquatic herbivores

Conclusions

The cumulative impacts assessment process described here can be used for problem analysis and program planning. Although the process is simple, it requires a detailed examination of the components of each problem. One key to a successful problem analysis or program preparation is careful specification of the problem statement or objective. Too broad a problem statement may lead to failure to examine each of the individual problem components in enough detail to be useful. The analysis and planning results can be most effectively displayed in a diagram or flowchart because of the difficulty in presenting the complex information in a text format. While all of the interrelationships could be represented in the more traditional box-and-arrow diagram, we believe that the information is more understandable and easier to communicate in the cause-effect diagram format. Application of the process showed that it is a valuable organizer of group thinking, an easy way to obtain understanding and general acceptance, and a comprehensive method for identifying specific restoration tasks.

Potential users of the products should be involved and have primary responsibility for the analysis. This creates a sense of commitment and responsibility in each participant. Involvement of personnel from many agencies and disciplines will result in closer coordination and greater cooperation in the achievement of resource objectives. Decision maker support will be more likely because of the logical and defensible approach, well-defined scope of the effort, and easily understood process. Additionally, individuals in each participating agency can understand how their activities and responsibilities mesh.

In this cumulative impacts assessment process, we used a few simple techniques to study a complex situation. We generated a hypothesis that restoration of submerged aquatic vegetation should be a major thrust of the Chesapeake Bay restoration. Submerged aquatic vegetation decline is a keystone problem that can be measured, monitored and managed, and it directly relates to declines in abundance of migratory fish and wildlife species (Price et al. 1985). We believe that submerged aquatic vegetation should be a central focus of the restoration of Chesapeake Bay. Distribution and biomass of submerged aquatic vegetation, as opposed to measurements of nutrient concentrations and toxic chemical loadings, can serve as an integrator of human impacts on the Bay and as a quantitative indicator of the environmental quality of the Bay. Living resources instead of water quality parameters would serve as excellent long-term measures of the success of our restoration efforts.

Acknowledgments

We thank the staff of the National Ecology Center and the Annapolis Field Office for their assistance with trying this approach to cumulative impacts assessment.

References Cited

Bakus, G.E., W.E. Stillwell, S.M. Latter, and M.C. Wallerstein. 1982. Decision making: With applications for environmental management. Environmental Management 6:493-504.

Capper, J., G. Power, and F.R. Shivers, Jr. 1983. Chesapeake waters: Pollution, public health and public opinion, 1607-1972. Tidewater Publishers, Centreville, Maryland. 201p.

Erickson, S.M. 1981. Management tools for everyone: Twenty analytical techniques that are easy to learn and valuable to know. Petrocelli Books, Inc., New York. 170p.

Flemer, D.A., G.B. Mackiernan, W. Nehlsen, V.K. Tippie, R.B. Biggs, D. Blaylock, N.H. Burger, L.C. Davidson, D. Haberman, K.S. Price, and J.L. Taft. 1983. Chesapeake Bay: A profile of environment change. U.S. Environmental Protection Agency, Philadelphia, Pennsylvania. 200p.

Lippson, R.L. 1985. Chesapeake Bay: History and management needs. Trans. N. Amer. Wildl. and Natur. Resour. Conf. 50:495-501.

Meanley, B. 1982. Waterfowl of the Chesapeake Bay country. Tidewater Publishers, Centreville, Maryland. 210p.

National Research Council. 1986. Ecological knowledge and environmental problem solving: Concepts and case studies. National Academic Press, Washington, D.C. 388p.

Odum, W.E. 1982. Environmental degradation and the tyranny of small decisions. BioScience 32:728-729.

Price, K.S., D.A. Flemer, J.L. Taft, G.B. Mackiernan, W. Nehlsen, R.B. Biggs, N.H. Burger, and D.A. Blaylock. 1985. Nutrient enrichment of Chesapeake Bay and its impacts on the habitat of striped bass: A speculative hypothesis. Trans. Amer. Fish Soc. 114:97-106.

Riggs, J.L., and M.S. Inoue. 1975. Introduction to operations research and management science. McGraw-Hill, New York. 497p.

Salwasser, H., and F.B. Samson. 1985. Cumulative effects analysis: An advance in wildlife planning and management. Trans. N. Amer. Wildl. and Natur. Resour. Conf. 50:313-321.

Schenk, T. 1986. A robust solution to the line-matching problem in photogrammetry and cartography. Photgr. Eng. Remote Sensing 52:1,779-1,784.

Stevenson, J.C., N. Confer, and C.B. Pieper. 1979. The decline of submerged aquatic plants in Chesapeake Bay. U.S. Fish Wildl. Serv. FWS/OBS-79/24.

Williamson, S.C., C.L. Armour, and R.L. Johnson. 1986. Preparing an FWS cumulative impacts program: January 1985 workshop proceedings. U.S. Fish Wildl. Serv. Biol. Rep. 85(11.2). 37p.