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Wetland and Riparian Restoration: Taking A Broader View

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Society for Ecological Restoration
International Conference, September 14-16, 1995,
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Editors: Keith B. Macdonald
Fred Weinmann

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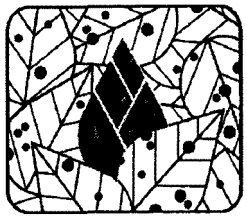
Sponsorship

The Northwest Chapter of the Society for Ecological Restoration and the Pacific Northwest Chapter of the Society of Wetland Scientists are pleased to sponsor the publication of this series of papers and abstracts on wetland and riparian restoration. The decade of the 90s has been a period of emergence for the expanding field of aquatic ecosystem restoration and during this time a confluence has occurred between the critical need for action and increased technical understanding of "how to do it." At the same time, the credibility of aquatic restoration as a scientific discipline was increased by two published compendiums: *Restoration of Aquatic Ecosystems* by the National Research Council of the National Academy of Science (National Academy Press, 1992) and *Restoring the Nation's Marine Environment* (Gordon W. Thayer, ed., Maryland Sea Grant College, 1992).

The list of authors represented in this Proceedings is a virtual who's who in the field of habitat restoration, particularly within the Pacific Northwest. The papers reflect the high quality work of many authors. We believe that these papers—in their diversity—represent a collection of thoughtful approaches to restoration issues, a source of reference information for other practitioners, and a measure of the state-of-the-art at the time of their preparation. As such, we believe they will assist restoration science through the planning and implementation of sound habitat restoration projects on the ground.



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The initial invitation that the senior editor organize a conference session devoted to wetland restoration came from 1995 SER Conference Co-Chairs Sono Hashisaki (Springwood Associates Inc.) and Tim White (CH2M HILL). During an early planning session with Sono and Tim, we came up with the conference theme, ***Taking A Broader View***—that we subsequently invited wetland contributors to consider in their talks and papers. With Mary Kentula's (U.S. EPA, Environmental Research Laboratory, Corvallis) enthusiastic support the topic rapidly expanded to include riparian restoration, and two full days of talks were organized. During the SER Conference, significant interest was expressed in soliciting papers from the speakers and publishing a proceedings focused on broadening perspectives in wetland and riparian restoration. Our special thanks go to all those who by making the 1995 SER conference such a success, indirectly made our task much easier.

Special thanks also go to Chapter Presidents David Chapin (SER), Janet Morlan and Curtis Tanner (SWS), and their respective Chapter boards, for their joint support and endorsement of these proceedings.

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The senior editor particularly thanks Fred Weinmann (U.S. EPA, Region 10) who enthusiastically endorsed the concept of publishing these proceedings, obtained EPA funding to do so, and then shepherded the final manuscript through the printers.

About These Proceedings

The 1995 International Conference of the Society for Ecological Restoration, organized around the theme ***Taking A Broader View***, was held on the University of Washington, Seattle Campus in September 1995. During the conference's wetlands sessions, significant interest was expressed in publishing a wetland/riparian-based Proceedings volume. The papers and abstracts included here were solicited from conference participants throughout 1996. A summary paper highlighting changing perspectives on wetland and riparian restoration was then added to complete the Proceedings.

The papers and abstracts included in this Proceedings volume have been divided among three groups: Taking a Broader View, Taking a Closer Look, and Works in Progress. Where possible, papers and abstracts sharing related themes have been grouped together.

Taking a Broader View—These 14 contributions are generally broad in scope and context. They consider the role of wetland and riparian habitats on a landscape or watershed scale, emphasizing regional approaches to wetlands restoration planning. Each contribution demonstrates the value to be gained from taking a broader view. In apparent contrast, Klaus Richter's paper focuses on detailed habitat requirements of lentic breeding amphibians—yet his conclusions stress critical "landscape linkages" between wetland and upland habitats, without which the amphibians would not survive. Ron Thom et al. provide a unique perspective on restoring shallow-water eelgrass habitat in Puget Sound, Washington.

The 11 contributions in ***Taking a Closer Look***, cover a broad range of topics but generally focus in greater depth on individual restoration sites: Jetty Island, the Salmon River, Batiquitos Lagoon, an Illinois Kettle marsh, and South Carolina bottomland hardwoods. Thom et al. explore the surprisingly difficult task of accurately determining habitat restoration project costs, while Marc Boule addresses the sensitive issue of balancing restoration priorities among different agencies during a complex wetland mitigation negotiation. Papers by Lisa Levin et al., Scott Simon et al., and Poole and Trometer each recommend methods for evaluating individual restoration projects that could be broadly applied in other locations.

Works in Progress, include 38 extended and short abstracts. The first 17 of these are extended abstracts—modified and expanded by the authors following the SER 95 Conference. In many cases the addition of literature citations will help the reader find what they need more efficiently. The remaining 21 short abstracts are reprinted just as they appeared in the SER 95 Conference Program. They are included here to illustrate the tremendous breadth of studies being pursued by wetland and riparian habitat restorationists.

The real purposes of a Proceedings volume such as this are to stimulate interest in the topics covered and encourage active networking among practitioners across the country. Current author's addresses are included for all contributions and readers are urged to contact workers involved in projects of particular interest to them.

New Directions in Wetland and Riparian Restoration

Keith B. Macdonald[1]
Fred Weinmann[2]

Abstract: **Before passage of federal environmental legislation in the late 60s and early 70s, wetland and riparian studies exhibited a wide variety of disparate research interests. Due to their rapid decline and the suggestion that detrital export subsidized valuable coastal fish and shellfish resources, coastal wetlands captured the greatest attention.**

More focused research themes emerged after passage of the Federal Water Pollution Control Act Amendments of 1972 (Section 404). Regulatory agencies and the development community focused on defining the shared fundamental characteristics of all wetland types—hydrology, hydric soils, and hydrophytic vegetation—as well as how they were to be identified and delineated in the field. Once adequately defined and classified, then inventories of wetland types and trends in habitat abundance were pursued. Clearer definition of wetland fundamentals also resulted in a greater focus on unique regional differences among wetland types, wetland functions, and the perceived "values" of wetlands across the nation. Two additional research themes emerged during this period: the development of treatment wetlands, engineered to enhance water quality; and the rapid-expansion—driven largely by mitigation requirements—of habitat restoration projects. Subsequent studies examined the relative success of mitigation wetlands—particularly the degree to which they duplicated the functions and processes of natural wetlands. In the 70s and 80s, wetland and riparian habitat restoration projects were planned and installed project-by-project, site-by-site. Typically, we stood at the edge of the wetland looking in—focusing on what we saw, how it functioned, and how it could be duplicated.

Important new trends have emerged since the early 90s, changes captured by the SER Conference theme of *Taking A Broader View*. These new trends are being driven by the need for cost-effective regulatory streamlining, as well as increasing recognition that critical issues such as water quality enhancement, sensitive species protection, and habitat restoration need to be addressed in a broader landscape, watershed, or ecosystem management context. Wetland and riparian restoration projects are now seen as integral parts of the broader landscape and there is a critical need to identify landscape linkages and focus on interactive processes that "drive" the structure and function of these habitats. Wetland and riparian habitat restoration is rapidly moving beyond regulatory-driven mitigation projects into incentive-based, larger scale, watershed enhancement and habitat conservation planning programs. These larger programs require more careful analysis of project goals and objectives, more effective decisionmaking, and increasing public involvement.

Introduction

One could say that wetlands, as we think of them today, were officially created after passage of the Federal Water Pollution Control Act Amendments of 1972 (FWPCA, PL 92-500; amended again as the Clean Water Act in 1977). Prior to that time "wetlands" was a colloquial term used mostly by farmers and sportsmen to describe riverbottom lands and seasonally flooded meadows and forests. A contemporary dictionary definition was, "land containing much soil moisture" (Webster 1965). While wetland research had been expanding in the 50s and 60s it stressed specific habitats—salt marsh, prairie potholes, Cypress swamps, peatlands—and had not

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yet focused on their underlying shared fundamentals of hydrology, hydric soils, and hydrophytic vegetation. Conversion of wetlands for agriculture or construction was still the norm and while the U.S. Fish and Wildlife Service supported wetland protection (Shaw and Fredine, 1956), some other federal agencies promoted drainage and conversion. Along with the scientists, conservation and sporting groups were increasingly drawing attention to the broader environmental consequences of continued wetland and riparian habitat losses.

Section 404 of the FWPCA, the Dredge-and-Fill Permit Program, provided the first effective tool for wetlands protection. In 1977, President Carter's Executive Orders on the Protection of Wetlands and Floodplain Management (11990 and 11988, respectively) initiated federal policy reviews that expanded wetlands protection. Subsequently, in 1988, presidential candidate George Bush's support for the concept of "No Net Loss" (Conservation Foundation 1988) provided the foundation for present policies that seek to increase wetland and riparian acreages across the United States.

Much of the growth in wetland and riparian research, as well as today's expanding restoration programs, ultimately sprang from these changing federal mandates and their incorporation into state and local statues.

So Where Have We Been?

What then were some of the older themes of wetland and riparian research? And how has the direction of research changed to meet the needs of new wetland management regulations?

Early Wetland Research[3]

The earliest systematic studies of United States wetlands followed passage of the Swamp Land Act (1849, as amended). This Act transferred "swamp and overflow lands" from the federal to state governments and actively encouraged wetland drainage and "reclamation". Various reports document the reclamation potential for both interior riverbottom (Mississippi Basin) and coastal wetlands (Wright 1907, Nesbit 1885). U.S. Coast Survey charts produced at about this same time often include detailed mapping of coastal wetlands.

Wetland and riparian publications were scarce before 1960 and generally focused on specific disciplinary concerns. Early interests included the land-building role of coastal marshes (Shaler 1885) and mangroves (Davis 1940), and their relationship to changing sea levels (Chapman 1938). Coastal wetland natural history observations (Yapp 1917, Dexter 1947) and floristics (Purer 1942, Mason 1957) progressed into studies of primary production (Penfound 1956, Pomeroy 1958) and energy flow (Teal 1958, Smalley 1959).

Vegetation dynamics and energy flow within cool temperate peatlands–bogs (muskegs) and fens–also received early attention (Transeau 1903, Lindeman 1942). Study and protection of prairie pothole marshes was promoted by duck hunting organizations as early as 1916 to support production of migratory waterfowl (Linduska and Nelson 1964). Early riparian studies focused on vegetation changes in response to river channel dynamics and flooding (Fitzpatrick and Fitzpatrick 1902, Turner 1936, Thompson 1961).

Ecosystem Studies

The close of the 50s saw two important benchmarks for wetland research. In March 1958, the National Science Foundation sponsored the first ever *interdisciplinary* conference on coastal salt marshes, at the University of Georgia Marine Institute, Sapelo Island (Ragotzkie 1959). Then, 2 years later, publication of *Salt Marshes and Salt Deserts of the World* (Chapman 1960) provided the first monographic treatment of wetland research. Both events greatly increased awareness and visibility of research on coastal wetlands. Soon after the Sapelo Island conference, John Teal published his pivotal salt marsh trophic synthesis paper suggesting that 45 percent of marsh production was exported "to support an abundance of animals," in adjacent estuaries (Teal 1962). Coastal oceanography was a beneficiary of sharply increased federal research funding in the early 60s. Teal's suggestion of massive detrital export captured attention and wetland research rapidly expanded, especially along the Atlantic and Gulf Coasts.

U.S. Fish and Wildlife Service initiated the National Estuary Study (USFWS 1970), and major symposia were published on estuaries (Lauff 1967)

[3] Representative studies only; see Mitsch and Gosselink (1993) for others. Crowcroft (1993) and Wilson (1994) provide valuable insights into changing ecological perspectives since the 30s.

and coastal lagoons (Castañares and Phleger, 1969). Much of the steadily expanding research of the 60s and 70s focused on the role of marsh production (Kibby et al. 1980), nutrient flux, and detrital export in estuarine ecosystems (Nixon 1980). Research soon suggested a linkage between marsh production and offshore commercial fisheries (Turner 1977). Along with concerns over habitat loss from duck hunters and sports fisherman, it was the potential linkage between destruction of estuarine wetlands and declining finfish and shellfish resources that provided the strongest political support for regulatory protection of wetlands (Clark 1967, Chabreck 1973, Nixon 1980).

The federal regulatory landscape was also changing markedly during this period. The Fish and Wildlife Coordination Act (USFWS) was enacted in 1967, the National Environmental Policy Act (NEPA) in 1969, and both the Endangered Species Act and Coastal Zone Management Act in 1972. The FWPCA of 1972 was also further amended as the Clean Water Act in 1977. In many cases, passage of these federal regulations led to establishment of closely similar state mandates. The overall result was closer scrutiny of the potential impacts of construction projects, a variety of tools to protect wetland habitats, and the potential use of mitigation to offset wetland losses (ASWM 1986).

Regional Differences

The emergence of NEPA and the Clean Water Act as tools to protect wetland and riparian habitats against the "onslaught of development" was eagerly embraced in the environmentally active 70s and 80s. There was increasing concern, however, that the functions attributed to Atlantic Coast wetlands, especially Georgia's intensively studied Sapello Island marshes, were neither broad enough (Gosselink et al. 1974), nor universally characteristic (Onuf et al. 1979). A national symposium organized in 1978 by the National Wetlands Technical Council and the Conservation Foundation broadly addressed the issue of wetlands functions and values and resulted in a landmark publication (Gresson et al. 1979). Research on wetland functions led in two different directions, one focusing on the classification of different wetland types (Cowardin et al. 1979), while the second documented characteristics of wetlands in different regions of the United States.

Research on wetland functions and classification led to the development of methodologies for wetland functional assessment (Adamus 1983), that ultimately resulted in development and application of the presently evolving hydrogeomorphic wetlands classification system (Brinson 1993, 1995, 1996; Smith et al. 1995).

Descriptive literature on regional wetlands expanded rapidly in the 70s. The USFWS and California Department of Fish and Game published the *Coastal Wetland Series*, describing priority coastal wetlands in California (e.g., Macdonald 1976). Other states quickly followed with similar publications (e.g., USACOE 1976; Shapiro & Assoc. 1978). The immediate success of these reports, both as a foundation for future research and for management purposes, resulted in an expanded series of USFWS publications, *Community Profiles* and later, *Estuarine Profiles.* While the initial impetus was coastal wetlands (Zedler 1982, Nixon 1982), studies of freshwater wetlands and riparian habitats (Faber et al. 1989) soon followed. In addition, numerous studies of individual wetlands, nationwide (e.g., McCormick et al. 1970), and several national symposia focused on wetland types and processes (Good et al. 1978, Hamilton and Macdonald 1979, Brinson et al. 1981, Warner and Hendrix 1984).

Wetland Delineation

As a direct practical response to calls for broad regulatory protection of wetlands, both the regulatory agencies and the development community needed to know exactly what was being protected, how it was to be defined, and how a legally defensible boundary line could be drawn around it on the ground. This, of course, led to extensive debate concerning a legal definition of wetlands and/or "jurisdictional wetlands" and on standardized methodologies that could be employed for wetland delineation (USACOE 1979, 1987; Federal Interagency Committee 1989, EDF/WWF 1992). Publication of the National Research Council's, *Wetlands: Characteristics and Boundaries* (NRC 1995), has helped achieve federal consensus on these issues, but the debate continues on some state and local levels.

Wetland Inventories

Along with wetland delineation issues there was substantial interest in knowing how the nation's wetland and riparian habitats were distributed, how much acreage of different wetland types remained, and how much had been lost.

Early inventories are described by Shaw and Fredine (1956), who focused their own studies on wetlands for waterfowl support. Following development of the Cowardin et al. (1979) wetland classification, the USFWS initiated the National Wetlands Inventory (NWI) based on color-infrared aerial photography. Since field reconnaissance to confirm general wetland boundaries remains a slow and costly process, NWI status varies by location. Some areas of the country still remain to be mapped while in others NWI is already well into its second generation of map products. Earlier studies, along with the NWI data are summarized by Dahl (1990) and Dahl and Johnson (1991). These data offer our first quantitative overview of the nation's wetland resources and provide a baseline against which to measure future changes.

Treatment Wetlands

Increased understanding of wetland and riparian nutrient fluxes, along with monitoring of wastewater discharges into natural wetlands, identified the critical role these systems can play in maintaining water quality (Gosselink et al. 1974). In the 70s, this knowledge was applied to a new research theme—the development of treatment wetlands, expressly engineered and built for water quality treatment. This activity has steadily gained importance, contributing significantly to better understanding of natural wetland functions and practical field methods for wetland enhancement and creation (Kadlec and Knight 1996). Under appropriate circumstances, treatment wetlands have been shown to offer subsidiary benefits for wetland-related species (USEPA 1993). The possible accumulation of contaminants (trace metals, organic chemicals, etc.) can, however, remain a concern (Wren et al. 1997).

Restoration

An increasingly important research theme and the one driving these Proceedings, concerns wetland and riparian habitat restoration, enhancement, and creation. Although occasional pioneering wetland restoration projects have been implemented for many years, the field expanded rapidly in the late 70s (Seneca et al. 1975, Garbish 1977, Montanari and Kusler 1978, Josselyn 1982). This reflected the passage first of NEPA (1969), which provided the framework for mitigation programs to balance project impacts, and then Section 404 of the Clean Water Act (1977), which required compensation for wetland losses.

In subsequent years, analysis of 404 permit decisions showed that despite mitigation requirements wetlands continued to be lost (Marcus 1982, Gwin and Kentula 1990). Further, monitoring of early wetland restoration programs showed they often fell short of expectations, particularly in terms of replacing some of the functions and processes typical of comparable natural wetlands (Race 1985, Stickland 1986, Zedler 1988, PERL 1990). Clearly the time required for a restored or created wetland to replace natural wetland functions is much longer than first anticipated (>10 years) and some functions may not be fully replaced (Simenstad and Thom 1996, Zedler 1996b). These insights ultimately led to political support for the "No Net Loss" wetlands protection policy (Conservation Foundation 1988).

The late-80s and 90s have seen publication of increasing numbers of guidebooks recommending detailed procedures for planning, installing, and monitoring wetland and riparian restoration and enhancement programs (Kusler and Kentula 1990, Kentula et al. 1992, Zedler 1984, 1996a; Marble 1992, Vanbianchi et al. 1994, Erwin 1996).

Publications focusing on wetland support for waterfowl (Scott 1982, Smith et al. 1989) and wildlife (Payne 1992), as well as riparian/stream benefits for fisheries (Hunter 1991, Spence et al. 1996) have also become available. The credibility of aquatic restoration as a scientific discipline also gained substantially with publication of two major, national compendiums: *Restoration of Aquatic Ecosystems* by the National Research Council of the National Academy of Science (National Academy Press 1992) and *Restoring the Nation's Marine Environment* (Thayer 1992).

The newer research themes noted above never eliminated the older ones—rather they have steadily expanded the breadth of wetland and riparian

studies now in progress. There are now several excellent textbooks covering different aspects of wetlands or wetland types (Daiber 1982, Mitsch and Gosselink 1986, 1993; Adam 1990, Malanson 1993). Wetland-oriented professional societies (Society of Wetland Scientists, Association of State Wetland Managers) and journals (*Wetlands*, *Wetlands Ecology and Management*) have become popular, and both peer-reviewed and grey literature on wetlands has exploded. The first edition of Mitsch and Gosselink's text *Wetlands* (1986) included nearly 700 literature citations; just 8 years later in their second edition (1993) this number had risen to nearly 1,500 citations!

The Broader View Trend

The accelerating worldwide decline in biodiversity—the total variety of life forms living on the planet—has become the overarching environmental concern of the 90s. Human modification and destruction of habitat is clearly recognized as the principal cause of this decline, reflecting in turn the rapidly expanding human population. In Pogo's now famous admonition (Earth Day 1971), "We have met the enemy and he is us." Public opinion polls suggest that protecting biodiversity, like wilderness before it, has now become a significant quality of life issue.

The 90s may have seemed like a tough decade for the environment with a considerable backlash against the regulatory protections introduced in the 70s. Private property rights are certainly a contentious issue, but frustration over the complexity and cost of fragmented, overlapping, and sometimes contradictory regulatory mechanisms is a much more widespread concern (US General Accounting Office 1994, McCormick et al. 1995).

In the 70s and 80s most environmental concerns were dealt with on a case-by-case basis: site by site, outfall by outfall, tract by tract (whether forests or houses). Regulations were often interpreted with a narrow focus and public involvement was generally limited. Greater awareness of both the need for regulatory streamlining and the accelerating impact of habitat degradation has resulted in significant changes in how environmental issues are now being addressed. The 90s have witnessed some striking changes captured by, "Taking a Broader View," the guiding theme for the 1995 International Symposium of the Society of Ecological Restoration. Examples illustrating these newer approaches to several issues germane to wetland and riparian restoration are outlined below.

Multidisciplinary Training and Hypothesis Testing

Given the prominence of wetland issues today, it is easy to forget that as little as a decade ago few students were formally trained as wetland scientists or restorationists. Rather, individuals "got hooked" while studying wetland and riparian habitats from their own particular disciplinary perspective—geomorphology, groundwater, botany, migratory birds, nutrient cycling, etc. What most of us found was that to better interpret our own disciplinary concerns, we needed to better understand the system as a whole. Recognition of the benefits of a truly multidisciplinary/interdisciplinary approach to wetland and riparian science and restoration is now accepted as central to our further progress.

Another important lesson is that it is costly and ineffective to continually "rediscover" restoration fundamentals. We must learn from and build upon the experience of earlier workers. This requires that restoration projects always be designed as experiments to test clearly defined hypotheses—about hydrologic modifications perhaps, or soil amendments, planting methods, or control of introduced species. Effective monitoring is also a critical component if we are to learn anything from experience.

Water Quality Enhancement

Successive water quality initiatives, focusing first on point source controls, adding nonpoint source controls, and now evolving into the watershed-based "total maximum daily load" (TMDL) process, clearly follow the trend of taking a broader view.

For decades, the primary water quality concern was treatment of municipal sewage and industrial waste discharged into waterways through ditches, pipes, and outfalls—so-called point source pollution. The Federal Water Pollution Control Act Amendments of 1972 were broadly crafted to, "restore and maintain the chemical, physical, and biological integrity of the nation's waters." Implementation of the Act initially focused on point source discharges, subjecting them to water-quality

and technology-based controls imposed by the National Pollution Discharge Elimination System (NPDES) permit process. These controls are generally imposed outfall-by-outfall, one permit at a time. Overall, NPDES controls have been very successful, producing dramatic water quality improvements that allowed marine and freshwater fish, and other organisms, to return to previously degraded habitats. Yet significant pollution remained.

Since the 70s, EPA-sponsored studies of nonpoint source pollution have demonstrated that stormwater flows draining from diffuse locations spread out over the landscape—from street runoff, faulty septic systems, land clearing, agriculture, forestry, and construction practices—also cause significant pollution of wetlands, streams, and other aquatic habitats (AWRA 1988, Jackson 1994, Hopkinson and Vallino 1995, Pitt 1995). Under the FWPCA, these nonpoint sources have been subject to increasingly stringent state regulation through "best management practices" (BMPs) tied to specific industry types and land uses (Washington Department of Ecology 1992). Joint EPA-NOAA guidance for new state coastal nonpoint pollution control programs was implemented in 1993, following the Coastal Zone Act Reauthorization Amendments of 1990 (EPA 1993).

A potential shortcoming of prior cleanup actions has been that neither point source NPDES permits, nor state-imposed nonpoint BMPs, consider the cumulative load of pollutants entering a particular waterbody, nor its ability to satisfactorily assimilate and neutralize the total waste load. This need was foreseen in the FWPCA amendments but has only very recently begun to be implemented. Section 305(b) of the Act requires states to identify and prioritize stream segments or waters not meeting water quality standards. Section 303(d) then imposes the TMDL requirement that mandates reduction of pollution to meet water quality standards designed to protect beneficial uses of each stream segment or waterbody. A TMDL provides a mechanism for quantitative assessment of water quality problems and contributing pollutant sources; allocates pollution control responsibilities between sources in a watershed; and specifies the amount of pollutant or other stressor to be reduced to meet water quality standards. EPA recognizes habitat degradation impacts, and restoration of specific habitat endpoints (e.g., fish spawning habitat) may be allowed as part of the TMDL (EPA 1994, Dilks et al. 1994, WFCA 1997).

Endangered Species Recovery

Changing approaches to implementing the Endangered Species Act (Rohlf 1989, Pulliam and Babbit 1997) also illustrate the trend towards taking a broader view. Following passage of the Act in 1973 and the first federal listings of threatened and endangered species, the initial goal was to summarize ecological life history data for each species and develop an appropriate Recovery Plan (USFWS 1980, 1985). The Act relied on prohibition of a broadly defined "take" to protect endangered species. It soon became apparent, however, that despite each species' unique ecological requirements, the common theme shared by all was loss of critical habitat. The Act permitted "incidental take" of a listed species following preparation and approval of a habitat conservation plan (HCP). Early HCPs focused on the protection of single species, often by transferring development rights from critical habitat and clustering higher-density construction on nearby, less sensitive areas (San Bruno Mountain HCP, Beatley 1995).

Focusing habitat protection to favor single species, even endangered species, revealed several concerns: (a) complex, poorly understood life histories often make it difficult to decide exactly which habitat features—or other ecological interactions—favor survival; (b) enhancing habitat characteristics for one species may place other species occupying the same habitat at greater risk; and (c) a single endangered species often signals critical losses of a particular habitat, indicating that several other co-occurring species are soon likely to be at risk. Southern California coastal wetlands, for example, provide critical habitat for several different endangered species—each with different habitat requirements (Zedler 1996a). Similarly, local riparian willow thickets provide critical nesting habitat for an endangered neotropical migrant songbird, while several other migrants also appear to be in decline (Recon 1989). These concerns—along with new insights on habitat size and landscape distribution required to support sustainable species populations (derived from island biogeography, MacArthur and Wilson 1967, Soulé et al. 1988)—have ultimately led to the development of large-scale, regional,

multiple species HCPs (City/County of San Diego 1996). This suggests that HCPs are evolving from a process adopted primarily to address single species and (construction) development, to a broad-based, landscape level planning tool utilized to achieve long-term biological and regulatory goals (USFWS/ NMFS 1996). While the success of such plans is still in debate (Kaiser 1997, Mann and Plummer 1997, Shilling 1997, Luoma 1998), they certainly reflect taking a broader view.

Wetland Economic Benefits

Society's view of the benefits (values) provided by wetland and riparian resources has changed substantially as understanding of wetland processes and functions has grown (Greeson et al. 1979). Aboriginal uses of wetland resources focused on limited harvest of plants for food and fiber, fish and shellfish, and trapping of fur-bearers. Early trappers and pioneers showed less restraint and quickly exploited, then exhausted, many of these same resources. The "wetlands are worthless swamps" ethic persisted well into the 70s, until their role as nursery grounds for coastal fisheries was understood (Clark 1967). In addition to their biological productivity and food chain support, Gosselink et al. (1974) drew attention to wetlands' role in water quality enhancement and nutrient cycling. Others noted their role attenuating flood flows and mediating groundwater flux.

The critical role of riparian forests in creating and maintaining stream habitat for trout and salmon (Spence et al. 1996), as well as habitat for neotropical migrant songbirds (Terborgh 1989), and of prairie potholes and coastal marshes for migratory shorebirds and waterfowl, are all now more widely appreciated. Today wetland and riparian habitats are also endowed with recreational and aesthetic values, as well as a broad array of critical "ecosystem services" (ESA 1997). The long-term trend from harvestable goods to ecosystem services, and less tangible aesthetic values, again illustrates the trend of taking a broader view.

Wetland Restoration Incentives

Prior to the Clean Water Act 404 Permit Program, limited wetland and riparian habitat restoration was accomplished by USFWS, some state resources agencies, and private conservation groups such as the Native Conservancy, Ducks Unlimited, and Trout Unlimited. Since implementation of the 404 program, mitigation responsibilities have accounted for the majority of wetland and riparian habitat restoration. Since mitigation is necessarily driven by development activities, its proponents are not always the most knowledgeable about habitat restoration, nor committed to more than regulatory success. The present regulatory framework—with mitigation negotiated on a site-by-site basis—has often resulted in small, inappropriately sited wetlands with limited functional value and at a very high cost-per-acre restored (Weinmann and Kunz 1993). Proactive mitigation banking in which small mitigation projects are combined into larger units can provide several advantages, including a vehicle for basin-level planning (Reppert 1992, ELI 1993, Green 1995[a4]).

Some private sector consultants are reporting a decline in development-related wetland mitigation projects. Perhaps the regulations are working and more projects are successfully avoiding impacts to wetlands, as well as the regulatory expense and delays they are likely to cause. Agencies with major projects that cannot avoid wetland and riparian impacts (port expansions, highway construction), are actively promoting mitigation banking and river basin planning (Reppert 1992, USDOT 1997). A 1992 Seattle workshop, *Partnerships and Opportunities in Wetland Restoration* (USEPA 1993) provided a call to arms, encouraging innovative restoration opportunities beyond mitigation requirements (LaFountain 1996).

Historically, the greatest wetland and riparian losses have resulted from agriculture, silviculture, and grazing, yet these activities were exempt from 404 Permit provisions and until 1985, the Department of Agriculture continued to subsidize wetland drainage projects (Mitsch and Gosselink 1993). This changed in 1985 with passage of the Food Security Act "Swampbuster" provisions which replaced the drainage subsidies with incentives to farmers to set aside farmed wetlands for habitat protection (Conservation Reserve Program and Wetlands Reserve Program; Robinson 1996). The USFWS estimated that 90,000 acres of wetlands were restored under this program between 1987 and 1990 (Josephson 1992). In 1996 implementation of a CRP State Enhancement Program allowed the Department of Agriculture to approve state-submitted cost sharing proposals that use CRP to

[4] This designation refers to another paper included in this Proceedings volume.

address regional enhancement of a particular estuary, river, or ecosystem. Plans are presently under review to protect and restore environmentally sensitive lands surrounding Chesapeake Bay, Maryland and in the Minnesota River Watershed, Minnesota (Minnesota 1996). The potential to apply the program to further salmon recovery in the Pacific Northwest is also being explored (EDF 1997, Searchinger 1997).

Integration

Within the last few years several of the trends outlined above have been coming together—partly in response to federal and state initiatives that encourage both ecosystem management and watershed-based management. In many cases, these changes have equally important proactive regional/watershed planning and regulatory streamlining components. Some new players are leading these efforts.

With diverse habitats, "from the desert to the sea," and a high concentration of sensitive endemic species (Dobson et al. 1997), San Diego County has embarked on a comprehensive MSCP and growth management program. The plan protects a network of core habitat preserves (mostly local, state, and federal public lands) connected together by riparian corridors. Key parcels threatened by development can be purchased to complete the preserve network, while regulatory and financial incentives will focus development away from scarce critical habitats into less sensitive areas (City/County of San Diego 1996).

Further north, the Washington State Department of Transportation (WSDOT) is also on the cutting edge, proactively planning for long-term transportation growth on a watershed basis. Using GIS technology, WSDOT can pull up stream corridors, wetlands, fish passage problems, stormwater culverts, water quality data, sensitive species distributions—even selected road-kill data to determine wildlife migratory corridors—within a chosen radius of a proposed project. Working with the regulatory agencies and local communities, WSDOT can then determine the best combination of project design features and mitigation measures to maximize the overall environmental benefits. By achieving concensus, limited funding can be leveraged into more creative, cost-effective mitigation (USDOT 1997; Shari Schaftlein, WSDOT personal communication 1997).

These examples demonstrate that incentives for restoration and enhancement of wetland and riparian habitats are now moving well beyond the regulatory focus of mitigation requirements. The need for habitat restoration is clearly increasing, but now it is in the broader landscape context of watershed management (Wear et al. 1996, Kentula 1997). Reestablishing natural wetland and riparian habitat functions and processes is increasingly seen as an appropriate response to reduce flood damage, enhance water quality, and sustain populations of sensitive species.

What Does This Mean for Wetland and Riparian Restoration?

The 90s are proving to be a tremendously exciting and challenging time for wetland and riparian research and restoration. Largely through development of landscape ecology (Forman and Godron 1986) and increasing regulatory interest in watershed management, our perspective has been fundamentally shifted from developing individual restoration projects to seeing such projects as integral parts of the larger landscape. Now more than ever we are challenged to seek understanding and solutions "outside the box." What then, are some of the ways that the trend of "taking a broader view" might alter our practice of wetland and riparian restoration?

Wetlands Occur in Landscapes

Landscape ecology, integrating ecological concepts with human activities, has introduced new perspectives to wetland and riparian studies. How are landscape elements and ecosystems distributed; how do energy, materials, or biota flow among these elements; and what changes might occur in the landscape mosaic over time (Godron and Forman 1986)?

One of the most significant changes is that we must turn our focus from *inside* the wetland (or riparian corridor) to *outside* the wetland and to its surrounding landscape (Wear et al. 1996, Kentula 1997). Application of the hydrogeomorphic classification of wetlands (Brinson 1993, 1995, 1996) helps in this regard. How does a particular wetland

site fit into the surface and groundwater hydrologic patterns and other processes of the surrounding landscape? How do other landscape functions and processes impact the wetland, and visa versa? We need to better understand what is going on in the surrounding landscape *outside* the wetland and then apply this knowledge to better understand how it impacts what is happening *inside* the wetland. As Klaus Richter recently put it so elegantly, *"You can engineer these (wetland) systems in the best way possible–but if you put them in the wrong place they are not going to work!"*

Wetlands Function in Watersheds

It is now recognized that even modest watershed development—whether urbanization or construction of forestry roads—accelerates the initiation of runoff, as well as increasing its quantity and velocity. Significant changes begin to occur when as little as 10 percent of the watershed is converted to impervious surfaces. These changes increase the magnitude and frequency of flooding events in stream systems and wetlands. This typically results in channel downcutting (which in turn lowers the adjacent water table), increased bank erosion, and thus increased sediment flux—as well as the potential for increased contamination via the runoff (Booth and Reinelt 1993, Pitt et al. 1995, May, 1996). Wetland and riparian restoration design must accommodate these potential changes (REIC 1995, King County 1996).

Failure to account for these urban runoff and stormwater impacts is responsible for the failure of many wetland and riparian enhancement programs (Shaffer and Kentula 1995[a]). Horner (1992) has indicated however, that widely used engineering models systematically underestimate runoff rates, and stormwater specifications thus need to be significantly improved. Unless we clearly understand the significance and implications of these fundamental modifications to watershed hydrology—and thus to related wetland and stream processes (Poff et al. 1997)—our habitat restoration and enhancement activities are always likely to fall short of their goals (Frissel and Nawa 1992, Chapman 1996).

Focus on Processes

Collection of descriptive physical, chemical, and biological data from reference sites—typically, pristine or minimally disturbed habitats—will always play a key role in restoration planning, as well as in monitoring project success. Identifying and understanding the natural processes that underlie and "drive" these site characteristics is even more important, however (Frenkel 1995[a], Simon et al. 1995[a], Winfield et al. 1995[a]).

Fuerstenberg, Nelson, and Blomquist (1997) documented structural changes in stream characteristics, wetland and riparian habitat, and salmon populations of Puget Sound's Green River, as the Green River Valley was increasingly modified by 140 years of settlement, agriculture, and urbanization. Increasing human modification and control of river processes resulted in reduced natural variability within the river system, less habitat diversity, and declines in macroinvertebrate and salmonid richness. Fuerstenberg et al. concluded that the specific physical, chemical, and biological structures and functions observed in the field reflected the underlying, variable processes (e.g., flood pulses, sediment flux, woody debris recruitment, nutrient transformations) operating in aquatic ecosystems. Comparison of historic structure and function with present day attributes helps define how the underlying processes operate. This, in turn, provides an ecological basis and organizing framework for regional habitat restoration and management planning. To quote Bob Fuerstenberg (King County, Seattle, WA), *"You've got to ask yourself, what are the field data telling you about the system?"*

Commenting on watershed management plans in general, Dave Caraher (U.S. Forest Service, Portland, OR) recently echoed this theme, *"We do these detailed science studies but don't take the time to understand what they really mean. You need to go sit on a rock—out in the watershed—and think about what you've learned."*

Look for Landscape Linkages

In addition to physical (e.g., hydraulic connections, groundwater and sediment flux) and chemical (e.g., nutrient cycling) processes that link wetland and riparian habitats to their surrounding landscapes, potentially important biological interactions must also be identified (Fuerstenberg 1995[a]).

Wetland and riparian vegetation stabilizes the banks of rivers, streams, and tidal channels, reduces erosion and sediment flux, and can reduce nutrient and pollutant inputs moving between uplands and adjacent aquatic habitats. The height, density, and species composition of riparian vegetation can modify shade, water temperatures, litter fall, and the contribution of large woody debris (LWD) to adjacent streams, as well as influencing the availability of insect prey. The presence of LWD adds critical habitat complexity, trapping sediment and enhancing the development of pools and riffles for fish. The size and species of LWD—western cedar versus alder, for example—influences its persistence in the system (Berg 1995[a], Rot et al. 1995[a]).

While wetland and riparian habitats support their own unique flora and fauna, they also provide support during critical life stages of large numbers of species that live much of their life cycle in nonwetland habitats (Mitsch and Gosselink 1993). Restoration programs need to consider the needs of both resident and transient species. Richter's (1995[a]) lentic breeding amphibians include examples of both permanent wetland residents and other species that must move freely between wetland ponds and adjacent upland forest habitat to prosper. Anadromous fish, such as salmonids (Spence et al. 1996)—as well as many species of marine fish and crustaceans—depend on shallow wetland habitats during larval development, but move out to sea as adults. A wide variety of migrant shorebirds, waterfowl and songbirds depend on wetland and riparian areas during their seasonal migrations. Knowledge of the biological requirements and interactions of these species can significantly influence restoration location and design.

Species associated with upland habitats can also provide critical support for wetland dependent species. Zedler (PERL 1990) provides the example of upland pollinator species—bees characteristic of adjacent coastal-scrub dominated upland—critical to successful seed production in the endangered annual salt marsh bird's beak (*Cordylanthus maritimus* ssp. *maritimus*). *The Forgotten Pollinators* (Buchmann and Nabhan 1996) documents similar more broadly based concerns.

Accessibility of restored wetland and riparian habitats to native predators—or their protection from them (as well as from introduced pest species and domestic animals)—also needs to be given careful consideration (Soule, et al. 1988, Recon 1989, Jurek 1994).

As many as possible of these various "landscape linkages" need to be identified and appropriate design features or mechanisms for their continued successful operation be incorporated into both regional wetlands plans and individual restoration projects.

Regional Planning for Restoration

Like the old way of doing business for water quality enhancement or endangered species recovery, wetland and riparian restoration projects are typically planned and implemented site-by-site, one project at a time. But what if they all follow the same planting plan, or all emphasize achievement of the same hydrologic goals? Habitat diversity begets species diversity—and community resilience to environmental purtabations—clearly this is an important function to protect.

A promising approach to address this issue is the development of regional plans that specify the variety, relative abundance, and most appropriate locations for pursuing wetland and riparian restoration projects within a particular watershed, geographic region, or management area (Harris et al. 1995[a]; MacKay 1995[a]; Stanley and Roberts, 1995[a]). Zedler (1996), concerned about dredging of shallow-water habitats to favor fish over birds, proposed this approach for scarce southern California salt marsh habitats. The California Coastal Conservancy is presently exploring implementation of such a plan. Tanner (1990) favored a similar approach for restoring wetland sites along the heavily industrialized Duwamish Channel, that flows into Puget Sound; while Shreffler and Thom (1995[a]) suggest that a landscape ecology perspective is appropriate for developing such a plan.

More Effective Decisionmaking

Planning for larger restoration programs (Scodari et al. 1995) requires more careful analysis of project goals and objectives: What underlying problems generated the need for wetland or riparian restoration? What are the greatest restoration needs? How might they be most cost-effectively accomplished?

Computerized geographic information systems (GIS) have already emerged as the preferred method for merging and displaying all of the datasets (GIS layers) necessary for comprehensive restoration planning (MacKay 1995[a], Schulman 1995[a], Shuman et al. 1995[a], Toth 1995[a], GIS World 1996, Kyle et al. 1996). The quality of the decisionmaking and implementation process, however, is as important as the quality of the science that goes into the decisions.

Making good decisions can be difficult, especially when complex habitat restoration and sensitive species problems typically involve multiple stakeholders, have conflicting objectives, require technical expertise, deal with large amounts of money, have multiple objectives, and are surrounded by public interests. Most people do not realize that when such decisions fail it is rarely due to inadequate science but commonly because of organizational issues that could be managed. Most decisions fail because of organizational problems that reflect problems with leadership, a lack of a problemsolving process or training, poor teamwork, and poor commitment. These problems can, and often do, lead to conflict (Ury and Fischer 1981, Clemens 1996).

Providing leadership in a decision process can address these problems. Sound leadership requires that a systematic problem solving process be used; that a common vision be clearly understood; that the right people are involved; and that there be a well-defined problem, schedule, and commitment from participants. Remember, that implementation of habitat restoration programs begins with the decision process (CH2M HILL 1997).

Increasing Public Involvement

Where restoration programs encompass large areas—in watershed management or multiple species conservation plans, for example—public involvement (CH2M HILL 1995) becomes very important. In an era of strong competition for limited financial resources, the public increasingly helps prioritize restoration decisions. They can also provide dedicated, cost-effective help with field surveys, restoration plantings, and long-term monitoring programs (Goeldner et al. 1995[a], Pruess and Fischer 1995[a]). These roles for public involvement are likely to increase as wetland and riparian habitat restoration moves beyond regulatory-driven mitigation projects into more incentive-based, larger scale, watershed enhancement and habitat conservation planning programs.

Conclusions

Broad recognition of the critical need for cost-effective regulatory streamlining and the need to address the accelerating impact of habitat degradation has resulted in significant changes in how environmental issues are being addressed in the 90s. Water quality enhancement initiatives, having moved from point source to include nonpoint source controls, are now evolving into the watershed-based TMDL process. Endangered species recovery initiatives have evolved from early HCPs focused on the protection of single species, into large-scale MSCPs that provide a broad-based, landscape level planning tool more suitable to achieve long-term biological and regulatory goals. At the same time, incentives for wetland and riparian restoration are moving beyond site-by-site mitigation to encourage wetland mitigation banking and basin-level restoration planning.

What do these changes mean for wetland and riparian restoration? Largely through development of landscape ecology and increasing regulatory interest in watershed management, our perspective has been fundamentally changed from developing individual wetland restoration projects to seeing such projects as integral elements of the broader landscape. We must turn our focus from inside the wetland to the surrounding landscape outside the wetland. How do landscape functions and processes impact a particular wetland site or riparian corridor, or visa versa? How have landscape modifications altered natural processes and impacted wetlands? And how can these trends be reversed?

New directions for restoration in the 90s demand a clearer understanding of the landscape setting of wetland and riparian habitats; better understanding of their functions in watersheds; a sharper focus on the natural processes that drive these systems; and identification of the critical linkages between wetlands and their surrounding landscapes. There is also a critical need to both think about wetlands and plan for their protection and restoration, on a broader scale.

Given the strong competition for limited financial resources that has characterized the 90s, planning

for larger restoration programs will require more effective decisionmaking and increased public involvement. Wetland and riparian habitat restoration is rapidly moving beyond regulatory-driven mitigation projects into more incentive-based larger scale, watershed enhancement and habitat conservation planning programs.

Literature Cited

Adam, P. 1990. Saltmarsh ecology. Cambridge Studies in Ecology. Cambridge University Press, NY. 461 pp.

Adamus, P. R. 1983. A method for wetland functional assessment. 2 Vols. U.S. Dept. of Transportation, Federal Highway Administration Rpts. FHWA-IP-82-23/24. Washington, D. C.

American Water Resources Association (AWRA). 1988. Proceedings of the Symposium on Nonpoint Pollution, 1988 - Policy, Economy, Management and Appropriate Technology. Vladimir Novotny (ed). AWRA, Minneapolis, MN.

Assoc. State Wetland Managers (ASWM). 1986. National wetlands symposium: mitigation of impacts and losses. New Orleans, LA. ASWM, Berne, NY.

Beatley, T. 1995. Preserving biodiversity through the use of habitat conservation plans. pp 35-74, In Porter, D. R. and D. A. Salvesen, Collaborative planning for wetlands and wildlife. Island Press, Washington, D. C.

Booth, D. B. and L. Reinelt. 1993. Consequences of urbanization on aquatic systems–measured effects, degradation thresholds, and corrective strategies. In USEPA (1994). Watershed '93 (op cit).

Brinson, M. M. et al. 1981. Riparian ecosystems: their ecology and status. Biological Services Program, USFWS, Washington, D. C. FWS/OBS-81/17. 151 pp.

Brinson, M. M. 1993. A hydrogeomorphic classification of wetlands. Wetlands Research Program Technical Report WRP-DE-4. U.S. Army Corps of Engineers, Washington, D. C. 79 pp.

Brinson, M. M. 1995. The HGM approach explained. National Wetlands Newsletter, Nov-Dec. '95:7-13.

Brinson, M. M. 1996. Assessing wetland functions using HGM. National Wetlands Newsletter. Jan-Feb. '96:10-16.

Buchmann, S. L. and G. P. Nabhan. 1996. The forgotten pollinators. Island Press, Washington D. C. 312 pp.

Castañares, A. A. and F. B. Phleger (eds.) 1969. Coastal lagoons, a symposium (UNAM-UNESCO), Universidad Nacional Autónoma de Mexico, Mexico City.

CH2M HILL. 1995. Public involvement strategies: a manager's handbook. Sponsored by, American Water Works Association Research Foundation, Denver, CO.

CH2M HILL, 1997. Decision facilitation handbook. CH2M HILL, Denver, CO. 173 pp.

Chabreck, R. H. (ed.) 1973. Proceedings: Coastal marsh and estuary management symposium. Louisiana State University, Baton Rouge, LA. 316 pp.

Chapman, D. W. 1996. Efficacy of structural manipulations of instream habitat in the Columbia River Basin. Rivers 5(4):279-293.

Chapman, V. 1938. Coastal movement and the development of some New England saltmarshes. Proc. Geological Assoc. of England. 49:373-384.

Chapman, V. 1960. Salt marshes and salt deserts of the world. Plant Science Monographs Interscience Publishers, NY. 392 pp. Second Supplemented Reprint Edition (1974) Verlag Von J. Cramer, Germany. 494 pp.

City/County of San Diego. 1996. Multiple species conservation program: MSCP Plan. (County Subarea Plan 1997) San Diego Association of Governments, San Diego, CA.

Clark, J., 1967. Fish and man: conflict in the Atlantic estuaries. American Littoral Society. Special Publication No. 5. 78 pp.

Clemens, R. T. 1996. Making hard decisions, an introduction to decision analysis. (2nd edition). Duxberg Press, Belmont, MA.

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

Cowardin, L. M. et al. 1979. Classification of wetlands and deep water habitats of the United States. USFWS Special Pub. FWS/OBS-79/31. Washington, D. C. 103 pp.

Crowcroft, P., 1991. Elton's ecologists. A history of the Bureau of Animal Population. University of Chicago Press, Chicago, IL. 177 pp.

Dahl, T. E. 1990. Wetland losses in the United States, 1780s to 1980s. USFWS, Washington, D. C. 21 pp.

Dahl, T. E. and C. E. Johnson. 1991. Wetlands status and trends in the conterminous United States mid-1970s to mid-1980s. USFWS, Washington, D. C. 28 pp.

Daiber, F. G. 1982. Animals of the tidal marsh. Van Nostrand Reinhold Co., NY. 422 pp.

Davis, J. H. 1940. The ecology and geologic role of mangroves in Florida. Carnegie Institute, Washington, Pub. No. 517:303-412.

Dexter, R. W. 1947. The marine communities of a tidal inlet at Cape Ann, Massachusetts: a study in bio-ecology. Ecological Monographs 20:261-294.

Dilks, D. W. et al. 1994. Development of the San Luis Obispo Creek demonstration TMDL. pp 367-372, In USEPA (1994). Watershed '93 (op cit).

Dobson, A. P., et al. 1997. Geographic distribution of endangered species in the United States. Science 275:550-553.

Ecological Society of America (ESA). 1997. Ecosystem services: benefits supplied to human societies by natural ecosystems. Issues in Ecology 2. 16 pp.

EDF/WWF. 1992. How wet is a wetland? The impacts of the proposed revisions to the Federal Wetlands Delineation Manual. Environmental Defense Fund, NY, and World Wildlife Fund, Washington, D. C. 170 pp.

Environmental Defense Fund (EDF). 1997. About the Conservation Reserve Enhancement Program: a Federal/State partnership; and, Use of CRP to further salmon recovery in the Northwest. Information Pamphlets, EDF, Washington, D. C.

Environmental Law Institute (ELI). 1993. Wetlands mitigation banking. ELI, Washington, D. C. 159 pp.

Erwin, K. L. 1996. A bibliography of wetland creation and restoration literature. Assoc. State Wetland Managers, Berne, NY. 99 pp.

Faber, P. M. et al. 1989. The ecology of Riparian habitats of the Southern California coastal region: a community profile. USFWS, Washington, D. C. 153 pp.

Federal Interagency Committee for Wetlands Delineation. Federal manual for identifying and delineating jurisdictional wetlands. USACOE, USEPA, USFWS, and USDA Soil Conservation Service, Washington, D.C. Cooperative Tech. Pub. 76 pp. (1989 manual).

Fitzpatrick, T. J. and M. F. L. Fitzpatrick. 1902. A study of the island flora of the Mississippi River near Sabula, Iowa. Plant World 5:198-201.

Forman, R. and M. Godron. 1986. Landscape Ecology. John Wiley & Sons, NY. 620 pp.

Frissell, C. A. and R. K. Nawa. 1992. Incidence and causes of physical failure of artificial habitat structures in streams of Western Oregon and Washington. North American Journal of Fisheries Management 12:182-197.

Fuerstenberg, R. R., K. Nelson, and Rob Blomquist. 1997. Ecological conditions and limitations to salmonid diversity in the Green River, Washington, USA: Structure, function and process in river ecology. King County Dept. Natural Resources, Seattle, WA (in preparation).

Garbisch, E. W., Jr. 1977. Recent and planned marsh establishment work throughout the contiguous United States–a survey and basic guidelines. Contr. Rep. D-77-3. U.S. Army Eng. Waterways Exp. Sta., Vicksburg, MS.

GIS World, Inc. 1996. GIS applications in natural resources 2. (Papers from GIS '92-'95 Symposia, Vancouver, British Columbia.) GIS World Books, Fort Collins, CO.

Good, R. E., D. F. Whigham, and R. L. Simpson (eds.) 1978. Freshwater wetlands: ecological processes and management potential. Academic Press, NY. 378 pp.

Gosselink, J. G., E. P. Odum, and R. M. Pope. 1974. The value of the tidal marsh. Center for Wetland Resources, Louisiana State University, Baton Rouge, LA. LSU-SG-74-03. 30 pp.

Greeson, P. E., J. R. Clark, and J. E. Clark. (eds.) 1978. Wetland functions and values, the state of our understanding. American Water Resources Association, Technical Publication Series. Minneapolis, MN. 674 pp.

Gwin, S. E. and M. E. Kentula. 1990, Evaluating design and verifying compliance of wetlands created under Section 404 of the Clean Water Act in Oregon. EPA/600/3-90/061. U.S. EPA, Environmental Research Laboratory, Corvallis, OR. 122 pp.

Hamilton, P. and K. B. Macdonald. 1979. Estuarine and wetland processes, with emphasis on modeling. Marine Science 11, Plenum Press, NY. 653 pp.

Hopkinson, C. S. and J. J. Vallino. 1995. The relationships among man's activities in watersheds and estuaries: a model of runoff effects on portions of estuarine community metabolism. Estuaries 18(4):598-621.

Horner, R. 1992. Constructed wetlands for storm runoff water quality control, course materials. Center for Urban Water Resources Management, Univ. Washington, Seattle, WA.

Hunter, G. J. 1991. Better trout habitat: a guide to stream restoration and management. Island Press, Washington, D. C. 320 pp.

Jackson, G. W. 1994. Farmstead assessments–a voluntary approach for identifying and implementing farmstead practices to prevent pollution. pp 697-703, In USEPA (1994). Watershed '93 (op cit).

Josephson, J. 1992. Status of wetlands. Environmental Science and Technology 26:422.

Josselyn, M. (ed.). 1982. Wetland restoration and enhancement in California. California Sea Grant College Program. Report No. T-CSGCP-007. University of California, La Jolla, CA. 110 pp.

Jurek, R. M. 1994. A bibliography of feral, stray, and free-roaming domestic cats in relation to wildlife conservation. California Dept. Fish and Game, Nongame Bird and Mammal Program Report No. 94-5. April 1994. 24 pp.

Kadlec, R. H. and R. L. Knight. 1996. Treatment Wetlands. Lewis Publishers, New York. 893 pp.

Kaiser, J. 1997. When a habitat is not a home. Science 276: 1636-1638.

Kentula, M. E. et al. 1992. An approach to improving decision making in wetland restoration and creation. USEPA, Environmental Research Laboratory, Corvallis, OR.

Kentula, M. E. 1997. A step toward a landscape approach in riparian restoration. (Guest editor) Special Supplement, Riparian Restoration. Restoration Ecology 5(45): 1-121

Kibby, H. V., J. L. Gallagher, and W. D. Sanville. 1980. Field guide to evaluate net primary production of wetlands. Environmental Research Lab., USEPA, Corvallis, OR. EPA-600/8-80-037. 55 pp.

King County. 1996. Wetlands and urbanization: implications for the future. Puget Sound Wetlands and Stormwater Management Research Program, King County, Seattle, WA.

Kusler, J. and M. Kentula. (eds.) 1990. Wetland creation and restoration: the status of the science. Island Press, Washington, D. C. 594 pp.

Kyle, H. M., M. G. Schmidt, and K. P. Lertzman. 1996. A spatially dynamic riparian buffer model: conservation on the edge. pp. 341-346, In GIS '35- Ninth Annual Symposium on Geographic Information Systems, Vancouver, British Columbia, Canada.

LaFountain, C. 1996. The most for the money. National Wetlands Newsletter, Environmental Law Institute, Washington, D. C. 18(3):26-27.

Lauff, G. H. (ed.) 1967. Estuaries. Amer. Assoc. Advancement of Science, Washington, D. C. Pub. 83. 757 pp.

Lindeman, R. L. 1942. The trophic-dynamic aspect of ecology. Ecology 23:399-418.

Linduska, J. P. and A. L. Nelson (eds.). 1964. Waterfowl Tomorrow. U.S. Fish and Wildlife Service, Government Printing Office, Washington, D. C. 770 pp.

Luoma, J. R. 1998. Habitat conservation plans: compromise or capitulation? Audubon January-February 1998, 36-43.

MacArthur, R. H., and E. O. Wilson. 1967. The theory of island biogeography. Princeton University Press, Princeton.

Macdonald, K. B. 1976. The natural resources of Mugu Lagoon. Coastal Wetland Series No. 17. Calif. Dept. Fish and Game/USFWS. Sacramento, CA. 119 pp.

Malanson, G. P. 1993. Riparian landscapes. Cambridge University Press, Cambridge, England. 296 pp.

Mann, C. and M. Plummer. 1997. Qualified thumbs up for habitat plan science. Science 278:2052-2053.

Marble, A. D. 1992. A guide to wetland functional design. Lewis Publishers, Boca Raton, FL.

Marcus, M. L. 1982. Wetlands policy assessment: California case study. Prepared for Office of Technology Assessment, Washington, D. C. by ESA/Madrone Consultants. Novato, CA.

Mason, H. L. 1957. A flora of the marshes of California. University of California Press, Berkeley, CA. 878 pp.

May, C. W. 1996. Assessment of cumulative effects of urbanization on small streams in the Puget Sound Lowland ecoregion: implications for salmonid resource management. Ph.D. Dissertation, Univ. Washington, Seattle, WA. 383 pp.

McCormick, J., R. R. Grant, and R. Patrick (eds.) 1970. Two studies of Tinicum Marsh, Delaware and Philadelphia Counties, PA. The Conservation Foundation. Washington, D. C. 180 pp.

McCormick, Taylor & Assoc. Inc. 1995. Cost savings of expedited NEPA/404 approvals: an analysis of three project in Pennsylvania. Pennsylvania Dept. Transportation and Federal Highway Administration, Washington, D. C. 13 pp.

Minnesota, State of. 1996. Minnesota River watershed conservation reserve enhancement program. Submitted to U.S. Dept. of Agriculture. State of Minnesota (10/25/96). 48 pp.

Mitsch, W. J. and J. G. Gosselink. 1993. Wetlands. (Second Edition) Van Nostrand Reinhold Co., NY. 722 pp.

Montanari, J. H. and J. H. Kusler (eds.) 1977. Proceedings of the National Wetland Protection Symposium, Reston, VA, June 1977. U.S. Forest Service, Washington, D.C. FWS/DBS-78/97. 255 pp.

National Research Council (NRC). 1995. Wetlands: characteristics and boundaries. Committee on Characterization of Wetlands, National Academy of Sciences, Washington, D. C.

Nesbit, D. M. 1885. Tide marshes of the United States. USDA Misc. Special Report, No. 7, Washington, D. C. 259 pp.

Nixon, S. W. 1980. Between coastal marshes and coastal waters–A review of twenty years of speculation and research on the role of salt-marshes in estuarine productivity and water chemistry. pp. 437-525, In Hamilton, P. and K. B. Macdonald, Estuarine and wetland processes, with emphasis on modeling. Marine Science, Vol. 11. Plenum Press, NY.

Nixon, S. 1982. The ecology of New England high salt marshes: a community profile. Biological Services Program, USFWS, Washington, D. C. FWS/OBS-81/55. 71 pp.

Onuf, C. P. et al. 1979. An analysis of the values of Central and Southern California coastal wetlands. pp 186-199, In Greeson P.E., et al (eds.) Wetland functions and values: the state of our understanding (op cit).

Payne, N. F. 1992. Techniques for wildlife habitat management of wetlands. McGraw-Hill, NY.

Penfound, W. T. 1956. Primary production of vascular aquatic plants. Limnology and Oceanography 1:92-101.

PERL (Pacific Estuarine Research Laboratory). 1990. A manual for assessing restored and natural coastal wetlands with examples from southern California. Calif. Sea Grant Rep. T-CSGCP-021. San Diego State University, CA.

Pitt, R., et al. 1995. Urban stormwater toxic pollutants: assessment, services, and treatability. Water Environment Research 67(3): 260-275.

Poff, N. L. et al. 1997. The natural flow regime: a paradigm for river conservation and restoration. BioScience 47(11):769-784.

Pomeroy, L. R. 1958. Productivity of algae in salt marshes. pp. 88-95 In Ragotzkie, R. A. (ed) 1959, op cit.

Pulliam, H. R. and B. Babbit. 1997. Science and the protection of endangered species. Science 275:499-500.

Purer, E. A. 1942. Plant ecology of the coastal salt marshlands of San Diego County, California. Ecological Monographs 12:82-111.

Race, M. S. 1985. Critique of present wetlands mitigation policies in the United States based on an analysis of past restoration projects in San Francisco Bay. Environ. Management 9:71-82.

Ragotzkie, R. A. (ed). 1959. Salt marsh conference proceedings. The Marine Institute, University of Georgia, Sapelo Island, Georgia, March 25-28, 1958. Athens, GA (April 1959). 133 pp.

Recon (Regional Environmental Consultants). 1989. Comprehensive species management plan for the least Bell's vireo. Prepared for San Diego Association of Governments by Recon, San Diego, CA. 226 pp.

Regional Interagency Executive Committee (RIEC). 1995. Ecosystem analysis at the watershed scale: federal guide for watershed analysis. Revised August 1995, Version 2.2. Regional Ecosystem Office, Portland, OR. 26 pp.

Reppert, R. 1992. Fundamentals of wetlands mitigation banking. Concept Paper. U.S. Army Corps of Engineers, Institute for Water Resources. Report 92-WMB-1.

Robinson, A.Y. 1996. A wetland-friendly Farm Bill? National Wetlands Newsletter, Environmental Law Institute, Washington, D.C. 18(3):1,28-30.

Rohlf, D. J. 1989. The endangered species act: a guide to its protections and implementation. Stanford Environmental Law Society, Stanford Law School, Stanford, CA. 207 pp.

Scodari, P.F., et al. 1995. Prototype information tree for environmental restoration plan formulation and cost estimation. USACOE, Inst. for Water Resources, Alexandria, VA. IWR Report 95-R-3. 58 pp.

Scott, D. A. 1982. Managing wetlands and their birds: a manual of wetland and waterfowl management. International Waterfowl Research Bureau, Slimbridge, Glos., England. 368 pp.

Searchinger, T. D. 1997. Senior Attorney, Environmental Defense Fund, Washington D. C. Personal communication, October 1997.

Seneca, E. D., W. W. Woodhouse, Jr., and S. W. Broome. 1975. Saltwater marsh creation, pp 427-437. In L. E. Cronin (ed.), Estuarine Research Vol. II. Geology and Engineering, Academic Press, NY.

Shaler, N. S. 1885. Preliminary report on sea-coast swamps of the eastern United States. U.S. Geological Survey, 6th Annual Report. pp. 359-368.

Shapiro and Assoc., Inc. 1978. Snohomish Estuary wetlands study. Vols. I-IV. USACOE, Seattle District, Seattle, WA.

Shaw, S. P. and C. G. Fredine. 1956. Wetlands of the United States, their extent, and their value for waterfowl and other wildlife. USFWS, Circular 39, Washington, D. C. 67 pp.

Shilling, F. 1997. Do habitat conservation plans protect endangered species? Science 276:1662-1663.

Simenstad, C. A., and R. M. Thom. 1996. Functional equivalency trajectories of the restored Gog-Le-Hi-Te estuarine wetland. Ecological Applications 6:38-56.

Smalley, A. E. 1959. The role of two invertebrate populations Littorina irrorata and Orchelimum fidicinium in the energy flow of a salt marsh ecosystem. Ph.D. Dissertation, Univ. of Georgia, Athens, GA. 126 pp.

Smith, L. M., R. L. Pederson and R. M. Kaminski. 1989. Habitat management for migrating and wintering waterfowl in North America. Texas Tech University Press, Lubbock, TX. 560 pp.

Smith, R. D. et al. 1995. An approach for assessing wetland functions using hydrogeomorphic classification, reference wetlands, and functional indices. Wetlands Research Program Tech. Rpt. WRP-DE-9. USACOE, Waterways Experiment Station, Vicksburg, MS.

Soulé, M. E., et al. 1988. Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands. Conservation Biology 2:75-92.

Spence, B. C. et al. 1996. An ecosystem approach to salmonid conservation. TR-4501-96-6057. ManTech Environmental Research Services Corp., Corvallis, Oregon. (Available from National Marine Fisheries Services, Portland, Oregon.) 356 pp.

Strickland, R., (ed.). 1986. Wetland functions, rehabilitation and creation in the Pacific Northwest: the state of our understanding. Proceedings of Conference, April 30–May 2, 1986. Port Townsend, WA. Pub. No. 86-14. Washington State Department of Ecology, Olympia, WA.

Tanner, C. D. 1990. Mitigation planning on an estuary-wide scale: an alternative to case-by-case wetland mitigation policy. Masters thesis, University of Washington. Seattle, WA. 89 pp.

Teal, J. M. 1958. Energy flow in the salt marsh ecosystem. pp. 101-107, In Ragotzkie, R. A. (ed) 1959, op cit.

Teal, J. M. 1962. Energy flow in the salt marsh ecosystem of Georgia. Ecology 43:614-624.

Terborgh, J. 1989. Where have all the birds gone? Princeton University Press, Princeton, NJ. 207 pp.

Thayer, G. W. (ed.) 1992. Restoring the nation's marine environment. Maryland SeaGrant Book, College Park, MD. 716 pp.

Thompson, K. 1961. Riparian forests of the Sacramento Valley, California. Annals of Assoc. Amer. Geographers 51:294-315.

Transeau, E. N. 1903. On the geographic distribution and ecological relations of the bog plant societies of northern North America. Botanical Gazette 36:401-420.

Turner, L. M. 1936. Ecological studies in the Lower Illinois River Valley. Botanical Gazette 97:689-727.

Turner, R. E. 1977. Intertidal vegetation and commercial yields of Penaeid Shrimp. Trans. Amer. Fisheries Society 106:411-416.

U.S. Fish and Wildlife Service. 1970. National estuary study. Washington, D. C. 3 vols.

U.S. Fish and Wildlife Service. 1980. California least term. FWS/OBS-80/01.20. Biological Services Program, USFWS, Washington, D. C.

U.S. Fish and Wildlife Service. 1985. Coachella Valley fringe-toed lizard Recovery Plan. USFWS, Portland, OR.

U.S. Fish and Wildlife Service and National Marine Fisheries Services (NMFS). 1996. Endangered species habitat conservation planning handbook. USFWS/NMFS, Washington, D. C.

U.S. Department of Transportation (USDOT). 1997. Integrating watershed management and transportation planning. National Teleconference Series, Program No. TC-12. Center for Transportation and the Environment, North Carolina State University, Raleigh, NC (11/7/97).

U.S. General Accounting Office. 1994. Highway planning: agencies are attempting to expedite environmental reviews, but barriers remain. USGAO, Washington, D. C. GAO/RCED-94-211. 30 pp.

Ury, W. and R. Fischer. 1981. Getting to yes: negotiating an agreement without giving in. Houghton Mifflin Co., NY. (Penguin Books 1983). 161 pp.

USACOE. 1976. Alsea wetlands review. USACOE, Portland District, Portland, OR. 345 pp.

USACOE. 1979. Wetland Values: concepts and methods for wetlands evaluation. USACOE, Inst. For Water Resources, Research Rpt. 79-RI, Ft. Belvoir, VA. 109 pp.

USACOE. 1987. USACOE Wetlands Delineation Manual. Environmental Laboratory, USACOE, Waterways Experiment Station, Vicksburg, MS. Tech. Rpt. 4-87-1. (1987 Manual).

USEPA. 1993. Constructed wetlands for wastewater treatment and wildlife habitat: USEPA, Office of Wastewater Management, Municipal Technology Branch, Washington, D. C. EPA 832-R-93-005. 174 pp.

USEPA. 1993. Guidance specifying management measures for sources of nonpoint pollution in coastal waters. Report 840-B-92-002. USEPA, Office of Water, Washington, D. C.

USEPA. 1993. Partnerships and opportunities in wetland restoration. Workshop Proceedings,

Seattle, WA. April 1992. EPA 910/R-94-003. USEPA, Seattle, WA.

USEPA. 1994. Watershed '93: a national conference on watershed management. EPA 840/R-94-002 USEPA, Washington, D. C.

Vanbianchi, R. et al. 1994. A citizen's guide to wetland restoration. Approaches to restoring vegetation communities and wildlife habitat structure in freshwater wetland systems. Publication 910/R-94-006. USEPA, Region 10, Seattle, WA. 71 pp.

Warner, R. E. and K. M. Hendrix. 1984. California riparian systems: ecology, conservation and productive management. University of California Press, Berkeley, CA. 1035 pp.

Washington State Dept. of Ecology. 1992. Stormwater management manual for the Puget Sound Basin. Pub. #91-75. Washington State Dept. of Ecology, Olympia, WA.

Wear, D. N., M. G. Turner, and R. O. Flamm. 1996. Ecosystem management with multiple owners: landscape dynamics in a southern Appalachian watershed. Ecological Applications 6(4):1173-1188.

Weinmann, F. and K. Kunz. 1993. Wetlands: infusion of restoration into the management formula. pp 1-17, In USEPA, Partnership and opportunities in wetland restoration. Workshop Proceedings, Seattle, WA. April 1992. EPA 310/R-94-003. USEPA, Seattle, WA.

Western Forestry and Conservation Association (WFCA). 1997. Watershed analysis and the Total Maximum Daily Load allocation process. WFCA, Portland, OR.

Wilson, E. O., 1994. Naturalist. Island Press, Shearwater Books, Washington, D. C. (Warner Books edition 1995). 380 pp.

Wren, C. D. et al. 1997. Wildlife and contaminants in constructed wetlands and stormwater ponds. Environment Canada. Tech. Rpt. Series No. 269. Canadian Wildlife Service, Ontario Region. 109 pp.

Wright, J. O. 1907. Swamp and overflow lands in the United States. U.S. Dept. of Agriculture Circular No. 76. Washington, D.C.

Yapp, R. H. et al. 1917. The salt marshes of the Dovey Estuary. Journal of Ecology 5:65-103.

Zedler, J. B. 1982. The ecology of Southern California coastal salt marshes: a community profile. Biological Services Program, USFWS, Washington, D. C. FWS/OBS-81/54. 111 pp.

Zedler, J. B. 1984. Salt marsh restoration: A guidebook for Southern California. California Sea Grant College Program. Report No. 7-CSGCP-009. La Jolla, California.

Zedler, J. B. 1988. Salt marsh restoration: lessons from California, pp 123-238. In J. Cairns (ed.), Management for rehabilitation and enhancement of ecosystems. CRC Press, Boca Raton, FL.

Zedler, J. B. 1996. Coastal mitigation in southern California: the need for a regional restoration strategy. Ecological Applications 6:84-93.

Zedler, J. B. 1996a. Tidal wetland restoration: a scientific perspective and Southern California focus. Calif. Sea Grant College System. Univ. of California, La Jolla, CA. Report No. T-038. 129 pp.

Zedler, J. B. (ed.) 1996b. Forum: Wetland Mitigation. Ecological Applications 6(1):33-101.

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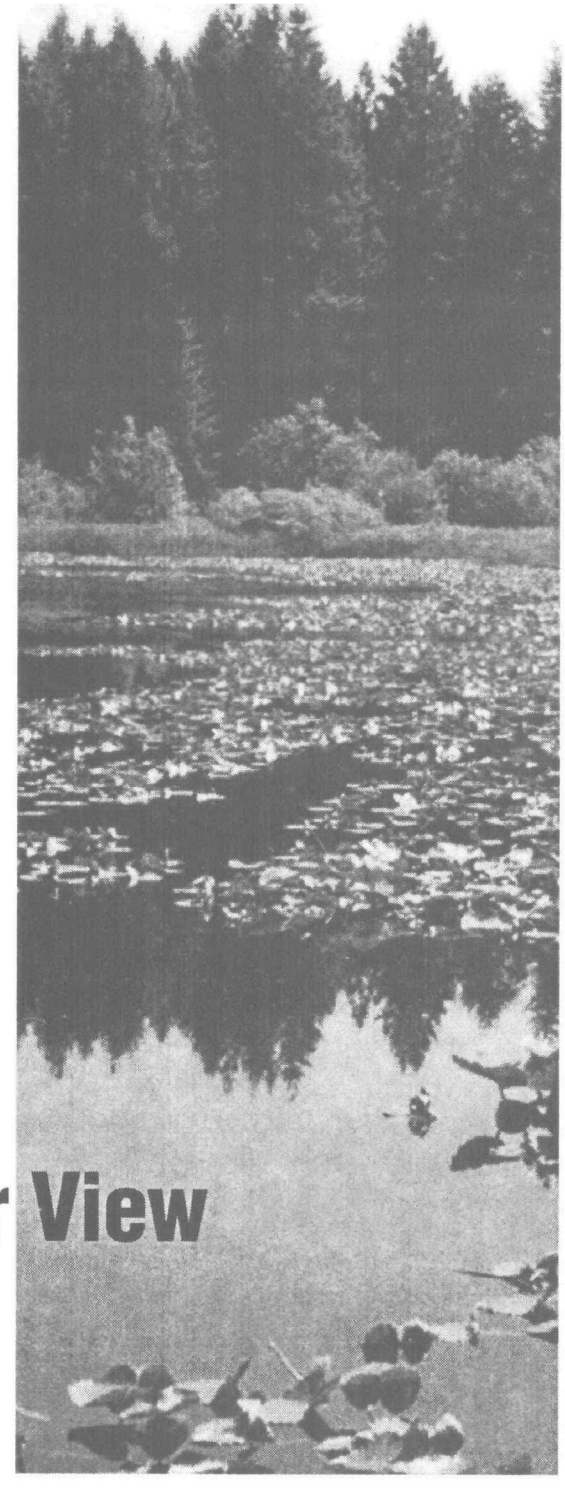
Taking A Broader View



"You can engineer these (wetland) systems in the best way possible–but if you put them in the wrong place they're not going to work."

Klaus Richter,
Wetland Ecologist
Seattle–1997

An Integrated Wetland Management Program for the Snohomish Estuary

Stephen Stanley[1]
Paul Roberts[1]

Abstract: **The Snohomish Estuary Wetland Integration Plan (SEWIP) is an ecologically based wetlands management plan, which provides the basis for a coordinated regulatory program for the protection, restoration, and enhancement of the Snohomish Estuary. The 12,000-acre Snohomish Estuary is located on the Puget Sound 25 miles north of Seattle in Snohomish County (Washington). The program is an adaptive management process involving technical experts from the Washington Departments of Ecology and Fish and Wildlife, in addition to the Corps of Engineers, Environmental Protection Agency, and local jurisdictions including the City of Everett, the Tulalip Tribe and Snohomish County. The 12,000 acres of the estuary were inventoried and assessed using the Indicator Value Assessment (IVA) protocol[2]. The IVA was designed to measure the performance of 16 wetland functions (e.g., anadromous fish, overwintering birds, mammals, sediment retention, etc.) based on a series of "indicator questions" developed by a team of wetland scientists (ecologists, wildlife biologists, fisheries biologists) and hydrologists familiar with the estuary. For planning purposes, it was assumed that the entire study area was wetland (not jurisdictionally delineated), and it was mapped as 367 individual wetland complexes on the basis of common hydrological boundaries. Wetland complexes were categorized as Group 1 (highest ranked), Group 2 and Group 3 (lowest ranked) based on assessment scores and their relative rankings. Analysis of the assessment data indicated that 86 percent of the wetland complexes within the estuary have a relatively high level of performance of the Wildlife, Fisheries and Water Quality Improvement Functions. It was concluded that the estuary is functioning as an integral ecosystem and has a high potential for successful restoration. Restoration sites have been identified with a priority given to tidal restoration. The conceptual management plan ensures the continuation of agriculture, primarily in one of the seven Ecological Management Units (1.0), with restoration efforts focusing on non-regulatory incentive measures within this unit, including leasing of edges of fields for "habitat crops" in order to provide wildlife corridors between larger habitat patches. Ecological Management Unit 2.1, an area of wetlands with low performance of assessed wetland functions, is being considered for potential future development. Phase II of the plan will include preparation of an alternatives analysis, compensation manual, and environmental assessment.**

Introduction

The "Snohomish Estuary Wetland Integration Plan," (SEWIP) was prepared for 11,623 acres of the Snohomish Estuary located on the Puget Sound in the west-central portion of Snohomish County of Washington State (Figure 1). The plan was completed in April 1995 and was funded by the Department of Ecology and the Environmental Protection Agency (EPA) as part of the State Wetland Integration Strategy (SWIS).

The plan consists of a wetland management program which provides for the protection and restoration of the most important wetland ecosystems within the estuary and a wetland compensation process designed to streamline the local, state and federal wetland permitting process. When an alternatives analysis for the plan is completed in 1996, the City and County will apply to the U.S. Army Corps of Engineers (Corps of Engineers) for a

[1]City of Everett Planning and Community Development, 2930 Weetmore Ave. Suite 100, Everett, Washington 98201-4044.
[2]Editor's Note–The IVA protocol is further explained in Thomas Hruby's paper this volume.

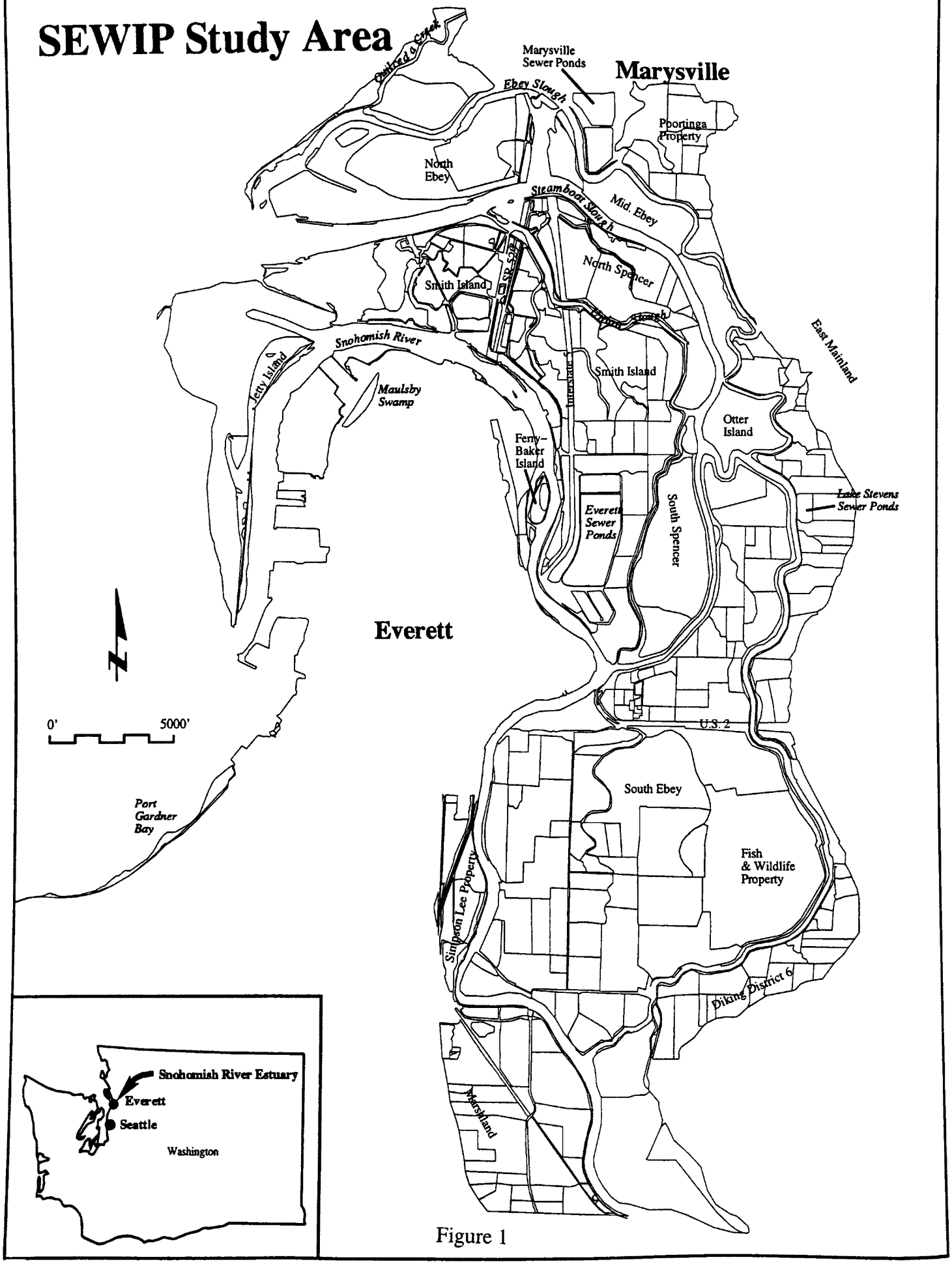

SEWIP Study Area
Quilceda Creek
Marysville Sewer Ponds
Marysville
Ebey Slough
Poorinqa Property
North Ebey
Steamboat Slough
Mid. Ebey
North Spencer
Smith Island
Snohomish River
Jetty Island
Maulsby Swamp
Interstate
Smith Island
East Mainland
Otter Island
Ferry-Baker Island
Everett Sewer Ponds
South Spencer
Lake Stevens Sewer Ponds
Everett
N
0'
5000'
U.S. 2
South Ebey
Fish & Wildlife Property
Simpson Lee Property
Diking District 6
Marshland
Port Gardner Bay
Snohomish River Estuary
Everett
Seattle
Washington

Figure 1

"blanket permit" through either a Letter of Permission (LOP) or Programmatic Permit. This will allow both local governments to issue permits for some types of wetland alteration (including enhancement and restoration) within wetland complexes with a low ranking. Alterations would only be permitted if conducted within the conditions of the overall permit issued by the Corps of Engineers. This process should provide for a substantial savings in time and money relative to the current permitting process.

Methods

In order to develop a regional wetland management plan that is scientifically based it is necessary to: 1) conduct an inventory of the number and type of wetland resources; and 2) conduct an assessment of the ecological functions of those wetlands. The methods for conducting wetland inventories are relatively well established because they are based on plant species, soil types and the hydrological regime present. Numerous comprehensive resources are available which list and identify wetland plants and soils, and assist in the identification of different hydrological regimes. These resources include national and regional plant lists which identify plants by species and their probability of occurrence in wetlands; soil surveys with maps from the Natural Resource Conservation Commission (formerly Soil Conservation Service) identifying soil makeup and whether it is hydric; and the 1987 and 1989 Federal Interagency Manuals for Identifying and Delineating Jurisdictional Wetlands.

Assessing the performance of wetland functions is a much more difficult undertaking. Functions are defined as the physical, chemical and biological processes or attributes that contribute to the self-maintenance of wetland ecosystems. To assign a "quantifiable" numeric value to wetland functions for a specific wetland requires an intricate knowledge of the interaction of the physical, chemical and biological components within that wetland's ecosystem. Typically this requires years of study to identify the habitat requirements and life forms of species within the ecosystem and to identify relationships between the species and their environment.

Ideally, with this type of in-depth knowledge a wetland ecologist would have to only identify the type of habitat (including basic vegetation structure, hydrology, landscape features and connections, and buffer type) in order to know how well wetland functions are being performed. Yet, this type of detailed knowledge of the "quantitative" performance of wetland functions is more than a decade away (Hruby et al. 1994). Further, the existing habitat assessment models, such as the Wetland Evaluation Technique (WET), the U.S. Fish and Wildlife Service's Habitat Evaluation Procedures (HEP) and the Reppert method (Reppert et al. 1979) do not specifically measure the performance of functions, or allow for the comparison of wetland performance and value between individual wetlands.

Nevertheless, there is sufficient knowledge regarding performance of a wetland function at a general level when certain basic indicators are present. For example, a wetland with a restricted outlet performs the "water quality improvement" function better than a wetland with an unrestricted outlet; a tidal wetland with dendritic channels performs the "fishery" function better than a tidal wetland without such channels. The SEWIP study selected a habitat evaluation method which measures performance at this more general "qualitative" level and ranks wetlands in a "semi-quantitative" manner.

In January of 1994, a Technical Advisory Committee was formed to assist in the development of the Inventory and Assessment methodology for the SEWIP effort. The committee consisted of wetland and resource scientists from the Corps of Engineers, EPA, Department of Ecology, Washington Department of Fisheries and Wildlife, Snohomish County, City of Everett, and local wetland consulting firms.

Inventory of Wetlands

The initial task of the Technical Advisory Committee was to determine how the estuary wetlands would be mapped and inventoried.

1. Mapping Units. The committee first dealt with whether the wetlands should be mapped according to the delineation methodology of the 1987 Federal

Interagency Manual on Wetland Delineation. It was determined that it would be too time consuming to field delineate and map wetlands in this manner. Though large portions of the estuary are diked and drained for farming, the majority of the soils are still hydric, and are saturated during the winter and early spring months. Because the majority of the diked portions of the estuary cannot be drained due to high river flows (e.g., in adjacent sloughs) prior to March, hydric soils are more than likely saturated to the surface for the requisite one week period during the growing season.

In light of this information, the Committee determined that all areas within the estuary should be mapped (unless all three indicators were not present–plants, soils and hydrology and the areas was clearly upland) on the basis of common hydrology. Each hydrological unit would be called a "wetland complex" and no attempt would be made to determine actual wetland boundaries within the complex.

2. Wetland Complex Inventories. Plant inventories for each complex were conducted with the estimated percent of dominant plant species for each class (emergent, scrub-shrub, and forested class) recorded, in addition to buffer condition, and wildlife observations.

3. GIS Mapping. Wetland complexes were base-mapped on computer generated copies of Corps of Engineers color-infrared aerial photos (8/18/93, scale 1:2000) according to common hydrological boundaries. Black and white aerial photos (2/24/91, scale 1:600) were also used to assist in the mapping of units. The COE color infrared aerials were scanned directly into the City's Geographic Information System and subsequently printed out to use for mapping in the field. This allowed for digitization of wetland complex boundaries directly off of the field maps.

4. Habitat Classification System for Field Mapping. Field teams used a modified Cowardin System (Cowardin et al. 1979) to describe wetland habitat within the estuary. The estuarine intertidal and palustrine systems were primarily used with the classes being limited to unconsolidated shore, aquatic bed, emergent, scrub-shrub, and forested. Dominant classes were based on a minimum 30 percent areal coverage of the wetland complex being inventoried and assessed. Secondary classes were based on a minimum 10 percent areal coverage of the wetland complex.

Habitat Assessment Model

In developing the habitat assessment model it was necessary to first identify those functions that were ecologically critical to the functioning of the estuary and socially important to humans. The Technical Advisory Committee selected 17 functions in the estuary (Table 1).

Table 1. Functions for vegetated wetlands (+6 MLLW)[1]

Social Significance	Stabilization of Shoreline and Channels
	Access to Transportation Corridors
	Recreation
	Priority Species Habitat
	Aesthetic Value
Water Quality Improvement	Sediment Stabilization
	Sediment Toxicant Retention
	Nutrient Retention and Transformation
	Invertebrates
Export of Primary Productivity	
Fish Habitat	(includes rearing, feeding, migration, and shallow water refuge functions)
	non-anadromous
	anadromous
Bird Habitat	migratory
	overwintering
	nesting
Other Species Habitat	Herptiles
	Mammals

[1]The functions selected for Sand/Mudflat and Subtidal Wetlands (-15 to +6 MLLW) where similar to those for vegetated wetlands except that herptiles, mammals, and primary productivity were dropped and epibenthic and infaunal invertebrates were added.

In reviewing several habitat assessment models, including the Oregon Freshwater Assessment and

Wetland Evaluation Technique (WET), it was determined that the Indicator Value Assessment (IVA) protocol would be used. The primary advantage of this model relative to others, is that it can be designed specifically for the wetlands being evaluated and it allows comparison of functional performance of wetlands within a defined regional area.

The IVA model is based on the assumption that a wetland with a greater number of indicators for a particular function is performing that function better than another study area wetland with fewer indicators present. Numeric scores are assigned to all wetlands within the study area on the basis of the number of indicators present for each wetland, and the wetlands are ranked according to their score from 1 to 100.

To determine the number and type of indicators present within an evaluated wetland, the committee devised a series of "indicator questions." These questions, when answered in the field, "indicate" the presence of biological, chemical and physical conditions that are important to the functioning of a wetland ecosystem. The model also includes questions pertaining to "social functions" that are important to humans such as recreation (includes shoreline access), navigable waterways for coastal-dependent industry, hunting, fishing, boating, and wildlife observation.

Based on their scientific knowledge of coastal wetlands within Washington and of the estuary, the Technical Advisory Committee assigned individual numeric values to each function of every indicator question. The assigned values were assigned depending on whether the indicator question was: very strongly associated (3); strongly associated (2); or weakly associated (1) with the function being assessed. A multiplier or fraction was applied when an indicator resulted in significantly higher values (e.g., the presence of dendritic channels indicating greatly improved fish habitat), or negative values (e.g., presence of cattle grazing).

The habitat questions are separated into categories of wetland "function" such as Fish, Birds and Other Species, Primary Productivity, and Water Quality Improvement. The model also contains questions which subtract from the overall habitat value if negative features are present on the wetland complex being evaluated, such as farming, drainage ditches, and filling activities. This evaluation model allows each parcel (as defined by a common hydrological boundary, such as ditches, dikes or sloughs) to be assigned a numeric value and, therefore, to be compared to all other wetland complexes within the estuary. This will allow the regulatory agencies to determine where the highest and lowest performance wetlands are located and provide a basis for offsite mitigation planning.

Two teams of biologists and resource specialists (wetland ecologist, fishery biologist, wildlife biologist, water quality scientist) spent five months from May 1994 to September 1994 mapping and assessing 367 wetland complexes within the study area. A List of Assumptions for applying the model was reviewed by a subcommittee of the technical advisory committee and any necessary changes made. A rigorous review of all data for consistency with the List of Assumptions was then conducted by the team members over a two-month period.

Results

Analysis of the assessment data indicate that 86 percent of the wetland complexes within the estuary have a relatively high level of performance of the Wildlife and Water Quality Improvement Functions. As such, the estuary is functioning as an integral ecosystem and has high potential for successful restoration given the presence of hydric soils, hydrology, and ample adjacent "pristine" sites which could serve as sources for plant and animal colonization.

Composite scores for the Water Quality Improvement and Wildlife Attributes for 367 wetlands within the Snohomish Estuary are presented in Figure 2. Based on the individual frequency distributions of data for each attribute, wetland complexes were separated into "Group 1, 2, or 3" categories.

The individual functions for the Social Significance functions were not combined for a final overall categorization because of the dissimilar nature of "transportation access" and "recreation/ aesthetics" functions.

Definitions for the three wetland groups are as follows: **Group 1** - Wetland complex is very

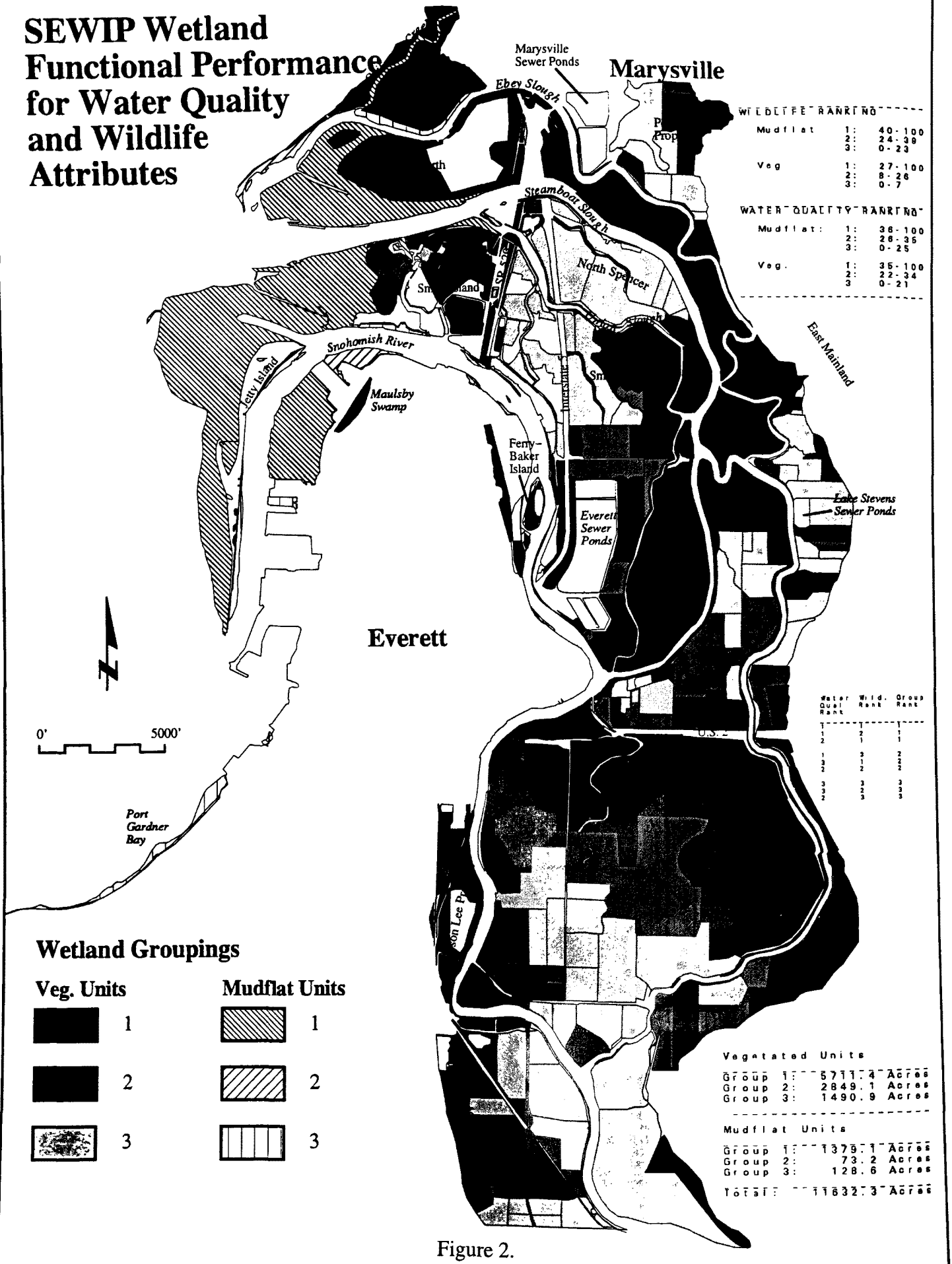

SEWIP Wetland Functional Performance for Water Quality and Wildlife Attributes
Marysville Sewer Ponds
Marysville
Ebey Slough
Steamboat Slough
North Spencer
Snohomish River
Ferry Island
Maulsby Swamp
Interstate
East Mainland
Ferry-Baker Island
Everett Sewer Ponds
Lake Stevens Sewer Ponds
Everett
0'
5000'
Port Gardner Bay
WILDLIFE RANKING
Mudflat 1: 40-100
2: 24-39
3: 0-23
Veg 1: 27-100
2: 8-26
3: 0-7
WATER QUALITY RANKING
Mudflat: 1: 36-100
2: 26-35
3: 0-25
Veg. 1: 35-100
2: 22-34
3 0-21
Water Qual Rank
Wild. Rank
Group Rank
Vegetated Units
Group 1: 5711.4 Acres
Group 2: 2849.1 Acres
Group 3: 1490.9 Acres
Mudflat Units
Group 1: 1379.1 Acres
Group 2: 73.2 Acres
Group 3: 128.6 Acres
Total: 11632.3 Acres
Wetland Groupings
Veg. Units
1
2
3
Mudflat Units
1
2
3

Figure 2.

strongly associated with the performance of estuary functions; **Group 2** - Wetland complex is strongly associated with the performance of estuary functions; and **Group 3** - Wetland complex is weakly associated with performance of estuary functions but may have a high potential for restoration.

Wetland rankings **should not be compared between** mudflat (non-vegetated) wetland complexes and vegetated wetland complexes due to the different indicator assessment models used for each wetland type (Table 1).

A total of 11,632 acres of non-vegetated and vegetated wetland were classified. A breakdown of the wetland categorization by acreage and the ranking scheme is included in Figure 2. In general, vegetated wetland complexes with ranking scores greater than 26 for the Wildlife function and 34 for the Water Quality Improvement functions were placed in Group 1; those with scores from 8 to 26 for Wildlife and 22 to 34 for Water Quality Improvement, were placed in Group 2; and those with scores less than 8 for Wildlife and less than 22 for Water Quality Improvement, were placed in Group 3.

Over 60 percent (7090 acres) of the estuary's wetland complexes were classified within Group 1 for both the Water Quality Improvement and Wildlife Attributes. This area represents an almost continuous band of Group 1 wetland complexes stretching from the western mudflats along Ebey and Steamboat Sloughs, through South Spencer Island and into the north portion (south of Highway 2) of South Ebey Island and across to Diking District 6 (Figure 2).

Edge units along the north portion of South Ebey Island appear to play an important role in connecting Group 1 wetland complexes to the north with similar areas south of Highway 2 on South Ebey Island.

Group 2 Wetland Complexes (2922 acres) comprise over 25 percent of estuary study area and are primarily in the central portion of the estuary adjacent to Group 1 wetlands. Group 3 Wetland Complexes represent only 14 percent (1619 acres) of the estuary study area and are entirely limited to heavily cultivated, grazed or mowed agricultural areas.

Both the presence of a significant area of Group 1 wetland complexes throughout the estuary and concentration of Group 2 wetland complexes adjacent to Group 1 wetland complexes, indicates that the estuary is functioning at a relatively high level of performance for the functions assessed and is still an intact estuary ecosystem that has a high potential for enhancement and restoration. Group 3 wetlands have a high potential for restoration because the existing agricultural practices have not permanently degraded the former estuarine habitat and do not prevent the return of these lands to full tidal influence.

Discussion - Management Plan Development

In developing a management plan the technical committee addressed the following seven tasks in sequence: 1) identify and rank the restoration and enhancement goals; 2) identify restoration and assessment sites; 3) identify specific restoration actions for each site, based on site visits by technical committee; 4) rank restoration and enhancement sites based on overall estuary goals, technical, and social feasibility; 5) calculate potential "compensation credits" for restoration and enhancement sites by re-scoring these sites with IVA model based on identified restoration and enhancement actions (i.e., #3); 6) identify development footprint based on location of lowest performance wetlands for habitat functions and highest performance for transportation and water access functions, and determine if "impact debits" generated can be offset by "compensation credits"; and 7) development compensation mechanism for "development footprint" impacts, including identification of compensation ratios.

Identification of Restoration and Enhancement Goals (Step 1)

The primary impact to the approximately 12,000 acres of wetlands within the Snohomish Estuary has been due to diking and draining for conversion to agricultural production. This has resulted in an estimated 74.4 percent (Pentec 1992) to 85 percent (Shapiro and Associates 1979) loss of the original estuarine wetland area. Estuarine wetlands are an

extremely important component in the marine and terrestrial food chain, providing critical habitat for fish, birds, and other wildlife including species which are commercially important. Studies in the Pacific Northwest have demonstrated that estuarine marshes are more productive than other plant communities. Frenkel measured the productivity of a diked pasture at 1200 grams/meter^2 relative to 2300 grams/meter^2 for the same pasture 10 years after it was restored as estuarine habitat (Frenkel 1990). The loss of estuarine habitat statewide has also been significant.

Based on this information the technical committee established that one of the principal goals of the management plan is to restore diked palustrine wetlands to tidal influence in order to provide fish habitat for the full range of estuary fish species, with emphasis placed on overwintering and freshwater/saltwater transition habitat for juvenile salmonids. Included within this goal was the restoration of shoreline migration corridors for juvenile salmonids and other fish species.

The second principal restoration and enhancement goal was enhancing agricultural lands for bird, herptile, and mammal habitat, and as wildlife corridors. This goal was based on the input of farmers and other users of the estuary who wished to maintain and protect agriculture and existing forested and scrub-shrub palustrine habitat on the 4000-acre South Ebey Island.

On a regional basis the Snohomish Estuary is an important habitat for overwintering and migrating waterfowl and shorebirds. It provides an alternative refuge site for waterbirds during the winter to escape severe weather present at estuaries in Northern Puget Sound and the Fraser Estuary in Canada. For example, ducks can move 15 to 110 km south from the Fraser River Delta to numerous bays and estuaries in Puget Sound (Shreffler & Thom 1993).

Agricultural fields are very important to a large number of waterbirds. Brennan reports that shorebirds and waterfowl feeding intertidally typically roost in adjacent fields at high tide or during storms (Brennan et al. 1985). Shreffler and Thom state that large intertidal areas without adjacent farmland usually support fewer ducks and do not provide complete wintering habitat (Shreffler & Thom 1993). Surface feeding ducks switch to feeding in freshwater wetlands and agricultural fields in the winter because eelgrass (*Zostera spp.)* is not available in intertidal areas. Therefore, this plan specifies the enhancement of agricultural fields through a combination of flooding and planting off-season crops in heavily cultivated fields, for forage.

Identification of Restoration and Enhancement Sites and Actions and Ranking of Sites (Steps 2, 3, and 4)

The central criteria for selecting appropriate restoration and enhancement sites and actions was that they must result in a stable, persistent, resistant and resilient ecosystem. A *stable* estuarine ecosystem does not readily transform over the long term from one system, such as an estuarine mudflat, to another such as a salt marsh, except when it has been significantly impacted by human or naturally induced perturbations such as subsidence or excess sedimentation. A *persistent* ecosystem is one such as a forested tidal swamp, which maintains its habitat structure over a long period of time. An emergent *Carex* marsh would be considered to be *resistant and resilient* if it could readily recover from disturbances such as flooding and erosion which removed large areas of vegetation.

The structure of the restored/enhanced estuary is critical in obtaining these characteristics of *stability, persistence, resistance, and resilience.* Structure consists of "patches" of habitat, their shape and location in the estuary, and the corridors connecting these patches. Without all of these elements, restoration and enhancement efforts have little chance of success (Shreffler & Thom 1993).

In general, tidal marshes of at least 250 to 300 acres appear to be stable and functional ecosystems (Collins et al. 1987). Patches of larger size tend to support more species than small patches. (Shreffler & Thom 1993). Patches should be connected in a manner that promotes export of primary productivity, movement of fish and animals, and a high diversity of "non-opportunistic" native species. Shreffler and Thom concluded that *"unconnected habitats are not preferable and that restoration of systems supporting large and small habitats, with*

maximum connectance is a more appropriate restoration concept."

Shreffler and Thom (1993) also state that *"the chances of successful restoration are higher if the landscape is relatively intact even though the degree of disturbance of the restoration site is high."* The habitat assessment demonstrates that the core of the estuary is intact, based on high functional performance scores. The potential restoration sites adjoining these areas have only been temporarily disrupted by agriculture.

To implement these landscape principles, the Management plan focused on restoring large areas of habitat that are adjacent to the "core" of Group 1 wetlands that are relatively undisturbed. These tidal restoration areas, selected and ranked by the technical committee, are depicted in Figure 3 and include the Poortinga Property on the east mainland, North Spencer Island, West Smith Island and Diking District 6. Enhancement areas within diked palustrine (agricultural areas) wetlands were concentrated primarily on South Ebey Island and Marshland area (not shown on Figure 3).

In addition to the broader landscape principals in selecting restoration and enhancement sites, the technical committee used specific criteria based on the IVA functional performance scores, elevational data and the presence of any fill or toxic materials. For example, any area that scored in Group I for the Overwintering Birds or Herptiles function was ranked lower for tidal restoration.

The technical committee also discussed the effect that subsidence in the estuary would have upon the success of tidal restoration. Considerable subsidence appears to have occurred in the estuary as the result of diking, draining and farming activities. Elevation is a primary control on most tidal marsh processes because it determines the frequency and period of daily and seasonal tidal inundation (Frenkel 1990). SEWIP elevation data and vegetation surveys indicate that mudflats are generally found below +6 feet MLLW; emergent wetlands generally above +7 feet MLLW; and scrub-shrub, forested marshes above +12 feet MLLW. Therefore, the lower limit for vegetated tidal marshes in the estuary appears to approximately be +7 to +8 MLLW. After considerable debate, the technical committee determined that the presence of substantial areas of diked palustrine wetland at +8 MLLW, should not decrease its feasibility for restoration to tidal habitat because: 1) the management plan is designed to ensure the long term protection of a tidal restoration site; and 2) physical, chemical and biological estuarine processes will eventually lead to the establishment of an intertidal marsh and forested swamp with high performance of all functions.

Tidal restoration actions included: the removal of the maximum practicable length of dike for tidal restoration; re-connect old sloughs and tidal creeks with the main tidal channel by overexcavating the old sloughs and tidal creeks where they cross the former dike location. Frenkel and Morlan found that sediment accretion was highest in those areas that had the greatest degree of tidal exchange (Frenkel 1990). This typically occurred around tidal creeks. The least amount of sediment increase occurred in areas which still had dikes in place. Frenkel and Morlan also found that tidal creeks were a major determinant in marsh hydrology (Frenkel 1990). Removal of large portions of the dike allows for the accumulation of large woody debris which will increase wetland elevation in severely subsided areas and allow for the formation of a more diverse wetland plant community. Other restoration actions identified by the committee included creating island habitat out of existing dikes. Enhancement actions for diked palustrine wetlands included increasing flooding in agricultural fields in order to improve bird and amphibian habitat and creating scrub-shrub corridors along ditches at the edge of agricultural fields, particularly between the forested habitat on South Ebey and along the "water-edge" units.

Identification of Development Footprint and Development of a Compensation Mechanism (Steps 5, 6, and 7)

One of the purposes of the SEWIP document is to determine if there is an adequate level of compensation credits to offset the impacts from the filling of wetlands with low functional performance. The technical and user committees identified a 352-acre development footprint (Figure 3) within an existing area of industrial development with Group III

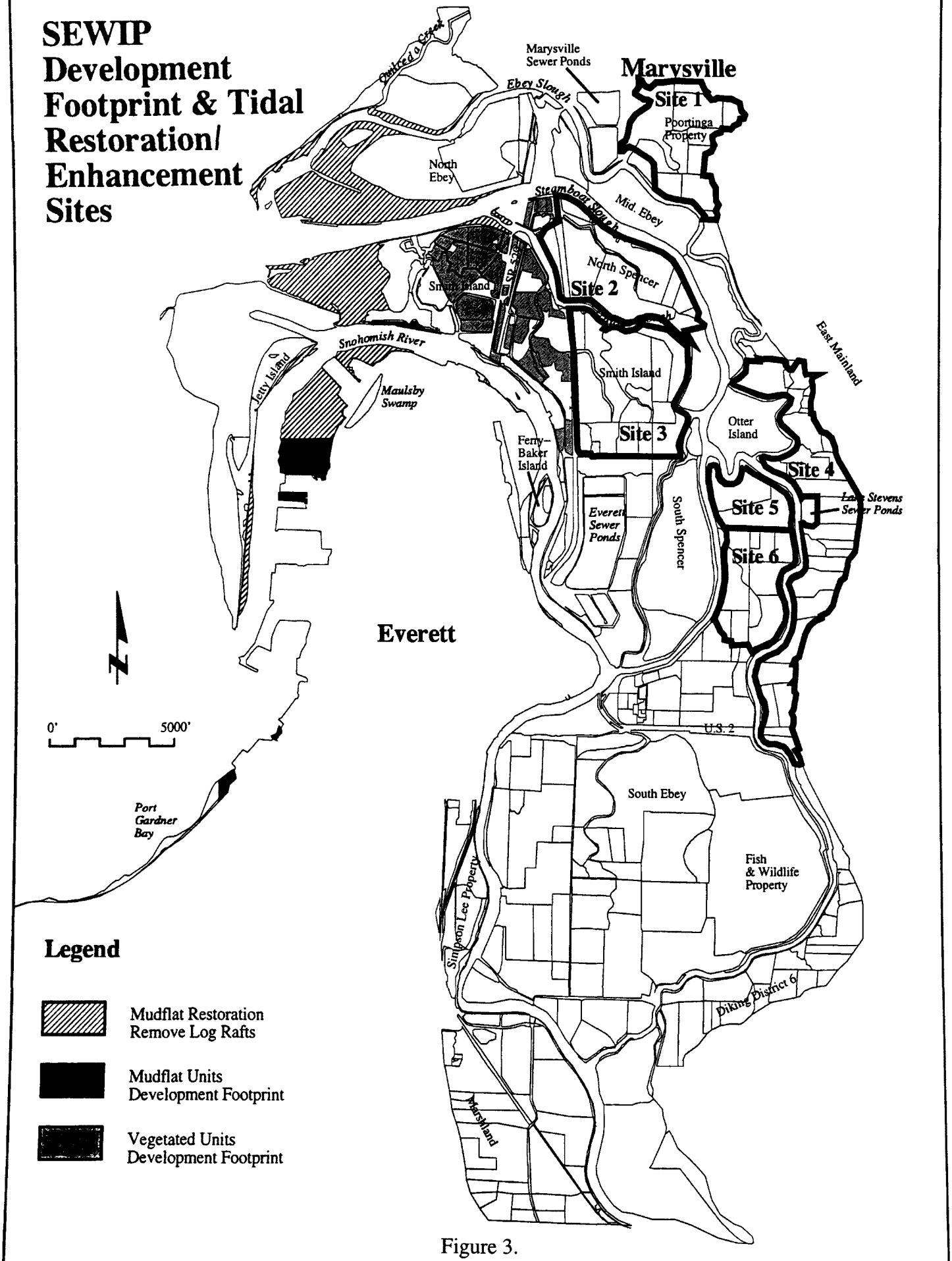

SEWIP
Development
Footprint & Tidal
Restoration/
Enhancement
Sites
Marysville
Sewer Ponds
Marysville
Site 1
Ebey Slough
Poortinga
Property
North
Ebey
Steamboat Slough
Mid. Ebey
North Spencer
Site 2
Smith Island
Snohomish River
Jetty Island
Maulsby
Swamp
Smith Island
East Mainland
Site 3
Otter
Island
Ferry-
Baker
Island
Site 4
Site 5
Everett
Sewer
Ponds
South Spencer
Site 6
Everett
N
0'
5000'
U.S. 2
South Ebey
Port
Gardner
Bay
Fish
& Wildlife
Property
Simpson Lee Property
Diking District 6
Marysland
Legend
Mudflat Restoration
Remove Log Rafts
Mudflat Units
Development Footprint
Vegetated Units
Development Footprint

Figure 3.

wetlands (low functional performance). This area scored high for the Access to Transportation Corridors function given the immediate presence of dredged tidal waterways, rail lines and two major freeways. The wetland impacts for this 352-acre area were 10,761-acre points (IVA ranking score x the acreage of the wetland complex) for the Water Quality function and 4,500-acre points for the Wildlife function (see Table 2).

Based on the "restoration/enhancement actions" identified for the estuary restoration/enhancement sites, the IVA model was run again for the affected restoration sites and the new "increased" scores tabulated. From these new scores or "compensation credits" it could then be determined if they were adequate to offset the impacts from the "development footprint" located within the area of Group III wetlands. The compensation scores for the Priority 1 restoration site (Poortinga Property) are displayed in Table 2. Table 2 demonstrates that the Priority #1 Restoration site has more than enough compensation credits to offset the total impacts from the development footprint for vegetated wetlands.

With adequate compensation credits established for the development footprint, the technical committee then considered the appropriate compensation ratio to apply. By examining the average increase for the Water Quality Improvement and Wildlife Functions for each restoration and enhancement site and comparing that increase to the impact debits for the corresponding functions in the Development Footprint, the overall compensation ratio was determined.

Table 3 shows that the net increase in function for the first four restoration sites usually exceeds the existing level of function performance. However, the Water Quality Improvement Function does provide less increase in function per acre than the Wildlife Function, so the Compensation Ratios presented in Table 3 are based on the average loss or gain in the Water Quality Improvement Function. Table 3 shows that the average wetland impacts in the development footprint relative to the average wetland restoration gains will result in a less than 1:1 acreage replacement ratio for priority site 1. However, because there are policies among resource agencies requiring that compensation ratios should be at least a 1:1 ratio on a acreage basis, a 1:1 compensation ratio was selected for the priority 1 restoration site.

Priority sites 2, 3, and 4 demonstrated an progressive increase in the compensation ratio to replace loss of Water Quality Improvement functions. This is due to the decreasing wetland functional performance at these restoration sites based on IVA model results. The compensation ratio based on acreage (column 6) was increased by approximately 25 percent above the functional loss ratio (column 5) to account for the lower priority of the site and to encourage restoration of the higher ranked site.

Conclusions

Though not yet adopted by local governments or state and federal agencies, the SEWIP document has already proved valuable to local planning agencies, the Port District, and local industries. Prior to the management plan, compensation within the estuary was often improperly sited and not appropriately designed. Purchase of compensation sites took place without any overview to determine what the benefit of the compensation site would be to the overall estuary. This resulted in isolated compensation sites that were much more expensive to engineer and construct than if a larger, better located, compensation area was purchased and restored. The assessment of the estuary's wetland resources on a landscape basis has encouraged resource agencies to coordinate efforts to purchase higher ranked restoration sites and undertake the restoration actions identified in the plan which are designed to provide the greatest ecological benefit to the estuary.

More importantly, the SEWIP process has brought together local governments and resource agencies that normally have conflicting mandates for the estuary and forced them to plan for the future of the estuary from a ecological perspective, rather than from a fragmented "market demand" perspective.

The IVA method proved to be a relatively accurate and rapid assessment tool for establishing the relative value of wetlands within the estuary for the purposes of developing a detailed wetland

Table 2. Summary of development footprint impacts and restoration credits for Poortinga site in Snohomish Estuary

Site	Impacts to Water Quality Functions in Acre Points	Restoration Credits for Water Quality Functions in Acre Points	Impacts to Wildlife Functions in Acre Points	Restoration Credits for Water Quality Functions in Acre Points
Development Footprint for Vegetated Wetlands	10,761	None	4,500	
Priority #1 Restoration Site, Poortinga Property (350 acres)	None	16,350	None	18,326

Table 3. Calculated compensation ratios for Snohomish Estuary - vegetated wetlands

Restoration Priority and Location	Wetland Functions	Average Performance of Functions for Development Footprint (per acre)	Average Increase in Performance for Restoration-Enhancement Area. (per acre)	Compensation Ratio to offset function loss & temporal loss (i.e. 1.25 x functional loss ratio)	Compensation Ratio Based on Priority & Acreage	Reasons for Selected Compensation Ratio
Priority 1 - Poortinga	Wildlife	13	57	0.3:1	**1x**	Using less than 1:1 would not be consistent with National Policies for no net loss.
Priority 1 - Poortinga	Water Quality	31	51	**0.8:1**		
Priority 2 - N. Spencer (Beringer)	Wildlife	13	54.5	0.3:1	**1.3x**	The higher ratio creates an incentive for restoration of Priority site 1 first
Priority 2 - N. Spencer (Beringer)	Water Quality	31	41.8	**0.9:1**		
Priority 3 - E. Smith Island	Wildlife	13	51.2	0.3:1	**1.5x**	The higher ratio creates an incentive for restoration of Priority site 1 and 2 first
Priority 3 - E. Smith Island	Water Quality	31	32.8	**1.2:1**		
Priority 4 & 4A - East Mainland	Wildlife	13	53.4	0.3:1	**2x**	The higher ratio creates an incentive for restoration of Priority site 1, 2 and 3 first
Priority 4 & 4A - East Mainland	Water Quality	31	26.6	**1.5:1**		

management plan. More rigorous "quantitative" models based on years of qualitative observations would not have allowed this regional planning effort to proceed in a timely fashion, thereby allowing development to further fragment and possibly degrade the overall ecosystem permanently.

Acknowledgments

The development of the SEWIP document would not have been possible without an unusual level of support and cooperation between resource agencies and local governments. Due to the foresight of individuals, this process became a partnership in which jurisdictional politics were set aside so that land use planning based on ecosystems rather than political boundaries could take place. We are grateful to the Environmental Protection Agency (EPA), Department of Ecology (DOE) and Puget Sound Water Quality Authority for funding this planning effort; to the members of the SEWIP technical and user committees; to Lynn Beaton for her assistance during the grant and development of the plan; and to Dr. Tom Hruby for his assistance throughout the process on the development and application of the IVA model.

Literature Cited

Adams, L. W., and L. E. Dove. 1989. Wildlife Reserves and Corridors in the Urban Environment: A Guide to Ecological Landscape Planning and Resource Conservation. National Institute for Urban Wildlife, Columbia, MD.

Aronson, J., C. Floret, E. LeFloc'h, C. Ovalle, and R. Pontanier. 1993. Restoration and Rehabilitation of Degraded Ecosystems in Arid and Semi-Arid Lands. I. A view from the South. Restoration Ecology 1(1): 8-17.

Bortleson, G. C., M. J. Chrzastowski, and A. K. Helgerson. 1980. Historical changes of shoreline and wetland at eleven major deltas in the Puget Sound region, Washington. Hydrologic Investigations Atlas HA-617, USGS.

Boule, M. E., G. B. Shea, and A. L. Driscoll. 1978. Snohomish estuary wetlands study: delineation of wetland boundaries. Volume 4. Report of Northwest Environmental Consultants, Inc., to United States Army Corps of Engineers, Seattle District, WA.

Burrell, G. 1978. Snohomish estuary wetlands study: classification and mapping. Volume 3. Report of Washington State Department of Game to United States Army Corps of Engineers, Seattle District, WA.

Collins, L. M., J. N. Collins, and L.B. Leopold. 1987. Geomorphic Processes of an Estuarine Tidal Marsh: Preliminary Results and Hypotheses. pp. 1049 - 1072 In V. Garner (ed). International Geomorphology 1986 Part I.. John Wiley and Sons, New York, NY.

_____. 1991. Preliminary draft of Jetty Island management plan. Prepared for the Port of Everett.

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

Driscoll, A. L. 1979. Snohomish Estuary Wetlands Study (4 volumes). Shapiro and Associates, Inc., Seattle, WA.

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

Frenkel, R. E. and J. C. Morlan. 1990. Restoration of the Salmon River Salt Marshes: Retrospect and Prospect. Department of Geosciences Oregon State University, Corvallis, Oregon. Final Report to the U.S. Environmental Protection Agency, Seattle, WA.

Hruby, T., W. E. Cesanek, and K. E. Miller. In preparation. Establishing relative wetland values on a regional scale: a step beyond WET.

Josselyn, M., J. Zedler, and T. Griswold. 1989. Wetland Mitigation Along the Pacific Coast of the United States. pp. 3-36 In J. A. Kusler and M. E. Kentula (eds). 1990. Wetland Creation and Restoration: The Status of the Science.. Island Press, Covelo, CA.

Kusler, J. A. and M. E. Kentula (eds). 1990. Wetland Creation and Restoration: The Status of the Science. Island Press, Covelo, CA.

Leibowitz, S. G., B. Abbruzzeze, P. R. Adamus, L. E. Hughes, and G. T. Irish. 1992. A Synoptic Approach to Cumulative Impact Assessment. EPA/600/R-92/167 October 1992. EPA Environmental Research Laboratory, Corvallis, OR.

Lewis, Roy R. III. 1982. Creation and Restoration of Coastal Plant Communities. CRC Press, Boca Raton.

Lovvorn, J. 1995. Panel discussion (January 13, 1995) at Puget Sound Research '95, Bellevue, WA. University of Wyoming, Zoology Department, Laramie, WY.

McGuiness, M., et al. 1988. Snohomish County Watershed Ranking Project, Final Report. Public Works, Snohomish County, Everett, WA.

National Research Council (NRC). 1992. Restoration of Aquatic Ecosystems. National Academy Press. Washington, D.C.

Pentec Environmental, Inc. 1992. Port of Everett Landscape Analysis for Port Gardner and the Snohomish River Estuary. Pentec Environmental, Inc., Edmonds, WA.

Pickett, S. T. A. and P. S. White (eds). 1985. The Ecology of Natural Disturbances and Patch Dynamics. Academic Press, Inc. Orlando, FL.

Pritchard, D. W. 1967. What is an estuary: physical viewpoint. pp. 3-5 In Lauff, G. H. (ed.), Estuaries. Pub No. 83, American Association for the Advancement of Science, Washington, D.C.

Schaefer, G. M., and M. T. Brown. 1992. Designing and protecting river corridors for wildlife. Rivers 3(1):14-26.

Shapiro and Associates, Inc. 1979. Snohomish Estuary Wetlands Study. Volume 1: Summary Report. Prepared for U.S. Army Corps of Engineers, Seattle District, WA.

Shapiro & Associates, Inc. 1979. Snohomish Estuary Wetlands Study. Volume 2: Baseline information and evaluation. Prepared for US Army Corps of Engineers, Seattle District, WA.

Shapiro & Associates, Inc. 1985. Snohomish River wetland units preservation management plan. Prepared for Snohomish County Department of Planning and Community Development, Everett, WA.

Shapiro & Associates, Inc. 1989. Snohomish River wetlands management plan. Final report to Snohomish County Department of Planning and Community Development, Everett, WA.

Shreffler, D. K., and R. M.Thom. 1993. Restoration of Urban Estuaries: New Approaches for Site Location and Design. Battelle/Marine Sciences Laboratory. Prepared for Washington State Department of Natural Resources Aquatic Lands Division, Olympia, WA.

Simenstad, C. A. 1983. The Ecology of Estuarine Channels of the Pacific Northwest Coast: A Community Profile. U.S. Fish and Wildlife Service, Division of Biological Services, WA, D. D. FWS/OBS-83/05.

Simenstad, C. A., C. D. Tanner, R. M. Thom, and L. Conquest. 1991. Estuarine Habitat Assessment Protocol. EPA 910/9-91-037. Report to U.S. Environmental Protection Agency, Region 10, Fish. Res. Inst., University of Washington, Seattle, WA. 201 pp.

Stevens, M. L., and R. Vanbianchi. Restoring Wetlands in Washington. 1993. Department of Ecology Publication #93-17, Olympia, WA.

Williams, J. R., H. F. Pearson, and J. D. Wilson. 1985. Stream flow statistics and drainage-basin characteristics for the Puget Sound region, WA. USGS: 84-144 B.

Zedler, J. B. and M. W. Weller. 1990. Overview and Future Directions. pp. 405-413 In J. A. Kusler and M. E. Kentula (eds). Wetland Creation and Restoration: The Status of the Science. Island Press, Covelo, CA.

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Planning for Wetland Restoration and Enhancement in the Context of Regional Wetland Management Plans

Thomas Hruby[1]

Abstract: **As development pressures on wetlands increase in urbanizing areas, many are turning to regional wetland management plans as a solution for the problem of balancing social and environmental needs. As these regional plans are being developed they provide significant opportunities for the restoration and enhancement of wetlands in the planning areas. One of the issues, however, in using restoration as a mitigation tool is to quantify the loss of wetland functions through development and determine whether any planned restoration and enhancement will actually mitigate for lost functions. This paper describes how the Indicator Value Assessment method (IVA) is being used assess the restoration potential of wetlands, and meet mitigation needs for the Mill Creek Special Area Management Plan.**

The IVA method provides a relative estimate of the performance of wetland functions and their value based on specific indicators. Numeric models use the indicators as variables that are summed or multiplied. Value is then estimated based on: 1) the performance score, 2) the relative value assigned the function by a local oversight/citizens committee, and 3) the area of the wetland. The restoration potential of a wetland is assessed by estimating the value of functions after restoration based on changes to indicators. The mitigation potential is then set as the difference in value between existing and future conditions.

Of the 128 wetlands assessed in the Mill Creek SAMP, 42 have been identified as suitable for enhancement. Initial calculations indicate that the value of the floodflow alteration functions in these wetlands might be increased by about 20 percent; the value of fish habitat functions might be increased by about 50 percent; the value of habitat for other wildlife species might be increased by about 50 percent, and the value of water quality improvement functions might be increased by about 30 percent. The potential increase in the value of wetland functions through restoration/ enhancement also sets the upper limit of the impacts to wetland values that might be allowed through development. Even when wetlands which have lower relative values for the different functions are chosen for development, the potential impacts have to be limited to less than 400 acres of wetlands out of the 2500 acres present in the basin. To balance the value of functions lost through filling 400 acres of wetlands will require restoring/enhancing all 1465 acres of wetlands identified in the restoration plan.

Introduction

Development pressures on wetlands are increasing as buildable land becomes scarce in urbanizing areas. One solution for accommodating both economic and environmental needs is to develop regional management plans that identify, in advance, wetlands suitable for development and those for preservation or restoration. Moreover, such regional plans provide significant opportunities for the restoration and enhancement of wetlands. The restoration or enhancement of wetland functions can be used as compensation for functions lost through the development when the process is linked in a regional plan. One issue, however, in using restoration as a tool for compensation is to find a method that will adequately quantify the loss of wetland functions and determine whether any planned restoration and enhancement will actually compensate for wetland functions lost through development. (NOTE: In this paper I use the term "restoration" to mean both true restoration and enhancement: where restoration is the re-establishment of wetland functions in areas that were once wetlands, and enhancement is an increase in the performance of some functions in areas that are currently wetlands.)

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This paper describes the process being used to integrate restoration planning with requirements for compensatory mitigation in three regional wetland management plans. The three wetland management plans under development are the Hackensack Meadowlands Special Area Management Plan in New Jersey (Meadowlands SAMP); the Mill Creek Special Area Management Plan in Washington State (Mill Creek SAMP), and the Snohomish Estuary Wetland Integration Plan also in Washington (SEWIP).

Unavoidable impacts to wetland functions in all three plans are to be compensated by undertaking specific restoration actions in the watershed. The wetland management plans are the basis for U.S. Army Corps of Engineers Regional General Permits under the Clean Water Act, Section 404. Compensatory mitigation for impacts outlined in a development plan is constrained to specific activities outlined in a restoration plan. This means that anyone proposing to impact wetlands will have to limit their compensation to activities outlined in the plan. The advantage of such an approach for a developer is a more predictable and faster per-mitting process. The advantages from a restoration perspective is that the compensation requirements provide some of the capital necessary to begin restoration efforts on a watershed scale.

The basic premise in the three wetland management plans is that unavoidable impacts are compensated by enhancing or restoring other wetlands in the planning region. The goal is to achieve "no net loss of functions and values." Wetland values lost through development are to be compensated by increasing values elsewhere in the watershed. In order for such a process to work, a method is needed that will quantify both the impacts to wetland functions and those that may be gained through restoration activities. No numeric method existed for assessing wetland functions that met planning needs when we started the process over five years ago in the Meadowlands. As a result a new method was developed called the Indicator Value Assessment Method (IVA) (Hruby, et al. 1995).

The IVA is used to track changes in the performance of functions and their values resulting from unavoidable impacts to wetlands and those chosen for restoration. Losses in wetland values can be assessed directly from the IVA value scores of wetlands that will be impacted. The amount of compensation that can be achieved is estimated by recalculating the performance and value scores of functions for areas that are restored or enhanced. Indicators that would change or be added in the "restored" condition are identified and added to the assessment models. The compensation potential is then set as the difference in value between existing and potential future conditions.

Briefly: The IVA method provides a relative estimate of the performance of wetland functions and their value based on specific indicators. Numeric models use the indicators as variables that are summed or multiplied. Models take the following form:

$$\text{Performance} = f(\text{indicators})$$

The coefficients reflecting the importance of an indicator in the expression of function are assigned and developed based on local conditions. The wetland with the highest performance in the basin is scored at 100 and the rest are scored relative to this. Value is then estimated based on: 1) the performance score, 2) the relative value assigned the function by a local oversight/citizens committee, and 3) the area of the wetland.

$$\text{Value} = f(\text{performance, value of function, area of wetland})$$

The wetland functions assessed in these planning processes are chosen locally and reflect local values. Ten functions were assessed in the Meadowlands SAMP; 12 in the Mill Creek Samp, and 15 in the Snohomish estuary plan (Table 1). To simplify the valuation process, the performance scores for the individual functions were combined into scores for groups of functions such as Wildlife Habitat, Fish Habitat, and Water Quality Improvement. These scores represent the relative value per acre and not performance, because the performance scores for the individual functions were combined based on the relative values of the functions (see Hruby et al. 1995 for a more detailed discussion of the calculations).

Developing A Restoration Plan

The following section describes the process used to plan restoration and enhancement activities, and this is graphically represented in Figure 1. Planning restoration and enhancement activities in a watershed is initially independent of the development of the wetland management plan and the assessment

Table 1. Wetland functions assessed in three wetland management plans.

Meadowlands SAMP	*Mill Creek SAMP*	*Lower Snohomish SEWIP*
Nutrient Uptake	Nutrient Uptake	Nutrient Uptake
Retention of Toxics	Retention of Toxics	Retention of Toxics
Export of Production	Export of Production	Export of Production
Floodflow Alteration	Floodflow Alteration	Sediment Stabilization
Recreation	Sediment Stabilization	Recreation
Conservation Potential	Groundwater Discharge	Channel Stabilization
		Access to Deep Water (transportation)
	Habitat Functions	
Aquatic species	Aquatic species	Anadromous Fish
General Fish	Anadromous Fish	Resident Fish
General Waterfowl	Resident Fish	Migratory Bird
General Wildlife	Migratory Bird	Overwintering Bird
	Resident Bird	Breeding Bird
	All Other Species	Invertebrate
		Reptile and Amphibian
		Mammal

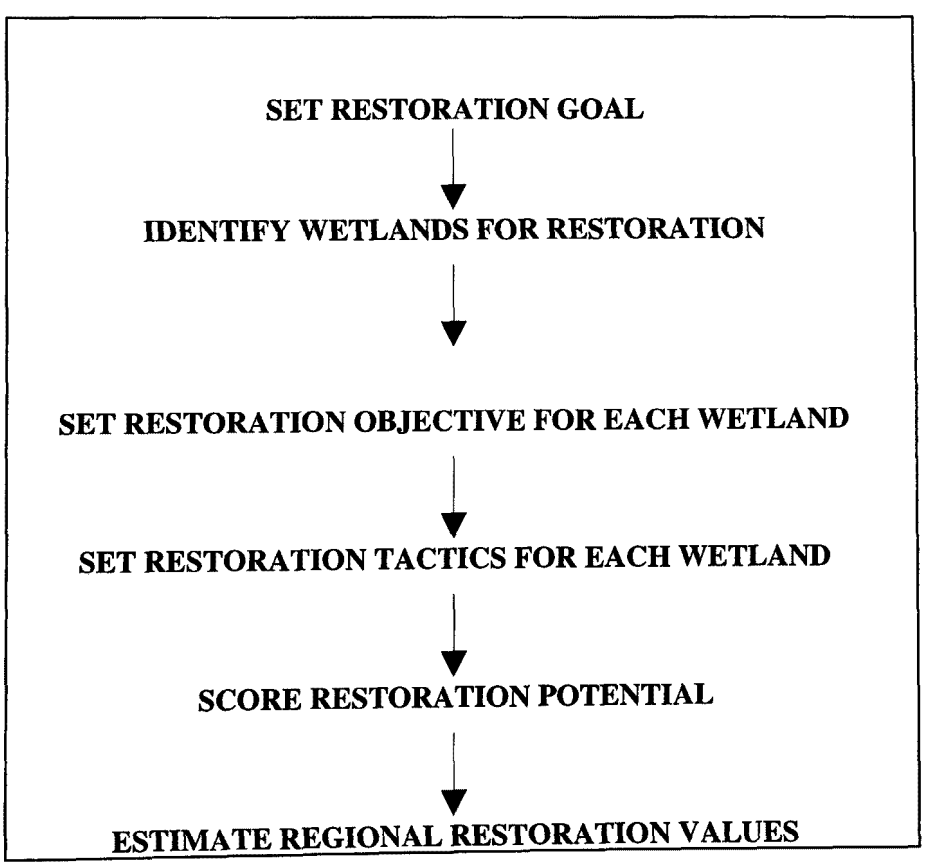

Figure 1. Planning steps in developing a restoration plan linked to a regional wetland management plan.

of wetland functions. The linkage between the restoration plan and the wetland management plan occurs in the last two steps. The same planning process has been used in the two SAMP's and in the SEWIP. For the sake of brevity, I will be using examples only from the Mill Creek SAMP and the SEWIP to illustrate the process.

Step 1: Setting a Restoration/ Enhancement Goal

Most wetland management plans are developed for a small watershed or basin that is under heavy development pressures (e.g., City of Eugene 1992). In such cases any remaining wetlands are often degraded by agricultural activities, stormwater runoff, or other human impacts. Many of the original functions performed by the wetlands have been lost, and the ecosystems in the watershed could benefit from some type of wetland restoration or enhancement.

The first step in developing a restoration plan is the crafting of a restoration goal suited for the watershed. In many cases in may not be possible to "move back the clock" and restore the wetland ecosystem to a pre-industrial state. Furthermore, the remaining wetlands may have had their functions changed and now perform important new functions. For example, in the Mill Creek watershed the agricultural wetlands have become important habitat for migratory waterfowl and raptors, when in the past they were forested and not accessible to these species. Restoration of the wetlands to their original forested state would eliminate this important current function. The restoration goal for Mill Creek therefore is "to establish an interconnected system of wetlands and uplands along Mill Creek that will effectively function as a natural ecosystem that supports fish and other wildlife." Some forested wetlands will be restored but the goal is not to restore the full extent of forested wetlands found in the valley before settlement by Europeans.

Watershed planning is effective only when cooperation and consensus can be achieved among the different owners and "users" in the watershed (Euphrat and Warkentin 1994). Thus, it is important that the restoration goal for a watershed be set by a group with representatives from resource agencies, local landowners, and other interested parties.

Step 2: Identifying Wetlands for Restoration

Once a goal has been established the next step is to identify the wetlands best suited for restoration. There are several ways in which this task may be approached including 1) developing a map by consensus of the "restoration committee," 2) developing specific restoration criteria and identifying areas that meet them; or 3) using a geographic information system (GIS) (an approach to restoration planning currently being developed by the Washington State Department of Ecology for the Puget Sound Water Quality Authority).

Wetlands in the Mill Creek SAMP were ranked as having a High, Medium, or Low potential for restoration based on the following criteria: 1) proximity to the major streams in the watershed, 2) landscape position, 3) low relative performance of existing functions, 4) adequate water regime, 5) location relative to the 100-year floodplain, and 6) technical feasibility. Of the 128 wetlands in the planning area, 42 had a High or Medium potential for restoration.

Step 3: Developing Restoration Objectives for Each Wetland

Once suitable wetlands have been identified, one or more restoration objectives are developed for each wetland. Objectives are chosen that best accomplish the overall goal within the limits of feasibility at the site. At this point it is important to identify where specific functions are to be restored in the watershed so these can be targeted to the individual wetlands. Not all functions can be restored to all wetlands. Decisions should be made by a group of wetland experts who know and understand the local ecosystems. Restoration objectives were set by such a group in all three plans developed. The process involved both general discussions in meetings and at least 2 to 3 days of field visits to the wetlands that met the original criteria.

Examples of restoration objectives for three wetlands in the Mill Creek basin are listed below:

Wetland 2E: Enhance wetland for waterfowl habitat and recreate a forested riparian buffer along the tributaries of Mullen Slough.

Wetland 5KKK: Raise dissolved oxygen to acceptable levels.

Wetland 5L: Increase water supply in spring and early summer to encourage nesting by waterfowl.

Step 4: Setting Restoration Tactics for Each Wetland

Restoration objectives are implemented by carrying out specific actions in a wetland. These actions can be considered as "restoration tactics," and also need to be specified in the plan. Although several wetlands may share similar objectives the tactics to be used depend on conditions in that specific wetland. Tactics need to be determined by a group of local experts and verified by site visits. Because tactics are very specific to the sites in question, they will vary between wetlands, basins and watersheds. Furthermore a single wetland may be targeted for more than one tactic.

The 19 restoration tactics chosen for wetlands in the Mill Creek basin and the 15 chosen for the lower Snohomish River estuary are listed Table 2.

Step 5: Scoring Restoration Potential

Many indicators of functions in a wetland will change if restoration tactics are implemented as proposed in a plan. It is the change in indicators that forms the basis of estimating the increases or decreases in the performance and values of functions using the IVA. New IVA scores are calculated for each wetland identified in the restoration plan by postulating the indicators that would be present after restoration. Thus, if the tactic is to add interspersion between land and water the indicator for interspersion is changed to reflect this (e.g., from "no interspersion" to "moderate interspersion" or "mosaic").

IVA performance models are re-calculated for each function based on the changes to the indicators. If a wetland already has the indicator the score is not changed. This means that wetlands that are performing functions well would not have their scores changed as much as wetlands where functions are poorly performed. Furthermore, the increase in potential performance is dictated by the tactics identified for that wetland. As a result each wetland ends up with a different potential increase in performance.

The performance scores for each function have been combined in the IVA models developed into scores for the per hectare value of major groups of functions in each wetland, as described in Hruby, et al. (1995). Table 3 shows the potential increase in the per acre value scores of wildlife habitat functions and water quality improvement functions for 11 wetlands in the lower Snohomish River estuary. Most of these 11 wetlands currently have low value scores for relative performance and thus provide opportunities for significant increases in value. The scores suggest that the performance of functions may be increased two or three times over existing levels by appropriate enhancements.

Such was not the case in the Mill Creek, however, where the wetlands identified for restoration only averaged a 50 percent increase in the per acre value of the habitat functions, and only a 30 percent increase in the performance of water quality improvement functions (Table 4). The existing wetlands had fairly high levels of performance and thus could not be enhanced much further.

Step 6: Calculating Increase In Value

The potential for compensation at each wetland is estimated by multiplying the potential increase in the performance score by the acreage of the wetland, as shown in Table 3. The potential is represented as a dimensionless number we have called "hectare-points" (or "acre-points" if English measures are used). These numbers represent the "values" in functions that can be accrued through restoration and enhancement.

Discussion

Impacts are estimated in terms of "hectare-points" that will be lost by multiplying the values for groups of functions in the wetlands to be impacted by the size of the impact. Because the value scores generated in the restoration plan are comparable, it is easy to establish policies by which impacts to some wetlands are to be compensated for by restoration or enhancement of other wetlands. State and federal policies state that the compensation for losses to wetland functions should be at least on a one to one basis. Thus every hectare-point of impact needs to be compensated by the increase of one hectare-point through restoration or enhancement. In practice, however, the policies being developed for the regional management plans call for replacement ratios greater than one to one, to compensate

Table 2. Tactics used to enhance wetland functions.

Mill Creek SAMP	*Everett SEWIP*
Riparian planting of trees	Remove the maximum possible area of dike
Plant native emergent species	Grade dike to provide natural transitions
Plant native scrub/shrub species	Create islands
Restore native forest	Reconnect remnant tidal streams
Restore aquatic bed habitat elements	Excavate channels to "pre-dike" depths
Excavate ponds	Fill interior drainage ditches
Add interspersion between vegetation classes	Restore upland buffers
Add interspersion between water and land	Restore a scrub/shrub edge to ditches and streams
Restore natural stream characteristics	Restore emergent tidal marsh species
Restrict access	Add native emergent wetland plants
Construct check dams to hold back water	Restore full tidal regime
Reconfigure the stream channel	Increase seasonal flooding to 25 percent of wetland
Enhance access of floodwaters to wetland	
Add side channels off main stream	
Add structures to control water flows	
Control or remove livestock from wetland	
Control exotic or invasive plant species	
Restore upland buffers	
Modify existing flapgates	
Add specific habitat structures	

Table 3. Changes in the IVA scores resulting from proposed restoration actions in the lower Snohomish estuary (SEWIP).

		Wildlife Habitat Group of Functions				***Water Quality Improvement Group of Functions***			
SITE	Hectares	Score Before Restoration	After	**Net Increase Pts/ha**	Total Increase ha/Pts	Score Before Restoration	After	**Net Increase Pts/ha**	Total Increase ha/Pts
146	9.2	15	72	**58**	531	10	79	**69**	637
147	33.9	14	75	**61**	2,064	14	82	**68**	2,315
339	7.8	12	67	**55**	423	38	79	**41**	317
340	6.8	10	67	**57**	387	47	79	**33**	223
341	23.8	13	66	**53**	1,260	41	79	**38**	907
342	5.0	18	66	**4.8**	239	26	50	**23**	116
343	7.0	15	68	**54**	373	41	79	**38**	265
360	1.8	11	70	**59**	105	17	66	**50**	89
361	9.9	15	74	**59**	583	10	79	**68**	683
362	14.9	12	68	**57**	848	9	50	**40**	605
363	8.8	9	64	**56**	489	9	50	**40**	356

Table 4. Average scores for impacts and restoration potential used in developing policy for the Mill Creek SAMP.

	GROUP OF FUNCTIONS		
	Fish Habitat	*Habitat For Other Species*	*Water Quality Improvement*
AVERAGE SCORE/ACRE IN:			
Development "footprint"	10	56	65
Wetlands BEFORE restoration	42	64	75
Wetlands AFTER restoration	65	98	105
NET gain through restoration	**23**	**34**	**30**
RATIO of net loss/acre to net gain/acre	0.4	1.7	2.2
AVERAGE compensation ratio by acre of impact (including 1.25 multiplier)	**0.5**	**2.1**	**2.8**

for the loss of functions between the time a wetland is impacted and the time that the restoration actions have achieved their planned level of performance.

The estimates of values lost and gained for each wetland generated by the IVA can be summarized in different ways to help guide policy decisions in the plan development. Results from the Mill Creek SAMP are presented in Table 4 as an example. The policies that are being developed from such data are as follows:

1. The water quality improvement group of functions are the limiting ones in terms of establishing the amount of compensation required. On the average, an acre of impacts to wetlands will require the enhancement of 2.8 acres of wetlands identified in the restoration plan.

2. Given that the average increases in values differ among the three groups of functions, it is the limiting one (i.e., water quality improvement) that will be used to determine the size of compensation necessary. Any extra values to be gained in the non-limiting function groups represent a "benefit" for the natural resources in the area, and are not tradeable for impacts from other projects.

3. Even when wetlands, which have lower relative values for the different functions are chosen for development, the potential impacts have to be limited to less than 400 acres of wetlands out of the 2500 acres present in the basin. Compensation for the value of functions lost through filling 400 acres of wetlands will require restoring/enhancing all 1,465 acres of wetlands identified in the restoration plan.

In conclusion, I would like to note that the full process described has now been completed for the Mill Creek and the Meadowlands wetland management plans, and we are just beginning the process for the Lower Snohomish River Plan. Final decisions on the how the exchange of credits is to take place are still to be made.

Literature Cited

City of Eugene. 1992. West Eugene Wetlands Plan. City of Eugene, OR.

Euphrat, F. D., and B. P. Warkentin. 1994. A watershed assessment primer. EPA 910/B-94-005.

Hruby, T., W. E. Cesanek, and K.E. Miller. 1995. Estimating relative wetland values for regional planning. Wetlands 15:93-107.

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Integrating Wetland Compensation Banking With LocalWetland Planning

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Abstract: **Two recent alternatives to traditional wetland compensatory mitigation under the section 404 program of the Clean Water Act are being experimented with around the nation: wetland mitigation banking and local wetland planning programs. Wetland mitigation banking is a process that coordinates multiple compensatory mitigation efforts into one mitigation plan or project. Local wetland planning is a process that coordinates the mitigation sequence steps for multiple development proposals within a specific landscape management unit, with the goal of balancing wetland protection and economic development. There are proponents and opponents of both processes. One major advantage often cited for these processes is that they allow compensatory mitigation to occur within a landscape perspective or, in keeping with the theme of this meeting, they encourage "taking a broader view." This paper outlines the opportunities as well as the difficulties associated with "taking a broader view" in wetland compensatory mitigation and focuses attention on future research needs.**

Introduction

Throughout the United States, communities are experimenting with wetland compensation banking and local wetland planning as alternative approaches to satisfy the "compensatory mitigation" and "practicable alternatives test" requirements of the Section 404 program of the Clean Water Act. There are both advantages and disadvantages to these approaches. One major advantage is that they encourage implementation of the Section 404 program within a landscape perspective, or, in keeping with the theme of this meeting, they encourage "taking a broader view." This paper explores the potential of integrating wetland compensation banking efforts with local wetland planning. First, I present the results of a recent research project that indicates interest among local government institutions for integrating wetland banking efforts with local wetland planning. Second, I review wetland compensation banking and local wetland planning and consider their link to the Section 404 program of the Clean Water Act, the range of activities involved in each approach, and the advantages and disadvantages associated with each approach. Third, I explore the idea of integrating the two approaches by looking at how their activities either overlap or complement one another, and the advantages and disadvantages of integrating the approaches. Finally, I conclude by providing a set of research recommendations that will provide information to improve the integration of wetland compensation banking with local wetland planning and allow the Section 404 program to move towards "taking a broader view."

Wetland Compensation Banking Study

On September 15, 1994, the Washington State Department of Transportation (WSDOT), five federal agencies, and two state agencies reached a Memorandum of Agreement (the Agreement) establishing the WSDOT Wetland Compensation Banking Program. The purpose of the Agreement is to "set forth the principles and procedures that all signatories to the Agreement will adhere to in establishing, implementing, and maintaining the Washington State Department of Transportation Wetland Compensation Bank Program" (WSDOT, 1994). The state and federal agencies did not include local and tribal governments in the negotiation process for the Agreement due to the sheer number of county governments (39), tribal governments (26), and many more city governments in the State of Washington. However, local and tribal governments may participate in establishing wetland compensation banks under the Agreement. Many of these governments have their own wetland protection regulations and programs.

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In February of 1994, WSDOT contracted with Professor Marc Hershman, Director, School of Marine Affairs, University of Washington, to investigate the implementation of the Agreement within the legal and institutional framework of county and tribal governments. The goal of the study was to identify important issues WSDOT will confront when it implements the Agreement within county and tribal government jurisdictions. The research team published the results of the study in a final report titled *The Washington State Department of Transportation Memorandum of Agreement for Wetland Compensation Banking: County and Tribal Participation* (Hershman and Green, 1995).

Four key findings of the study relate directly to the issue of integrating wetland compensation banking with local wetland planning. First, each of the county governments chose their planning department as the lead agency to respond to the Agreement. They see wetland compensation banking as directly linked to planning issues. Second, nearly all the county and tribal governments desired a primary role in locating wetland compensation bank projects. They believe that local knowledge of land use and environmental protection priorities should determine the location of wetland bank sites. Third, the county and tribal governments exhibit more flexibility in locating wetland compensatory mitigation projects than currently allowed by state and federal regulations. They believe that limiting compensatory mitigation to "on-site" and/or "in-kind" projects is too rigid. They encourage "off-site" and "out-of-kind" compensatory mitigation projects especially in urbanized areas. Fourth, most of the county and tribal governments participate in local planning efforts that include wetland protection.

These findings indicate that county and tribal governments have an interest in integrating wetland compensation banking and local wetland planning.

Wetlands Compensation Banking

The U.S. Fish and Wildlife Service explored the idea of "banking" in response to a number of requests to "bank" fish and wildlife habitat "credits" to offset future adverse impacts to those resources from development activity (Short, 1988). Soileau et al. (1985), offer a good description of banking:

> *"Mitigation banking is similar to maintaining a bank account. A developer undertakes measures to create, restore, or preserve fish and wildlife habitat in advance of an anticipated need for mitigation for project construction impacts. The benefits attributable to these measures are quantified, and the developer receives mitigation credits from the appropriate regulatory and/or planning agencies. These credits are placed in a mitigation bank account from which withdrawals can be made. When the developer proposes a project involving unavoidable losses of fish and wildlife resources, the losses (debits) are quantified using the same method that was used to determine credits, and a withdrawal equal to that amount is deducted (debited) from the bank. This can be repeated as long as mitigation credits remain available in the bank. . (Soileau et. al., 1985)"*

Links to the Section 404 Program

Commentators now commonly refer to the original idea of "habitat banks" as "wetland mitigation banks" because they nearly always associate "banking" with wetland mitigation requirements. Federal mitigation authority is in the National Environmental Policy Act (NEPA) and the Fish and Wildlife Coordination Act. The Council on Environmental Quality (CEQ) issued regulations implementing NEPA and clarifying mitigation in 1978. The Council regulations defined the various mitigation alternatives "to include: avoiding impacts, minimizing impacts, rectifying impacts, reducing impacts over time, and compensating for impacts" due to a major federal action including the issuance of a federal permit (Strand, 1993). The U.S. Army Corps of Engineers (COE) had authority through these statutes and regulations, and as the lead federal agency implementing the Section 404 permit program of the Clean Water Act, to require mitigation for adverse impacts to wetlands. However, no law or regulation required the COE to include mitigation as a permit condition. The CEQ mitigation alternatives were the foundation for the current "wetland mitigation sequence" established by the Section 404 (b)(1) Guidelines issued by the Environmental Protection Agency (EPA) in 1989. In 1990 the COE signed a Memorandum of Agreement (MOA) with the EPA whereby the COE adopted the Section 404 (b)(1) Guidelines approach to mitigation and agreed to attach mitigation requirements when issuing a permit to a project that adversely impacts wetlands (EPA/COE, 1990). The Section 404 (b)(1) Guidelines "mitigation sequence" involves "avoiding" and "minimizing" adverse impacts to wetlands by modifying or

redesigning the development project, and then "compensating" for unmitigated, or unavoidable, adverse impacts to wetlands as a condition of issuing a Section 404 permit. Wetland compensatory mitigation is the replacement of adversely impacted wetland functions and values by restoring, enhancing, and/or preserving existing wetlands or by creating new wetlands. Compensatory mitigation is a tool to off-set any loss of wetland functions and values due to development projects and achieve the overall national goal of "no net loss" of wetland functions and values (Conservation Foundation, 1988). The 1990 MOA also states that *"mitigation banking may be an acceptable form of compensatory mitigation under specific criteria designed to ensure an environmentally successful bank."* (I use the term "wetland compensation banking" in this paper rather than "wetland mitigation banking" because banking only involves the last step in the mitigation sequence—"compensatory mitigation.")

Major Activities

The following is a list of the major activities associated with wetland compensation banking.

1. A landscape approach to locating compensatory mitigation projects including locations that are "off-site" and "out-of-kind" from the anticipated future unavoidable, adversely impacted wetlands.
2. The restoration, enhancement, and/or preservation of existing wetlands and/or the creation of new wetlands from uplands for the purpose of compensating in advance for anticipated future unavoidable, adverse impacts to wetlands due to development activity. In most cases, wetland compensation banking activity occurs within the boundaries of the same landscape unit (watershed, sub-watershed, estuary, etc.) in which adverse impacts occur.
3. The assessment and evaluation of wetland functions and values before and after the occurrence of unavoidable, adverse impacts to wetlands to measure net loss.
4. The assessment and evaluation of wetland functions and values before and after the occurrence of compensatory mitigation projects to measure net gain.
5. The establishment of a currency system to allow exchanges between the functions and values of adversely impacted wetlands (debit wetlands) and compensatory mitigation wetlands (credit wetlands).
6. The management, maintenance, and monitoring of compensatory mitigation projects to ensure compliance with the permit conditions.
7. Legal protection "in perpetuity" for compensatory mitigation projects using in fee purchases, easement restrictions, or some other legally binding method.

Advantages and Disadvantages

Wetland compensation banking attracts both proponents and opponents. A healthy debate has ensued regarding the approach since its conception in the early 1980s. The following is a list of the major advantages and disadvantages associated with wetland compensation banking versus the traditional case-by-case approach to compensatory mitigation.

Advantages

1. Compensation projects occur in advance of unavoidable, adverse impacts to wetlands allowing regulators to measure their success over a span of years.
2. Small impacts, for which compensation is often "not practicable," can be consolidated into a larger compensatory mitigation project.
3. Large projects promote efficient "economies of scale" in financing, planning, implementing, assessing, evaluating, monitoring, managing, and maintaining compensatory mitigation projects.
4. Assessing and evaluating adversely impacted wetlands and compensatory mitigation wetlands is standardized resulting in more consistent implementation of the compensatory mitigation requirements.
5. Compensatory mitigation is linked to the functions and values of a landscape rather than limited to the specific functions and values of an adversely impacted wetland.
6. Permit certainty is increased and conflict reduced by removing parts of the compensation process from the permit process.
7. Public awareness of compensatory mitigation projects is increased because wetland compensation banks tend to be large, high profile projects.

8. Advanced compensatory mitigation "credits" offset the constitutional "takings" issue by creating value for wetlands as wetlands.

9. Advanced compensatory mitigation "credits" provide incentive for credit producers to purchase land appropriate for wetland restoration, enhancement, preservation, and/or creation when these sites are available rather than limiting land acquisition to the time period when compensatory mitigation is required.

Disadvantages

1. Mitigation planning is weakened when advanced compensatory mitigation projects create incentives to substitute "compensation" for "avoidance" and "minimization" under the mitigation sequence.

2. Wetland function and value assessment and evaluation methods are not adequate to establish an accurate currency system for the complex transactions between credit wetlands and debit wetlands.

3. Large, consolidated compensatory mitigation projects effect surrounding land use, land value, and the local tax base.

4. It is difficult to finance compensatory mitigation projects in advance unless they are linked directly to the budget of a development project.

5. Planning for a bank is expensive.

6. Potential loss of small wetlands and wetlands that are difficult to restore, enhance, or create for compensatory mitigation.

7. Off-site and out-of-kind compensatory mitigation fragments the environment.

8. Legacy of past compensatory mitigation failures. Wetland restoration, enhancement, and creation technology and science are not adequate to achieve success in compensatory mitigation projects.

Local Wetland Planning Programs

The term "local wetland planning" can describe many types of processes. This paper considers a specific type of local wetland planning process described below.

Local wetland planning is a process that balances wetland protection and development within a specific landscape area. Urban and suburban areas that contain limited upland development sites and very intense development pressure are the most likely areas for wetland planning initiatives. The approach is to allow future development in low value wetlands when an upland site is not available, while preserving high value wetlands and providing "compensatory mitigation" for any adverse impacts to low value wetlands. The intention is to create a "win-win" situation where development occurs in the least valuable environments, and wetland preservation and "compensatory mitigation" occur in the most valuable environments.

Links to the Section 404 Program

Local wetland planning links to the Section 404 program in two ways. First, the process is similar to the "practicable alternatives test" required by the Section 404 (b)(1) Guidelines issued by the EPA in 1989. The Guidelines assume that if a project is not "water dependent," meaning it does not require access to, or location within, a wetland, then there will be practicable, upland alternatives, unless clearly demonstrated otherwise (Strand, 1993). The Guidelines also assume that a practicable, upland alternative will have less adverse impact to the aquatic ecosystem unless, again, clearly demonstrated otherwise. The "practicable alternatives test" that occurs in a wetland planning process is different from the traditional test under an individual permit review. The analysis involves numerous, future development proposals rather than a single, immediate development proposal. And the analysis does not consider alternative development sites outside the boundaries of the landscape unit.

The second link between local wetland planning and the Section 404 program is the process of ranking wetland functions and values within the landscape and identifying wetland sites suitable for future development projects, wetland sites not suitable for future development, and wetland sites appropriate for future compensatory mitigation projects. The "Special Area Management Plan" (SAMP), authorized and funded under the federal Coastal Zone Management Act, can include legally binding agreements on wetland sites suitable and not suitable for future development that can be used to either support or deny a Section 404 permit application. The EPA provides funding for wetland surveys, assessments, and evaluations through its "Advanced Identification" program. However, the COE and EPA do not have to abide by these determinations when reviewing a Section 404 permit application unless the agencies sign a

legal agreement such as a SAMP or a MOA. The purpose of the determinations is to facilitate future Section 404 permit application reviews.

Major Activities

Federal, state, county, city, and/or tribal governments as well as non-governmental groups sponsor and participate in local wetland plans. Local wetland planning usually involves relatively small landscape scales such as a single watershed, a sub-watershed, a river corridor, a greenway, an estuary, or a complex of wetlands. The following is a list of the major activities associated with local wetland planning.

1. Determine the boundaries of the landscape planning unit.
2. Identify wetlands in the landscape unit.
3. Assess and evaluate the functions and values of the wetlands before the occurrence of potential development or compensatory mitigation projects.
4. Rank wetlands based on their functions and values.
5. Prioritize wetlands for preservation, restoration, and enhancement based on their ranks (some of these sites may be identified as appropriate for compensatory mitigation projects).
6. Prioritize wetlands suitable for future development based on their ranks (include consideration of "practicable alternatives" to locating a development in a wetland).
7. Analysis of present land ownership and zoning patterns for potential land swaps, transactions, and/or re-zoning that supports the wetland planning effort.
8. Integrate other land use and resource management programs into the wetland planning process (transportation, surface water management, flood management, fish and wildlife management, etc.).
9. In some cases, establish a streamlined permit process for regulated activities that are in accordance with the wetland plan.

Advantages and Disadvantages

Local wetland planning also attracts proponents and opponents. The following is a list of the major advantages and disadvantages associated with local wetland planning versus the traditional case-by-case approach to reviewing proposed development projects that adversely impact regulated wetland area and determining compensatory mitigation requirements.

Advantages

1. Future development, compensatory mitigation, and wetland protection decisions are based on analysis that considers landscape scale processes.
2. Increased certainty and predictability for both development and environmental protection.
3. Increased consistency and efficiency in the permit review process.
4. Creative opportunities to create "win-win" solutions by swapping, purchasing, and/or re-zoning property.
5. Cooperation and coordination between programs with overlapping and/or complementary goals and objectives.
6. Planning includes local needs, conditions, and perspectives.

Disadvantages

1. The boundaries of the landscape unit limit the "practicable alternatives test." Local wetland planning usually does not consider an alternative development site outside the landscape planning unit.
2. The "practicable alternatives test" is more speculative when applied in a planning context. It is difficult to predict future development demands and needs, especially over the 20 year planning period often used in planning processes.
3. Identifying wetlands that rank as "low" value wetlands suitable for future development reduces incentives for developers to "avoid" and "minimize" adverse impacts to "low" value wetlands.
4. The public can only review an anticipated future development project during the planning process rather than the actual development project during the permit review process.
5. Legacy of past compensatory mitigation failures. Wetland restoration, enhancement, and creation technology and science is not adequate to achieve success in compensatory mitigation projects.

Integrating Wetland Compensation Banking Into Local Wetland Planning

It is a recurring theme in the wetland compensation banking literature that the future of banking will link with local wetland planning processes (Short, 1988; Castelle, 1992; Kusler and Lassonde, 1992; Reppert, 1992; World Wildlife Fund, 1992; Dennison and Berry, 1993; McKenzie and Rylko, 1993; National Association of Industrial and Office Parks, 1993; ELI, 1994; Brumbaugh and Reppert, 1994; Silverstein, 1994; Association of State Wetland Managers, 1995; COE et. al., 1996). Successful compensatory mitigation is more likely when projects recognize the system in which they are located (National Research Council, 1992). Planning is the best tool wetland regulators and developers have to locate compensatory mitigation projects in a manner that is sensitive to the surrounding ecological systems, land-use practices, and long–term social goals. "It is sometimes said about real estate that value depends upon location, location, location. Similarly, the importance of wetland functions and values also depend upon location, location, location" (Kusler, 1992). Coordinating the activities of wetland compensation banking with local wetland planning provides opportunities to reduce the costs of wetland protection, increase regulatory consistency, improve wetland functions and values at the landscape scale, and facilitate local community goals, conditions, and perspectives.

Overlapping and Complementary Activities

Many of the activities associated with wetland compensation banking and local wetland planning, directly overlap or complement one another. An overlapping activity is one that occurs in both approaches; a complementary activity is one that occurs in only one of the approaches, but it facilitates the goals of the other approach. Integrating the two approaches would reduce the replication of the overlapping activities and facilitate the complementary activities. The following are lists of overlapping and complementary activities between wetland compensation banking and local wetland planning.

Overlapping Activities

1. Landscape scale analysis to identify of wetland sites for restoration, enhancement, and preservation.
2. Assessment and evaluation of the functions and values of wetland sites before the occurrence of development impacts.
3. Assessment and evaluation of the functions and values of wetland sites before the occurrence of restoration, enhancement, and/or preservation projects for compensatory mitigation.
4. Establishment of a currency system to allow complex exchanges between the functions and values of adversely impacted wetlands and compensatory mitigation wetlands.
5. Planning and analysis confined to a landscape unit (watershed, sub-watershed, estuary, etc.).

Complementary Activities

1. Wetland banking's actual restoration, enhancement, and/or preservation projects for compensatory mitigation.
2. Wetland banking's assessment and evaluations of wetland functions and values after the occurrence of development impacts.
3. Wetland banking's assessment and evaluation of the functions and values of wetland sites after the occurrence of restoration, enhancement, and/or preservation projects for compensatory mitigation.
4. Wetland banking's long-term management, maintenance, and monitoring of compensatory mitigation projects.
5. Local wetland planning's determination of the boundaries of the landscape planning unit.
6. Local wetland planning's identification of wetlands in the landscape planning unit.
7. Local wetland planning's ranking of wetlands based on their functions and values.
8. Local wetland planning's analysis of present land ownership and zoning patterns for potential land swaps, transaction, and re-zoning.
9. Local wetland planning's integration of other land use and resource management programs.
10. Local wetland planning's establishment of a streamlined permit process for regulated activities that are in accordance with the wetland plan.

Discussion of Overlapping and Complementary Activities

Obviously, it would be a waste of effort for a wetland banking project and a local wetland planning process to produce 2 landscape analyses, 2 assessments and evaluations of wetland functions and values, and 2 currency systems for the same wetlands in the same landscape unit. Not only would that result in a waste of time and money, but it would lead to inconsistent and perhaps conflicting results.

Imagine a situation in which a wetland compensation bank moves, independent of local consideration, to purchase a degraded wetland area along a river corridor that is deemed appropriate for restoration and enhancement by its own analysis. Imagine the same wetland area is considered ideal for future development by a local wetland planning process. In fact, the area is zoned for development and is considered an excellent location for shipping and water dependent type activities. The local community needs this type of economic investment to provide jobs. The local government is also aware that the amount of land zoned for development is shrinking and they can not afford to loose the tax revenues. This scenario is not so imaginary. Several county governments in Washington State mentioned this type of concern when they were asked to review and comment on the Washington State Department of Transportation Wetland Banking Agreement (Hershman and Green, 1995). Combining overlapping activities between wetland banking and planning is a very effective way to reduce costs, improve regulatory consistency, and, perhaps most importantly, provide for coordination and cooperation between potentially conflicting activities at the local level.

Complementary activities present a similar opportunity. There are many activities associated with wetland compensation banking that do not occur in local wetland planning and vice versa. Coordinating these activities can also reduce conflicts and improve the success of both approaches. For example, local wetland planning processes identify all wetlands in a landscape unit and ranks each according to their functions and values relative to other wetlands in a landscape unit. This is more thorough than the analysis that is likely to occur in a wetland banking project. Conflicts between the two approaches would be reduced if a wetland banking project participated and/or adhered to the decisions of a local wetland plan. Complementary activities also represent an extra tool for both approaches. For instance, the actual wetland restoration, enhancement, creation, and preservation efforts that occur as compensatory mitigation in a banking framework directly accomplish objectives of a local wetland plan to protect and improve the wetland resources in the landscape unit. The streamlined permit process for development projects that are consistent with the local wetland plan directly improves the objectives of a banking project to increase permit certainty and predictability for developers.

It is possible for complementary activities to overcome a disadvantage associated with a single approach implemented independently of the other approach. A good example of this is when a local wetland planning process reduces the potential for a wetland banking project to negatively impact surrounding land use, land value, and the local tax base; to fragment the environment by using "off-site" or "out-of-kind" compensatory mitigation; or to accumulate large planning expenses. These are included in the list of disadvantages when using a wetland compensation banking approach. Complementary activities are an opportunity to improve the success of both approaches.

Advantages and Disadvantages

Integrating wetland compensation banking with local wetland planning presents a number of advantages and a few disadvantages. The following is a list of the most important advantages and disadvantages of integrating the two approaches versus implementing them independently.

Advantages

1. Combining overlapping activities reduces costs and increases regulatory consistency.
2. Complementary activities improve both approaches in some cases overcoming a disadvantage associated with a single approach implemented independent of the other.
3. Compensatory mitigation projects occur in the most appropriate sites and target priority functions and values for the broader ecosystem.

Disadvantages

1. Lack of regulatory guidance and experience for implementing wetland compensation banking and local wetland planning.

2. Legacy of past compensatory mitigation failures. Wetland restoration, enhancement, and creation technology and science are not adequate to achieve success in compensatory mitigation projects.

Discussion of Advantages and Disadvantages

Wetland compensation banking and local wetland planning, implemented independent of one another, have advantages and disadvantages that are listed earlier in this paper. It is a different issue to consider the advantages and disadvantages of integrating these approaches versus implementing them separately. I already discussed the three advantages listed above, the disadvantages listed above require some clarification. Currently, there is very little regulatory guidance or experience for integrating wetland compensation banking with local wetland planning. An interagency work group just recently issued formal guidance for establishing wetland compensation banks (COE et. al., 1995). The northwest regional offices of the COE and EPA are working on guidance for local wetland planning (personal communication with COE staff in the Seattle District Office). Guidance documents clarify how these approaches should be structured so they comply with the Section 404 program requirements. It will be difficult to integrate wetland banking and planning until guidance exist for both. Even with guidance, integrating the two approaches will require an adaptive learning period before enough experience accumulates to avoid the hidden pitfalls in innovative approaches.

Finally, the legacy of past compensatory mitigation failures is inescapable. Wetland restoration, enhancement, and creation technology and science are not adequate to produce the kind of results necessary to achieve "no net loss" of wetland functions and values within the compensatory mitigation framework. A recent series of papers presented as a "Forum" in *Ecological Applications* is an excellent treatment of the subject (the most applicable were: Zedler, 1996; Race and Fonseca, 1996; Mitsch and Wilson, 1996).

Research Needs

A number of areas of research will help improve the successful integration of wetland compensation banking with local wetland planning. First, both approaches rely on landscape analysis methods to prioritize wetland areas appropriate for restoration, enhancement, or preservation within a specified landscape unit such as a watershed. A greater understanding of how landscape ecology applies to wetlands will be very useful. Issues such as the most useful scale of analysis for wetlands should be determined (watershed, migratory pathways, material flow pathways, etc.). Second, both approaches rely on assessment and evaluation methods to measure wetland functions and values. Methods need to be established that are both flexible enough to apply in different regions and rigid enough to provide consistency within regions. These methods must also be lay person "friendly" at some level to provide for public participation. They should also be transferable between compensatory mitigation activities at the local wetland planning stage and during the actual implementation stage within a wetland banking framework. Third, regulators and developers need to know exactly how to comply with the regulatory requirements of the Section 404 program while experimenting with innovative approaches like wetland banking and planning. Finally, further understanding of wetland restoration, enhancement, and creation technology and science will greatly improve the odds for compensatory mitigation planning and implementation success.

Research will not answer these needs overnight. Ongoing wetland compensation banking projects and local wetland planning processes, and efforts to integrate the two approaches, should be designed as experiments to produce learning opportunities. Future efforts can adapt and improve based on the results from the earlier experiments (Lee, 1993). This "adaptive management" approach is the best way to move forward with wetland compensation banking and local wetland planning, conducting important research to improve the two approaches, and "taking a broader view" while implementing the Section 404 program requirements.

Literature Cited

Association of State Wetland Managers. 1994. Effective Mitigation: Mitigation Banks and Joint Projects In the Context of Wetland Plans. Association of State Wetland Managers, Berne, NY.

Association of State Wetland Managers. 1995. Watershed Management and Wetland Ecosystems: Implementing Fair, Flexible, And Effective Approaches, Background Report. Wetlands '95

National Symposium in Tampa Bay, Florida. ASWM, Berne, NY.

Brumbaugh, Robert and Richard Reppert. 1994. First Phase Report. U.S. Army Corps of Engineers, Institute for Water Resources, IWR Report 94-WMB-4, Alexandria, VA.

Castelle, Andrew et. al. 1992. Wetlands Mitigation Banking, Washington State Department of Ecology, Publication #92-12, Olympia, WA.

Conservation Foundation. 1988. Protecting America's Wetlands: An Action Agenda. National Wetlands Policy Forum. Washington, DC.

Dennison, Mark S. and James F. Berry. 1993. Wetlands: Guide to Science, Law, and Technology. Noyes Publications. Park Ridge, NJ.

Environmental Law Institute. 1994. Wetland Mitigation Banking. U.S. Army Corps of Engineers. Institute for Water Resources. IWR Report 94–WMB–6. Alexandria, VA.

Hershman, Marc and William J. Green. 1995. The Washington State Department of Transportation Memorandum of Agreement for Wetland Compensation Banking: County and Tribal Participation. Washington State Transportation Center. Seattle, WA.

Kusler, Jon A. 1992. Mitigation Banks and the Replacement of Wetland Functions and Values. In Effective Mitigation: Mitigation Banks and Joint Projects In the Context of Wetland Management Plans. Association of State Wetland Managers, Berne, NY.

Kusler, Jon A. and Cindy Lassonde. 1992. Effective Mitigation: Mitigation Banks and Joint Projects In the Context of Wetland Management Plans. The Association of State Wetland Managers, Berne, NY.

Lee, Kai N. 1993. Compass and Gyroscope: Integrating Science and Politics for the Environment. Island Press. Washington, D.C.

McKenzie, Tracy P. and Micheal Rylko. 1993. Partnerships in Restoration Mitigation Banking. In Partnerships and Opportunities in Wetland Restoration. Fred Weinman and et. al. (eds) U.S. Environmental Protection Agency, Region 10, Seattle, WA.

Mitsch, W. J. and Renee F. Wilson. 1996. Improving the Success of Wetland Creation and Restoration with Know-How, Time, and Self-Design. Ecological Applications 6(1):77-83.

National Association of Industrial and Office Parks. 1993. Wetland Incentives: Non–Regulatory Approaches to Protecting Wetlands. Puget Sound Water Quality Authority: Olympia, WA.

National Research Council. 1992. Restoration of Aquatic Ecosystem. National Academy Press. Washington, DC.

Race, M. and M. S. Fonseca. 1996. Fixing Compensatory Mitigation: What Will It Take? Ecological Applications 6(1):94-101.

Reppert, Richard. 1992. Wetland Mitigation Banking Concepts. U.S. Army Corps of Engineers, Institute for Water Resources, IWR Report 92-WMB-1, Alexandria, VA.

Short, Cathleen. 1988. Mitigation Banking. U.S. Department of Interior, Fish and Wildlife Service, Washington, DC.

Silverstein, Jonathon. 1994. Taking Wetlands to the Bank: The Role of Wetland Mitigation Banking in a Comprehensive Approach to Wetlands Protection. Environmental Affairs 22:129-161.

Soileau, et. al. 1985. Mitigation Banking: A Mechanism for Compensating Unavoidable Fish and Wildlife Habitat Losses. In Transcripts from the North American Wildlife and Natural Resource Conference. pp. 465-474.

Strand, Margaret N. 1993. Federal Wetlands Law. In The Environmental Law Reporter: Wetlands Deskbook. Environmental Law Institute. Washington, DC. pp. 1-107.

U.S. Army Corps of Engineers, et. al. 1995. Federal Guidance for the Establishment, Use and Operation of Mitigation Banks; Notice. Federal Register 60(228):58605-58614, November 28, 1995.

U.S. Environmental Protection Agency and the U.S. Army Corps of Engineers. 1990. Memorandum of Agreement Between the Environmental Protection Agency and the Department of the Army Concerning the Determination of Mitigation Under the Clean Water Act Section 404(b)(1) Guidelines. pp. 331-336, In The Environmental Law Reporter: Wetlands Deskbook. Environmental Law Institute. Washington, DC. pp. 331-336.

Washington State Department of Transportation 1994. Washington State Department of

Transportation Wetland Compensation Bank Program Memorandum of Agreement. Olympia, WA. September 15, 1994.

Zedler, J. B. 1996. Ecological Issues in Wetland Mitigation: An Introduction to the Forum. In Ecological Applications 6 (1): 33-37.

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Watershed Analysis as a Tool in Watershed Restoration

Steven Toth[1]

Abstract: **Watershed analysis in Washington State was developed by constituents of the Timber Fish and Wildlife (TFW) agreement (Native American tribes, state agencies, environmental groups, and the timber industry). The methodology addresses the cumulative effects of forest practices on fish and water resources in the context of natural ecological processes. While other watershed analysis methodologies have been developed recently (e.g., Federal Agencies Guide, Idaho), TFW watershed analysis is unique in its scientific rigor and ability to affect land management/restoration decisions. Close interaction between scientists and policy makers allowed development of a method that is repeatable, defensible, and accountable to constituents.**

The watershed analysis process consists of three components: resource assessment, prescriptions and monitoring. An interdisciplinary team of scientists with expertise in geology, hydrology, fish biology, soils, and forest ecology assesses the physical and biological components of the watershed. Natural disturbance processes (e.g., floods, fire) and human-related disturbance (e.g., timber harvest, grazing) are evaluated in the context of present land management activities. The evaluation relies primarily on aerial photographs, monitoring data, landform maps and geographic information systems, with some additional field work. The team delineates specific areas on the landscape that are sensitive to management practices and have the potential to impact public resources. The resource assessment and synthesis of information for watersheds ranging in size from 100 to 200 square kilometers and takes approximately 2 to 3 months.

In the prescriptions phase, land managers work with scientists to develop options for operating in sensitive areas. Potential restoration actions are also identified and prioritized to improve or restore aquatic habitat conditions. The entire watershed is addressed regardless of administrative boundaries. The direct connection between the scientists and land managers ensures that information generated by the assessment team is at a scale appropriate for guiding management decisions in the field.

Finally, a monitoring program is developed to evaluate the implementation of the prescriptions, test assumptions about watershed process, and assess recovery of aquatic resources. Monitoring priorities are developed jointly with stakeholders in the watershed. Preservation and restoration of ecological processes are evaluated and prioritized using the information generated from the watershed analysis. An example of a watershed analysis conducted in the central Cascade range of Washington state is used to illustrate the assessment of a watershed, development of prescriptions, and monitoring of watershed conditions.

Introduction

Strategies for protecting aquatic resources while allowing land management activities must be sufficiently flexible to accommodate landscape variability in the Pacific Northwest (FEMAT 1993). Methodologies for conducting watershed analysis in forested watersheds have been developed by Federal agencies (U.S. Department of Agriculture 1994), the state of Washington (Washington Forest Practices Board 1995), and the state of Idaho (Idaho Department of Lands 1995) to link land management with scientific assessments. These procedures are also being used on private lands in the states of California, Oregon and Montana and in British Columbia, Canada.

Although many similarities exist between these procedures, varying objectives have led to different methodologies for assessing watersheds. The federal watershed analysis process characterizes human,

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aquatic and terrestrial conditions and is more appropriately described as ecosystem analysis at the watershed scale. The federal analysis is driven primarily by issues identified prior to the start of the analysis. A list of approaches is provided for the interdisciplinary team of resource specialists, but no specific methodology has to be followed. The analysis is not a decision-making process, but establishes the context for subsequent planning (Regional Interagency Executive Committee 1995).

The Washington State and Idaho processes focus primarily on aquatic resources and the physical processes that influence aquatic habitat, but the Washington state process has more rigorous scientific analysis. Both processes have tightly choreographed methodologies that the analysts must follow. The Washington state process, however, requires scientific assessment of watershed processes by an interdisciplinary team of certified specialists. The Idaho process is designed to be implemented by a single trained resource manager. The Idaho analysis focuses on inherent hazards within a watershed and current stream conditions, rather than characterizing processes. Both procedures directly influence decision-making through development of management prescriptions based on the watershed assessment.

Many people have looked to watershed analysis as a means for guiding and implementing watershed restoration, despite the fact that most of these processes were not explicitly developed to meet this objective. For watershed analysis to be useful in development of restoration plans, data must be gathered at a scale that can be utilized at the project level. In particular, any in-stream or riparian restoration work requires an understanding of watershed processes such as flooding history, sediment supply, and geomorphic context at a local scale. Of the three procedures described previously, the Washington state process is probably the best suited for prioritizing and implementing land management, monitoring and restoration projects because of its more rigorous use of scientific analysis and ability to affect management/restoration decisions. This paper outlines the general methods for conducting Washington State watershed analysis, describes restoration approaches, and examines how information from watershed analysis can be used to prioritize and implement restoration projects.

Washington State Procedure for Watershed Analysis

Watershed analysis is a regulatory process administered by the Washington State Department of Natural Resources (DNR) on state and private land ownership in Washington. The analysis is designed to address the cumulative effects of forest practices on the public resources of fish, water, and capital improvements (e.g., bridges and county roads). A watershed analysis can be initiated by either the DNR or voluntarily by a private landowner who owns more than 10 percent of a watershed. Watersheds range in size from approximately 100 to 200 km^2.

The Washington state watershed analysis procedure consists of four distinct components:

1. Resource Assessment: Scientists identify hillslope hazards by assessing mass wasting, surface erosion, hydrology, and riparian condition. They also identify vulnerable resources by assessing fish habitat, stream channels, water quality, and capital improvements (Table 1). Sensitive areas of the watershed are delineated where hillslope hazards can affect a vulnerable resource (e.g., an area prone to landslides that delivers sediment to a fish-bearing stream).
2. Prescriptions: Land managers and scientists design prescriptions for each sensitive area. Prescriptions are methods for operating in sensitive areas to reduce or eliminate potential problems. Standard forest practice rules are applied on the remainder of the watershed.
3. Public Review: The public is given the opportunity to review and comment on the findings through the State Environmental Policy Act (SEPA).
4. Monitoring: A monitoring plan is developed to track changes in watershed conditions and test the effectiveness of prescriptions. Monitoring is voluntary, although most landowners have initiated some monitoring following completion of the analysis.

Watershed analysis was developed by scientists and managers from state agencies, Indian tribes and the timber industry working cooperatively under the state Timber/Fish/Wildlife agreement. Watershed analysis creates additional forest practice rules tailored to specific watersheds using a science-based assessment procedure. The assessment is typically conducted in two to three months by an interdisciplinary team of

Table 1. Summary of Watershed Processes and Resources Addressed by the Washington State Watershed Analysis Modules

Watershed Analysis Module	Watershed Processes and Resources Addressed
Mass Wasting	• Debris Torrents • Landslides • Earthflows
Surface Erosion	• Hillslope Surface Erosion – Gullying – Dry Ravel – Sheetwash • Road Erosion
Hydrology	• Peak Streamflows • Summer Low Flows
Riparian Function	• Large Woody Debris Recruitment • Shade/Water Temperature • Bank Stability
Channel Condition	• Historic Channel Disturbance • Current Channel Condition • Spatial Distribution of Channel Response Types • Dominant Habitat Forming/Geomorphic Processes
Fish Habitat	• Distribution and Relative Abundance of Salmonid Fish • Existing Habitat Condition • Fish Habitat Utilization and Preferences
Water Supply / Public Works	• Location and Sensitivity of Water Supplies/Public Works – Public State Roads and Bridges – Reservoir, Irrigation Structures – Municipal, Domestic, Hatchery Water Supplies

Table 2. Management Response for Areas of Resource Sensitivity

		Likelihood of Adverse Change and Deliverability		
		Low	*Medium*	*High*
Resource Vulnerability	**Low**	Standard Rules	Standard Rules	Prevent
	Medium	Standard Rules	Minimize	Prevent
	High	Standard Rules	Prevent	Prevent

natural resource specialists that are certified by the DNR. The team may include geomorphologists, hydrologists, soil scientists, biologists, and other specialists as needed. The assessment provides information on physical and biological processes within a watershed such as the spatial distribution of processes, recent changes in the condition of the watershed, and how these changes influence aquatic resources.

The resource assessment information is passed onto a prescriptions team that is typically composed of land managers and some of the specialists that participated in the resource assessment phase. The prescription team develops various options for operating in sensitive areas to provide as much flexibility for landowners as possible while still protecting public resources. The assessment report and prescriptions must be approved by the DNR and go through public review. The entire watershed analysis process typically takes a full year to complete.

Monitoring is a critical component of watershed analysis because there are often limitations in the scientific assessment and many of the prescriptions are experimental in nature. Guidelines for developing monitoring plans are provided in the watershed analysis manual (Washington Forest Practices Board 1995); however, monitoring is currently not required of landowners. A review of the watershed analyses, though, is required every five years and should provide information on the effectiveness of prescriptions in protecting resources. Analyses may also be reviewed sooner if fish habitat degrades, prescriptions are not working or new scientific information is developed.

Washington State watershed analysis is designed to create options for conducting timber management activities in a manner that maintains natural rates of sediment input from hillslopes, minimizes potential changes in streamflow and provides adequate riparian corridors for maintaining temperature, large woody debris and nutrient levels. The following sections describe in more detail the procedures used to evaluate these watershed processes and relating the processes to regulatory prescriptions. For a complete listing of procedures, refer to the Washington State Watershed Analysis Manual (Washington Forest Practices Board 1995).

Sediment

Mass wasting is the dominant source of sediment in many Pacific Northwest watersheds (Swanston and Dyrness 1973, Sidle et al. 1985, Megahan 1983). While landslides often initiate far above fish-bearing waters, debris flows, dam-break floods and fluvial processes can route sediment downstream and affect fish habitat far below the landslide (Swanston et al. 1987). Both the volume and rate of sediment influx is considered in light of its effects on fish habitat.

Delivery of coarse sediment to streams is addressed primarily by the Mass Wasting module. The mass wasting analyst maps all historical mass wasting within the watershed as defined by the aerial photograph record and evaluates the sediment delivery potential to streams. A landslide inventory is produced and correlations are made with a number of variables including geology, slope gradient, slope form, and forest practices. The analyst uses these data to produce a mass wasting hazard map that identifies areas with the potential for mass wasting.

Another significant concern is fine sediment (less than 2 mm) from mass wasting, hillslope erosion, and road erosion that can reduce the viability of eggs in spawning gravels, reduce rearing habitat by filling the interstitial spaces of cobbles and gravel, and increase turbidity and nutrient levels in streams (Lisle and Hilton 1992, Lisle 1989, Chapman 1988). Fine sediment input into streams is addressed by the Surface Erosion and Mass Wasting modules. The surface erosion analyst evaluates recent harvest units and road construction in the field to examine the potential for gullying and sheetwash erosion. The road network is also evaluated using an empirically-based model to estimate sediment production from roads. Predicted sediment yields are based primarily on road characteristics including traffic levels and road surfacing. Management-derived sediment inputs are compared to estimates of natural sediment input rates as part of a crude sediment budget to evaluate hazards. In this context, estimates of sediment input are useful for estimating relative contributions from the various sources of sediment in the watershed. Other sources of fine sediment from land use activities such as grazing and agriculture as well as episodic natural events such as fire are considered in the context of inputs from timber management activities.

Hydrology

Removal of forest cover can increase snow accumulation, allow for greater wind speeds, and increase solar radiation, thereby increasing the amount of snowmelt, especially during rain-on-snow events (Coffin and Harr 1992). Increasing the size and frequency of flood flows from timber harvest is a concern because larger and more frequent flood flows can change channel morphology through increased sediment and large woody debris transport and greater bank erosion. Larger and more frequent flood flows can also increase the depth of gravel bed scour, potentially scouring fish redds.

The hydrology module primarily addresses changes in peak flows during rain-on-snow events. The analysis procedure for peak flows consists of evaluating canopy coverage in relation to elevation. A modified U.S. Army Corps of Engineers (1956) empirical snowmelt model is used to estimate the increase in water available for runoff during various magnitude storm events. The water available for runoff is then related to streamflow using either gage data, field estimates of flood flows, or regional regression equations.

Riparian Corridors

Large Woody Debris

Large woody debris (LWD) is vital for maintaining fish habitat in most Cascade streams. LWD dissipates stream energy, influences sediment storage and transport, and provides habitat and cover for fish both directly and through changes in channel morphology (i.e., pool formation)(Bilby 1984, Bisson et al. 1987)

The riparian module addresses large woody debris recruitment to streams primarily by assessing the condition of riparian areas within 20 to 30 meters on either side of streams. This distance equates to approximately one-half the site potential tree height and generally accounts for 85 to 100 percent of the natural woody debris input to streams (McDade et al. 1990, Robison and Beschta 1990, Murphy and Koski 1989). Forest stand type (coniferous, mixed or deciduous), relative age, and density are measured for each stream reach from aerial photographs. The assessment concentrates on fish-bearing streams, but non-fish-bearing streams less than 20 percent gradient are also considered.

The potential for future LWD recruitment is considered together with the amount of present in-channel LWD to address concerns about present and future timber management. In-channel LWD data is collected in the field jointly with the channel and fish module team members. The riparian analyst interacts with the channel analyst to assess channel recruitment mechanisms such as meandering channels, debris flows and bank erosion. The width of riparian recruitment can, thus, be extended beyond 20 to 30 meters depending on site potential tree height and channel processes.

Temperature

Salmonids require relatively cool stream temperatures for all life history stages. During the summer when stream temperatures increase, cool temperatures are especially important for rearing juveniles and spawning adults. Empirical evidence in Washington shows that there is a strong relationship between canopy cover, elevation and stream temperature (Sullivan et al. 1990). This has led to the development of a temperature screen that specifies the amount of canopy coverage needed at a given elevation to meet or exceed state water quality standards for stream temperature.

The riparian module analyst uses aerial photographs to assess canopy cover and topographic maps to establish elevations. All fish-bearing streams as well as non-fish-bearing streams that contribute at least 20 percent of the flow to a fish-bearing stream are considered (Caldwell et al. 1991). At given elevation zones, targets for canopy coverage have been established through the state forest practice rules.

If canopy cover meets specified target levels, the stream is assumed to be below state maximum water temperature criteria at that location. If canopy cover does not meet specified target levels, it is assumed that maximum water temperature standards are exceeded. Available temperature data can be used to identify or verify problem areas.

Channel Condition

The Channel module assesses past changes in channel morphology and processes, current channel conditions, and the potential sensitivity of channels to changes in inputs of sediment, wood, and water. Channel morphology reflects and integrates processes operating in a watershed because material eroded from hillslopes ultimately is delivered to and routed through the channel network. Channel and fish

habitat condition, therefore, reflect the relative input of sediment, wood and water relative to the ability of the channel to either transport or store these inputs.

The stream channel network is stratified into geomorphic units. Geomorphic units are reaches of streams that respond in a similar fashion based on comparable channel-forming processes. Alluvial fans or steep-gradient canyon tributaries would be examples of geomorphic units. These channel-forming processes are derived from the general geology and climate which dictates stream gradient and confinement and hillslope topography and vegetation. Geomorphic units allow assessment of channel conditions on a watershed basis and provide a context for evaluating the influence of land management activities. These units become the basis for linking hillslope and channel processes during the synthesis portion of watershed analysis.

Fish Habitat

It is difficult to assess all the factors that affect salmonid production because of their wide-ranging life histories. Anadromous fish populations can vary substantially simply based on factors outside of their freshwater life history phase. For this reason, the fish habitat module assumes that evaluation of physical stream habitat characteristics will provide an adequate measurement of salmonid production during their freshwater life history phase. It is assumed that degradation of physical habitat features will result in reductions in salmonid production.

Two basic premises of the fish habitat evaluation are that: (1) physical habitat characteristics are strongly influenced by geomorphic setting, and (2) old-growth conditions most closely represent the conditions to which multiple species have adapted over the past several thousand years. This approach does not imply that preferred fish habitat only occurs in old-growth forests, but that knowledge of habitat in old-growth forests can form the basis for identifying changes in habitat conditions (Peterson et al. 1992). Indices of habitat conditions are based on habitat utilization and on stream characteristics that have supported a multitude of species prior to human-induced habitat changes. Physical habitat features which are evaluated include depth and velocity ranges (grouped as channel units such as pools and riffles), pool frequency, pool size, cover, spawning gravels, and temperature ranges. Other elements of fisheries evaluated by the module include historic and current salmonid fish distribution, relative abundance of salmonids, and an assessment of factors limiting fish production.

Synthesis of Channel Condition and Fish Habitat

The channel and fish module analysts jointly determine the vulnerability of fish habitat to changes in physical processes based on local channel conditions and fish biological requirements. The geomorphic units form the basis for assessing vulnerability because channel forming processes are assumed to be similar within a geomorphic unit. Vulnerability is assessed for processes such as: (1) debris torrents (debris flows and dam-break floods); (2) increases in coarse sediment; (3) increases in fine sediment; (4) changes in hydrology (primarily peak flows); (5) decreases in large woody debris and shade; and (6) removal of near-bank riparian vegetation.

The vulnerability of fish habitat is rated "high," "medium," or "low" based on the channel sensitivity to a change in these processes and the potential impact on fish production. A channel geomorphic unit may not be particularly sensitive to a given input process, but if there is concentrated fish use (e.g., a chum spawning reach) or limited habitat availability within that unit (e.g., a single area that accounts for most of the coho winter rearing habitat), the vulnerability rating is changed to reflect the unit's importance for fish production.

Synthesis, Causal Mechanism Reports, and Prescriptions

An integral part of conducting watershed analysis is linking hillslope and channel processes. This may involve routing sediment derived from management-induced landslides through the channel network or discussing the role of wildfire in producing surface erosion and its consequent effect on fish production.

During the synthesis process, the resource assessment team identifies areas of resource sensitivity based on the likelihood of adverse change and deliverability to vulnerable resources. Some hazard areas identified by the resource assessment team may not be considered resource sensitive areas if impacts cannot be delivered to the resource of concern (e.g., unstable slopes that do not deliver sediment to streams). The resource sensitive areas are designated

relative to the hazard area, rather than to the stream segments with the affected resource.

The resource assessment team rates both the likelihood of adverse change and the resource vulnerability as "low," "medium," or "high." These ratings are placed in a management response matrix (Table 2) to provide direction on which situations need prescriptions and the standard to which prescriptions will be written. There are three potential management responses (rule calls): (1) standard forest practice rules; (2) a prescription that minimizes potential impacts to the resource of concern; and (3) a prescription that prevents impacts that would damage or prevent the recovery of public resources. While the ratings end up simplifying complex technical problems, it provides an important link between scientific assessment and regulatory policy.

A primary tool for synthesizing the results of the resource assessment is the causal mechanism report. A causal mechanism report is produced for all areas of resource sensitivity. These reports provide a brief and focused summary of problem areas that can be used easily by the prescriptions team. The causal mechanism report summary has a brief problem statement, identifies linkages between land management and resources of concern, and provides additional comments to help guide the prescription team in developing appropriate options for timber management.

The prescription team develops management options consistent with the standard of protection required by the rule call in each causal mechanism report. The team must provide justifications for each prescription to ensure that the standards for protection are met. The DNR also provides technical review of the documents prior to initiation of the SEPA process to confirm that the watershed analysis meets regulatory standards.

Monitoring

Generally, three types of monitoring are employed upon completion of watershed analysis; validation, trend and effectiveness monitoring (MacDonald et al. 1991). Validation monitoring assesses the scientific validity underlying elements of the resource assessment. Examples of validation monitoring might include testing of empirical models with local data or evaluating assumptions about woody debris input. Trend monitoring assesses resource conditions over time. Examples of trend monitoring could include collecting large woody debris data over time or measuring streamflow. Effectiveness monitoring evaluates whether prescriptions produced the desired result. Examples of effectiveness monitoring could include evaluation of prescriptions for road construction near unstable slopes or silvicultural prescriptions in riparian areas.

The watershed analysis manual contains a monitoring module to assist in the development of a monitoring program. The module is designed to assist in prioritizing monitoring projects and to develop specific monitoring objectives and time frames for evaluating the effectiveness of the watershed analysis process. The monitoring program is typically developed in conjunction with all stakeholders in the watershed.

Quartz Mountain Watershed Analysis

In order to better understand how watershed analysis is implemented, the following example outlines the results of a watershed analysis in the Quartz Mountain watershed administrative unit (WAU). The Quartz Mountain watershed analysis was initiated by Plum Creek Timber Company in September 1993. The watershed analysis team was comprised of representatives from Plum Creek, Yakama Indian Nation, Department of Ecology, Department of Natural Resources, U.S. Forest Service, and the Washington Environmental Council. The entire watershed analysis process was open to public participation. A detailed description of the analysis is contained in the *Quartz Mountain Watershed Analysis Report* (1994).

Overview of the Watershed

The Quartz Mountain WAU consists of the area drained by the North and South Forks of Taneum Creek to the point of their confluence (Figure 1). The Taneum Creek watershed is located in mountainous forestland east of the Cascades crest, approximately 2 km (1.24 miles) southeast of the town of Cle Elum, in Kittitas County, Washington. The upper Taneum Creek watershed area is approximately 29,400 acres (46 mi^2). About 40 percent of the watershed is owned by Plum Creek with the remainder administered by the U.S. Forest Service.

Elevations in the watershed range from approximately 853 m (2,800 feet) at the confluence of the North and South Forks of Taneum Creek to 1,917 m

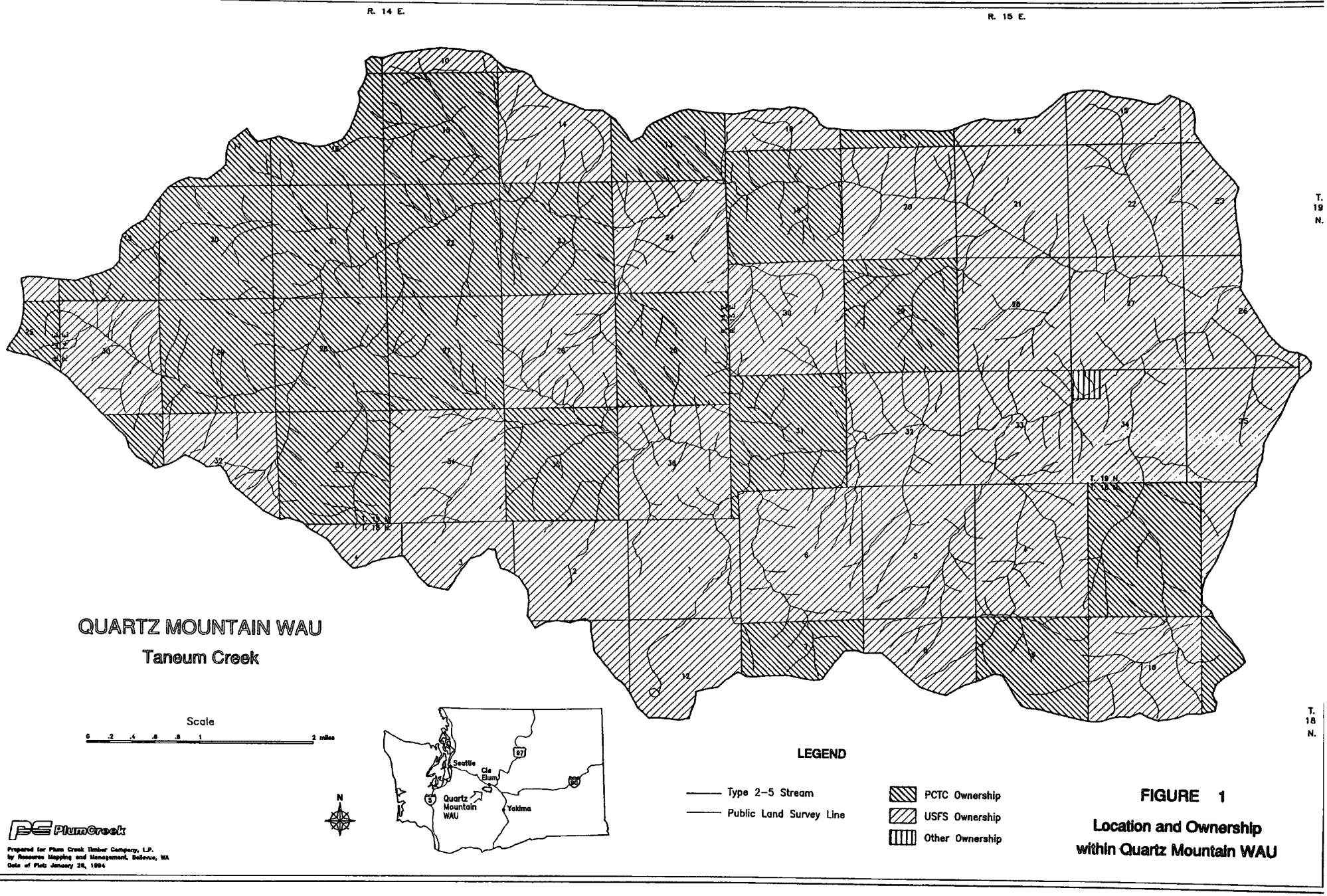

FIGURE 1

Location and Ownership within Quartz Mountain WAU

(6,290 feet) at Quartz Mountain. The watershed is characterized by a relatively wide low-gradient

floodplain along both forks of Taneum Creek with steep sideslopes ranging from 10 to greater than 50 degrees. The geology of the basin is characterized by a complex mix of volcanic, metamorphic and sedimentary rocks with a well-mixed soil mantle of volcanic ash.

Local climate within the Quartz Mountain WAU generally consists of warm, dry summers and cold, snowy winters. Based on records at Cle Elum, temperatures average -3 °C (26°F) in January and 19°C (66°F) in July. The basin receives most of its precipitation, principally as snow, during October through March. Annual precipitation increases with elevation from approximately 1,378 mm (35 inches) at lower elevations to 2,756 mm (70 inches) or more in the upper elevations.

Fish are present in most of North and South Fork Taneum Creek, as well as in the lower portions of the major tributary streams. Resident fish species include rainbow trout *(Oncorhynchus roykiss),* westslope cutthroat *(O. clarki),* brook trout *(Salvelinus fontmalis),* and sculpin *(Cottus* spp.). Historically, Taneum Creek supported anadromous runs of spring chinook *(O. tshawytscha),* coho *(0. kisutch),* and steelhead *(O. mykiss),* but an irrigation diversion ditch approximately 17.7 km (11 miles) downstream of the WAU boundary has served as a migration barrier since around 1910. Bull trout *(Salvelinus confluentus)* were not found in 1993 presence/absence surveys, but likely had historical access to the watershed from the Yakima River.

The watershed area is dominated by ponderosa pine forests at lower elevations, mixed-conifer forests at middle elevations, and by true fir forests at higher elevations. Before the development of modern methods of fire suppression, wildfire played a major role in shaping the forests of this region (Agee, 1994).

The Quartz Mountain WAU area was probably first used by settlers for grazing cattle and sheep between the 1880s and 1930s. Selective removal of large trees for railroad ties occurred in the lower watershed in the 1920s, with roughly 800 acres harvested in four locations. Road construction began in the mid-1950s, followed by extensive partial cutting in the east end of the Lower North Fork sub-basin and upper Frost Creek. The upper half of the WAU was not entered for harvest until 1984. Plum Creek harvests approximately 150-300 acres per year. The U.S. Forest Service is ending timber harvest on their ownership with current plans to place the area in Late-Successional Reserve.

Results of Resource Assessment

Stream channels are shaped by a number of important variables that interact to create characteristics unique to each stream. Some variables such as the gradient, valley confinement and drainage area of a stream are relatively unchanged by human activities. Other variables, however, such as the amount of coarse and fine sediment, the amount of large wood in the stream channel, and the volume and timing of floods can be influenced by timber management activities. These variables influence the channel morphology and dictate the quality of habitat available for fish. Studying the channel morphology, water quality, and riparian area, thus provides a surrogate assessment of the health of the stream system for fisheries.

The most pervasive problem identified in the Quartz Mountain Watershed Administrative Unit (WAU) is the excessive amount of fine sediment in both forks of Taneum Creek. Based on McNeil sampling, fine sediment less than 1 mm in diameter is abundant in the active channel, ranging from 17 to 20 percent of the total volume of sediment in 1993. Fine sediment and small gravel (up to about 16 mm), fills a significant portion of pool volume. Fine sediment accumulation in riffles and pools fills interstitial spaces in gravels, reducing the suitability of spawning and rearing habitat for resident fish. Winter rearing habitat may be the limiting factor for resident fish survival in Taneum Creek.

In general, the Quartz Mountain WAU stream channels may be classified as either steep tributaries or low-gradient mainstem reaches. The low-gradient mainstem reaches have historically stored and transported large quantities of fine sediment. Bankfull flows are sufficiently powerful to mobilize and transport fine sediment entrained in the channel bed; however, because of abundant large wood and other channel roughness elements, extensive secondary channels, and frequent opportunities for overbank flow, much fine sediment that is transportable is deposited in temporary storage sites.

The storage function in the low-gradient mainstem segments is strongly influenced by large woody debris. Both local storage associated with debris jams

and individual pieces of wood, and formation of secondary channels caused by debris jams are critical sediment storage functions of large wood. Moreover, abundant wood tends to reduce stream energy available for sediment transport and encourages overbank flow locally. These factors all contribute to the potential for fine sediment storage in low-gradient mainstem channels.

A major sediment routing function of steep tributary streams is the transport of fine sediment. Nevertheless, significant storage of coarse and fine sediment occurs in tributary channels. Large woody debris is the primary sediment storage element in tributary channels. The analysis recognized the importance of long-term recruitment of woody debris in Type 4 streams to provide storage for fine sediment.

Residence times for fine sediment in the mainstem channel are likely to be at least 8 years, and could be as much as much as 50 years or more. With respect to fine sediment, this residence time estimate suggests that a reduction of fine sediment inputs would not be reflected in stream channels for a period of several years to decades.

There are a number of sources for fine sediment input to the stream system. Soil creep, stream bank erosion, landslides, hillslope surface erosion, and road erosion all contribute fine sediment in varying proportions to streams. Soil creep is the rate of downslope movement of the soil column. The transfer process of sediment from hillslopes to streams occurs primarily through bank erosion and landsliding.

To identify the relative contributions of each input variable and to better assess the role of timber management activities in sediment production, a partial sediment budget was constructed. A partial sediment budget identifies sediment sources and quantifies the rate of sediment production, but does not completely address transport of sediment through the stream network. Sediment source estimates are within the order-of-magnitude accuracy assigned to this type of analysis.

Measurements of bank erosion and landslides were collected in sampled segments and extrapolated for the stream network. A total of 105 landslides were inventoried from field work and aerial photographs. Eighty-three percent of historic landslides have occurred under natural conditions. The remaining 17 percent are associated with timber management activities, specifically small, shallow stream-side failures in recent harvest units, failures in road cuts, and one landing sidecast failure.

Hillslope surface erosion in the watershed is limited due to the high natural permeability of native soils and duff cover which limits overland flow. In the vast majority of harvested areas, recent standard practices have prevented measurable sediment delivery to streams from surface erosion. It appears that the highest erosion potential is associated with soil-exposing activities (ground-based harvest or slash burning) which occur adjacent to streams, rather than on any certain soil types or slope gradients. Surface erosion of hillslopes was of insufficient quantity to include in the sediment budget.

Sediment delivery to streams from road erosion was particularly high in relation to other sources for this watershed. Fine sediment input to streams was high primarily due to roads with unvegetated cutslopes and the high proportion of native surface roads with direct entry to streams. In the lower portion of both forks of Taneum Creek, another problem was unsurfaced roads constructed on highly erodible phyllite-derived soil that ruts readily. Abandoned roads and motorbike trails contributed relatively small quantities of sediment compared to active secondary and low-use spur roads. The erosion from existing roads were rated for each sub-watershed. Most sub-watersheds (i.e., 7 of 9) received a high hazard rating because the estimate of fine sediment input from roads exceeded the total fine sediment input from all natural sources.

Based on a 15-year period, fine sediment from roads in North Fork Taneum Creek is approximately 1.5 times greater than from other sources including landslides, surface erosion of exposed slopes, and bank cutting from both hillslopes and remobilized stream alluvium. The results of the sediment budget, however, indicate that the potential errors in measurements and estimates are of the same magnitude as the comparative increases in road sediment over background. Although the road erosion rates may be overestimated, it is reasonable to assume that delivery of road sediment to streams is of the same magnitude as natural sources.

The input of coarse sediment from timber management activity was not perceived as a problem in the basin at this time. Coarse sediment is abundant in the mainstem channels and relatively mobile. The coarse sediment is stored on lateral bars (point bars) and in medial bars. Sources of coarse sediment are primarily the terraces adjacent to mainstem channels and, in the

lower portion of the watershed, streamside-landsliding. Channel migration and erosion of secondary channels are thought to be important fluvial processes generating coarse sediment inputs. It is likely that only the most extreme events (e.g., debris flow or massive, multiple streamside landslides) would significantly affect channel behavior as influenced by coarse sediment. Coarse sediment residence times are on the order of centuries in mainstem channels and terraces.

The primary process through which land management could accelerate coarse sediment input is landsliding. Slope stability is dependent on both slope and rock competence affected by fracture, bedding attitude and composition. Slopes in the upper watershed underlain primarily with competent metamorphic rock exhibit areas of small-scale rockfalls with talus aprons, snow avalanches, and debris flow morphology with attendant shallow soil failures. In contrast, the layered sedimentary and volcanic rocks are less competent and have planes of weakness in the form of shale, siltstone, or coal or tuffaceous interbeds. Failures occur along the weak interbeds.

Ancient, large-scale mass wasting features have occurred in the basin in response to past climatic regime and seismic activity. Indicators of recent movement of these features, however, are lacking, and we can assume stable conditions with the present climate and angle of repose. Local failures in these soils resulting from lateral support removal in road cuts indicate a sensitivity to disturbances on steeper slopes.

Eight mass wasting map units were created to describe the variety of slope processes, deliverability to streams, relative hazard ratings, and timber management implications related to slope stability in the basin. The only mass wasting unit with a high hazard rating represented approximately 40 percent of the mass wasting in the watershed. The mass wasting occurred on streamside slopes adjacent to middle and lower reaches of both forks of Taneum Creek, unrelated to management activities.

An important factor controlling dynamics of coarse sediment is the amount of large wood in the stream. It is important in retention of coarse sediment (gravel) in steep tributaries. In the low-gradient mainstem channels, large wood determines the location of many gravel bars, and also tends to reduce gravel transport distance. Bankfull flows appear to be sufficient to entrain gravel where the flow is unimpeded by obstructions. Where large wood is abundant, eddies develop and gravel bars tend to form in sheltered portions of the bed.

The majority of riparian areas in the WAU are made up of dense mature conifers that provide natural amounts of shade and large woody debris input to stream channels. The major exceptions include approximately 350 m (1,100 feet) of North Fork Taneum Creek above its confluence with Lookout Creek, and substantial portions of Lookout Creek, Peaches Creek, and upper Butte Creek, where the riparian canopy has been removed by timber harvest in the last ten years. Short and long-term large woody debris recruitment potential is greatly reduced. These reaches are also below minimum shade levels and, according to documentation, have exceeded temperature standards. Historic selective timber harvest and encroachment of roads have also reduced the amount of shade and potential for large woody debris recruitment in the lower portions of both forks of Taneum Creek. Some areas in the middle reaches of both forks have less shade and woody debris recruitment potential than expected due to natural factors such as stream meandering on the North Fork of Taneum Creek and a wet meadow on South Fork Taneum Creek.

One final variable that can be influenced by timber management is the duration and volume of peak streamflows. Removing vegetation can increase the amount of water available for runoff and increase peak flows that shape the channel morphology. The relative importance of rain-on-snow versus spring snowmelt generated peak flows in this watershed was difficult to assess because of the limited data on streamflow. Peak streamflows generally occur in the spring with the melting of the snowpack. Large rain-on-snow floods occur infrequently, probably on the order of once every 10 to 15 years. The Butte Creek sub-watershed was the only area where observed changes in the stream channel could potentially be linked to increased peak flows. Since the majority of runoff in east-side Cascade streams occurs during the spring, studying changes in seasonal runoff(i.e., the timing and volume of snow-melt generated flows) due to timber harvesting is probably more relevant than assessing potential changes in rain-on-snow runoff. The data and/or methods for such an analysis, however, are unavailable at present.

Prescriptions

A number of prescriptions were developed to improve the condition of the watershed and to avoid potential problems in the future. The prescriptions not only address forest practices, but identify monitoring and restoration opportunities as well.

A 5-year road improvement and maintenance plan is being developed to reduce the amount of fine sediment entering streams to 50 percent of natural background inputs. The improvement work includes the placement of additional culverts, revegetation of cutslopes, and abandonment of roads. New roads can be built provided that sediment production from all roads in the watershed is reduced to specified annual target levels.

Riparian areas without enough trees will have 80-foot no-entry buffers. Certain segments of North and South Fork Taneum Creek will have wider buffers likely ranging from 80 to 200 feet to accommodate natural stream meander. In most cases, these wider buffers will be no-entry zones, but limited partial-cut harvest will be allowed in some areas.

Timber harvest within the Butte Creek watershed will be deferred for three years. For the following two years, further harvest will be limited to partial cuts less than 120 acres or clearcut harvest of less than 50 acres. Monitoring of stream bank erosion and recovery of the riparian area will help determine the amount of allowable harvest after the initial five-year period.

Since mass wasting tends to be a very localized problem, timber harvest or road construction in delineated areas will require additional review by a qualified geologist.

Monitoring

Monitoring in the Taneum Creek watershed currently consists of trend and validation monitoring with other stakeholders through the Yakima Resource Management Cooperative. Annual McNeil stream sediment sampling is used to assess the amount of fine sediment in spawning gravels. High levels of fine sediment in spawning gravels reduces the viability of eggs in fish redds. A road sediment monitoring study has been initiated to determine the amount of sediment produced from roads under east-side Cascade conditions. Sediment production from roads with different soil types, road surfacing, and traffic levels are assessed at eight different sites within the watershed area. A streamflow monitoring study was also initiated with two stream gages installed in the South Fork Taneum Creek to assess potential changes in streamflow as timber harvest occurs within the watershed.

Channel conditions are being assessed over time in Butte Creek using photography, permanent channel cross-section measurements, and annual stream surveys. Also, the amount of in-channel large woody debris in North Fork Taneum Creek is being monitored to identify potential watershed restoration activities.

Watershed Analysis and Restoration

Restoration activities in forested watersheds such as road abandonment, riparian plantings, and in-stream structure placement have generally occurred on a site-specific basis without explicitly considering the watershed context. A given site is typically considered deficient in certain elements and the objective of the restoration project is to provide or increase the productivity of those elements. Unfortunately, few of these projects end up as effective long-term solutions because they fail to consider natural landscape disturbance processes. Many of the projects either fail to survive during floods or are not implemented in locations that provide maximum benefits.

Watershed analysis provides a unique opportunity to identify "natural" disturbance regimes for given elements in a landscape and provide guidance on prioritizing areas of the landscape where restoration will provide the greatest benefit to resources of concern.

Washington state watershed analysis is the only process at present that requires a rigorous methodology for identifying local physical process that affect channel morphology. Local information on channel-forming processes is critical for prioritizing restoration projects as well as identifying the potential for long-term success of the project. Data collected at broader scales as is done in the federal watershed analysis process may be useful for identifying general areas where restoration is desirable, but typically do not provide the necessary information to make decisions on the long-term success of restoration projects. For example, if pool frequency in a stream reach is considered deficient for good quality fish habitat, data is needed on the geomorphic context of the area (i.e., the natural geologic setting), current and potential future sediment supply to the area, flooding

Resource of Concern	Spring Chinook spawning habitat in Segments 12 and 13 of the Yakima River is degraded by fine sediment, lack of holding pools, and potential scour during high flows.
Threats to Resource	1. Landslides from roads built prior to 1980 directly into the Yakima River and from debris flows in Cabin Creek. 2. Surface erosion from extensive road network in Cabin Creek. 3. Reduction in large woody debris from power lines in riparian area. 4. Large proportion of watershed is hydrologically immature.
Confidence in Linkage	1. High 2. High 3. Moderate 4. Low
Time Frame for Recovery	1. Years for sediment to route through system. 2. Indefinite, continual input of fines as roads are used. 3. Indefinite for large woody debris input unless power lines relocated. 4. Years for hydrologic maturity
Action Plan	1. Road 2520 and 2550 will be abandoned with culverts removed and sidecast material pulled back. 2. Road 2500 will have any sidecast on slopes >60 percent pulled back. 3. Drainage problems and sediment delivery to streams will be evaluated and addressed for the entire road network. 4. Alternative silvicultural methods for maintaining vegetation under power lines will be investigated with the power company. 5. Placement of large woody debris to increase holding pool capacity will be explored
Objective with Time Frame	The action plan will reduce sediment delivery to Segment 12 and 13, but measurable improvement in percentages of fines in gravels is not expected for 10 years or more. Short- and long-term (-50 years) large woody debris input will increase with silvicultural treatments and placement of wood in-channel.

Figure 2. Example Restoration Action Plan

regime, and riparian vegetation conditions before any decisions are made on restoration projects. Such information is not collected or put in the context of watershed processes for either the federal or Idaho watershed analysis processes.

Watershed analysis can be a useful tool for guiding restoration activities, but many obvious site-specific problems exist that can be addressed without expensive, time-consuming landscape analysis. Data generated from watershed analysis is most useful for developing effective, long-term watershed-scale restoration plans and for guiding site-specific in-stream projects. Depending on the long-term objectives of the landowner or agency that is initiating watershed analysis, data collection can and should be modified to be the most practical and useful for guiding decision-making.

Five important elements are necessary to prioritize and help implement restoration activities:

1. Identification of resources of greatest concern and their conditions.
2. Identification of threats to resources and to the potential projects.
3. Evaluation of the linkages between the threats and the resources
4. A clear statement of the objectives of the project.
5. Identification of the time frame for expected recovery of the resources.

These elements provide a systematic approach to restoration that is critical for evaluating the costs and benefits of particular projects. Figure 2 contains an example restoration action plan.

Conclusions

Providing for long-term restoration of aquatic resources requires an analysis of landscape elements at a local scale. Watershed analysis that is scientifically rigorous and has direct links to management decision making can provide useful data for planning and implementing long-term restoration projects.

Due to the long-term nature of most recovery efforts, a systematic approach to restoration is also critical. Watershed analysis that provides data on watershed process at a local scale can ensure that restoration efforts will be successful in the long-term. Active restoration management will be necessary to speed the recovery of many aquatic systems, but will need to rely on good local data collected in the context of landscape processes. Finally, monitoring of restoration projects and the recovery of resources will provide a critical feedback mechanism for future restoration efforts.

Literature Cited

Agee, J. 1994. Fire and weather disturbances in terrestrial ecosystems of the eastern Cascades. General Technical Report PNW-GTR-320. U.S.D.A. Forest Service, Pacific Northwest Research Station, Portland, OR. 52 pp.

Bilby, R. E. 1984. Removal of woody debris may affect stream channel stability. Journal of Forestry 10: 609-613.

Bisson, P. A., R. E. Bilby, M. D. Bryant, C. A. Dolloff, G. B. Grette, R. A. House, M. L. Murphy, K. V. Koski, and J. R. Sedell. 1987. Large woody debris in forested streams in the Pacific Northwest: past, present, and future. pp. 143-180 In Proceedings, Streamside Management: Forestry-Fisheries Interactions. Edited by E.O. Salo and T.W. Cundy. University of Washington, Seattle, WA.

Caldwell, J. E., K. Doughty, and K. Sullivan. 1991. Evaluation of downstream temperature effects of Type 4/5 waters. Timber/Fish/Wildlife Report TFW-WQ5-91-004. Washington Department of Natural Resources. Olympia, WA. 71 pp.

Chapman, D. W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. Trans. Amer. Fish. Soc. 117: 1-21.

Coffin, B. A. and R. D. Harr. 1992. Effects of forest cover on volume of water delivery to soil during rain-on-snow. TFW-SH1-92-001. Washington Department of Natural Resources, Forest Practices Division, Olympia, WA. 118 pp.

Idaho Department of Lands. 1995. Forest Practices Cumulative Watershed Effects Process for Idaho. Boise, ID.

Lisle, T. E. 1989. Sediment transport and resulting deposition in spawning gravels, North Coastal California. Water Resources Research 25:1303-1319.

Lisle, T. E. and S. Hilton. 1992. The volume of fine sediment in pools: an index of sediment supply in gravel-bed streams. Water Resources Bulletin 28:371-383.

MacDonald, L. H., A. W. Smart, R. C. Wissmar. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. EPA 910/9-91-001. U.S. EPA. Seattle, WA. 166 pp.

McDade, M. H., F. J. Swanson, W. A. McKee, J. F. Franklin, and J. VanSickle. 1990. Source distances for coarse woody debris entering small streams in western Oregon and Washington. Canadian Journal of Forest Research 20:326-330.

Megahan, W. F. 1983. Hydrologic effects of clearcutting and wildlife on steep granitic slopes in Idaho. Water Resources Research 19:811-819.

Murphy, M. L. and K. V. Koski. 1989. Input and depletion of woody debris in Alaska streams and implications for stream side management. North American Journal of Fisheries Management 9: 427-436.

Peterson, N. P., A. Hendry, and T. P. Quinn. 1992. Assessment of cumulative effects on salmonid habitat: some suggested parameters and target conditions. Timber/Fish/Wildlife Report TFW-F3-92-001. Washington Department of Natural Resources. Olympia, WA.

Regional Interagency Executive Committee. 1995. Ecosystem Analysis at the Watershed Scale, Federal Guide for Watershed Analysis, Version 2.2. Portland, OR. 26 pp.

Robison, E. G. and R. L. Beschta. 1990. Identifying trees in riparian areas that can provide coarse woody debris to streams. Forest Science 36: 790-801.

Sidle, R. C., A. J. Pearce, and C. L. O'Loughlin. 1985. Hillslope stability and land use. Water Resources Monograph Vol. 11. AGU, Washington D.C. 140 pp.

Sullivan, K., J. Tooley, K. Doughty, J. E. Caldwell, and P. Knudsen. 1990. Evaluation of prediction models and characterization of stream temperature regimes in Washington. Timber/Fish/Wildlife Report TFW-WQ3-90-006. Washington Department of Natural Resources. Olympia, WA. 224 pp.

Swanson, F. J., L. Benda, S. Duncan, G. Grant, W. Megahan, L. Reid, and R. Ziemer. 1987. Mass failures and other processes of sediment production in Pacific Northwest landscapes. pp. 9-38 In Proceedings, Streamside Management: Forestry-Fisheries Interactions. Edited by E.O. Salo and T.W. Cundy. University of Washington, Seattle, WA.

Swanston, D. N. and C. T. Dyrness. 1973. Stability of steep land. Journal of Forestry 71: 264-269.

U.S. Army Corps of Engineers. 1956. Snow hydrology - summary report of the snow investigations. North Pacific Division. Portland, OR. 437 pp.

U.S. Department of Agriculture. 1994. A Federal Guide for Pilot Watershed Analysis, Version 1.2. Portland, OR. 202 pp.

Washington State Forest Practices Board. 1995. Standard Methodology for Conducting Watershed Analysis, Version 3.0. Washington Department of Natural Resources. Olympia, WA.

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Prioritization of Salmonid Habitat Problems and Habitat Restoration Projects in an Urbanizing Watershed

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An interdisciplinary design team identified and prioritized salmonid habitat deficiencies in approximately 14 miles of stream and riparian corridor in the Bear/Evans Creek watershed, King County, Washington. This watershed supports regionally significant runs of sockeye (*Oncorhynchus nerka*), chinook (*O. ishawyisha*), and coho (*O. kisutch*) salmon, and steel head (*O. mykiss*) and cutthroat (*O. clarki*) trout in a rapidly urbanizing area northeast of Seattle (KCSWM 1990). A two-step process was designed to first identify, locate, and rank the habitat problems in the stream system; and second, to apply financial, logistical, permitting, construction, and landowner-imposed constraints to define restoration projects for design and construction. Habitat restoration projects are undertaken only with the full cooperation of the landowner and are entirely at public expense.

The problem ranking process involved walking streams and documenting physical habitat conditions; analysis of aerial photos and map overlays; and collecting land ownership information on all streamside parcels. Spreadsheet software was used to organize the information and perform simple sorts and calculations. Principal categories of habitat problems included: fish passage barriers, streambank degradation, poor instream habitat diversity, and lack of riparian vegetation (Johnson and Caldwell 1994). The team developed prioritization criteria to reflect pertinent habitat issues and the differing characteristics of the four streams of interest in the watershed (Entranco et al. 1994). Prioritization criteria, in the form of questions, were applied to each habitat problem and stream (Table 1). Question scores for each habitat problem were summed yielding a "tributary score" representing the relative severity of a particular habitat problem in each stream. Tributary scores were assigned to individual reaches within a stream when distinctions in habitat problem characteristics were observed among reaches. Prioritization scores for each land parcel were derived by summing the products of the tributary scores and stream reach lengths of habitat problems observed on each parcel (Table 2). The resulting "land parcel habitat scores," reflecting the relative severity of habitat problems on each parcel, were ranked to determine priority parcels for stream restoration in the watershed. Raw scores (incorporating tributary scores but not weighted for stream length) were referenced as indicators of the number of problems on a parcel. Before ranking potential project sites, it was necessary to group parcels into clusters that defined a feasible project extent. Clusters were kept as small as possible and were formed by grouping: 1) contiguous parcels with the same owner and land use; 2) parcels directly across the stream from each other; and 3) parcels where problems could not be addressed without involvement of adjacent parcels. Land parcel habitat scores were summed to yield cluster scores, which were adjusted to ensure that instream diversity problems were not counted twice for parcels across the stream from each other. Cluster scores were ranked to determine priority sites for stream habitat restoration on each of the four streams and in the watershed as a whole.

A secondary set of criteria, which considered financial, logistical, landowner, permitting and other constraints, was then applied to determine restoration projects for design and implementation.

Implementation of habitat restoration projects began in 1995 and will continue through 1997. Addressing barriers to fish passage is the highest priority. Other components of restoration projects include: enhancing instream habitat, biostabilizing eroding streambanks, fencing, and revegetating riparian corridors.

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Table 1. Prioritization Questions and Scoring for Tributary/Habitat Problems

	Question	Score
1.	If nothing is done, will the problem get worse?	Yes = 10 points; No = 0 points
2.	How much of the stream in the subbasin is affected by the problem? (For example, if a potential fish barrier is located downstream, the potential for blocking a majority of upstream areas exists.)	If >67 percent of stream, score high = 10 points; if 33 to 66 percent, score medium = 5 points; if <33 percent of stream score low = 0 points.
3.	How many species of salmon or trout using the steam are affected by the problem?	If the number of species using the affected reach is all, score high = 10 points; if 50 percent of species, score medium = 5 points; if only 1 species, score low = 0 points.
4.	Of the following, how many problems are present: streambank degradation, lack of instream diversity, degraded riparian corridor, fish passage barrier, miscellaneous (e.g., trash or debris, water quality concerns)?	If 5 of 5 problems exist, score 10 points; if 4 of 5, score 8 points; if 3 of 5, score 6 points; if 2 of 5, score 4 points; if 1 of 5, score 2 points.
5.	Has the tributary or stream reach been designated either a regionally or locally significant resource area by King County?	Regionally significant score 10 points; locally significant score 5 points; 0 points for others.

We found that a clear definition of the goals and methodology of the prioritization process is imperative to gather useful, well-organized data and enable cost-effective allocation of limited public funds to address the most pressing habitat problems. Most project sites are privately-owned and landowner participation in restoration efforts is voluntary. Some of the highest-ranking projects have not been undertaken because of unwilling landowners. When implementing restoration projects, we found that close coordination with landowners is essential to ensure that a project meets intended goals. Clearly communicating the intent of the project and providing the landowner with a vision of the final product, through graphic design illustrations, pictures, or visits to similar projects is crucial to ensuring good working relationships with landowners and successful implementation of projects.

Literature Cited

Entranco, King County Surface Water Management Division, J. F. Osborn, Caldwell and Associates, and Lee and Associates. 1994. Bear, Evans, Cottage Lake, and Mackey Creeks: Habitat Problems, Prioritization, and Solution Development. 40 pp. 4 app.

Johnson, A. W. and J. E. Caldwell, 1994. Bear, Evans, Cottage Lake, and Mackey Creeks Habitat Assessment. Prepared for Entranco and King County Surface Water Management Division. 17 pp. + app.

KCSWM. 1990. Bear Creek Basin Plan, King County Surface Water Management Division. Seattle, WA. 117 pp.

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Table 2. Formula for Land Parcel Habitat Scores

Land Parcel Habitat Score =

Tributary Riparian Score x (Riparian Disturbance Zone Weight x Reach Length of Riparian Disturbance Zone)

+

Tributary Bank Erosion Score x Reach Length of Bank Erosion

+

Riparian Shade Score x Total Reach Length

+

Tributary Fish Barrier Score x Reach Length of Fish Barrier

+

Tributary Instream Diversity Score x Reach Length of Poor Diversity

all divided by 100

The formula components are defined as:

Tributary Riparian Score = a number between 0 and 50, which weights riparian problems for each tributary.

Tributary Bank Erosion Score = a number between 0 and 50, which weights bank erosion problems for each tributary.

Tributary Fish Barrier Score = a number between 0 and 50, which weights fish passage problems for each tributary.

Tributary Instream Diversity Score = a number between 0 and 50, which weights instream diversity problems for each tributary.

Riparian Shade Score(1) = either the number 2, which represents <50 percent shade for at least 200′ of stream; or the number 0, which represents >50 percent shade for at least 200′ of stream.

Riparian Disturbance Zone Weight(2) = a percentage assigned to disturbance zones 1 through 5.

Disturbance Zone	Riparian Condition Weight
1 - high disturbance	1.00
2 - high to moderate	0.66
3 - moderate	0.33
4 - moderate to low	0
5 - low disturbance	0

(1) The score was derived from streamwalk data.

(2) The weight is based on aerial photo analysis. In the case where more than one disturbance zone occurs on a land parcel, each zone length is multiplied by the appropriate weight.

Active Management of Riparian Habitats

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Abstract: **Current approaches to protecting biotic diversity are passive—let nature take its course. This approach leaves degraded ecosystems. If forest and stream processes are disrupted, these ecosystems may not recover for decades. Short term project life, as is the case with many restoration projects, limit the ability to implement long term solutions, such as the recovery of native riparian forest structure and in stream populations. Only a long term solution, actively managed forests and streams, can restore natural processes. Forest managers can use silviculture to create benefits for both aquatic and terrestrial habitats. Release of growing space for selected trees allows development of larger trees for LWD recruitment and increases light to the forest floor and stream bottom. Currently, no systematic design to accomplish these goals exists. An active management approach, such as described here, is prescriptive and adaptive while providing returns to both landowners and the riparian ecosystem.**

Introduction

Forests supply logs to streams, critical elements of the typical step pool morphology (Grant et al 1990). Logs, or large woody debris (LWD), set up pools and help retain sediment in low order streams (Heede 1985, Keller and Swanson 1979); LWD provides the matrix for the capture of fine sediment and small debris (Megahan 1982 Bilby 1981). In addition to sediment retention, the logs play an important role in both aquatic (Bisson et al. 1987, Bisson et al. 1982) and terrestrial (Franklin et al. 1996, 1986) habitat structure. In large rivers, riparian forests are vulnerable to the risk of large flood flows; here islands are often only vegetated for decades (Swanson and Lienkaemper 1982). But in the upper reaches where salmon spawn and rear, wood is a primary building block for the complex habitat structure (Sedell et al. 1989, Lienkaemper and Swanson 1987, Harmon et al. 1986, Grette 1985, Bryant 1983, Triska et al. 1982).

A mix of both deciduous and conifer forest is typical on the fresh geomorphic surfaces left from floods (Agee 1988, Oliver and Hinkley 1987). In many places where early seral hardwood species have colonized, such as off-channel pools, habitat is very productive for over-wintering salmon (Heede 1985, Cederholm and Scarlet 1982). Riparian forests are species diverse (Nilsson et al. 1989) and should be protected and cultivated for the functions they provide. Hardwood dominated forests do provide necessary functions and have always been present along streams and rivers in the Pacific Northwest (Naiman et al. 1992).

Rivers change in many ways as they move downstream from the headwaters to the sea (Vannote et al. 1980). Our streams originate in the alpine zone and flow through the upper reaches of the watershed where landform influences control the channel, and forests supply wood and leaf litter. The valley floor begins to widen and the river begins to develop terraces. At higher elevations the valley walls are steep, and cold runoff has great force as it falls through the upper watershed (Naiman et al. 1992, Vannote et al. 1980). Headwaters are biological storehouses for the food webs that develop downstream. Forests adjacent to rivers and streams transpire and process runoff. Intricate webs of tree roots form along the banks, anchoring topsoil.

In-channel LWD attrition (e.g., decay, mobilization, extraction) increases the difficulty of reestablishing conifers in degraded riparian forest stands. Because large jams are not present to protect bars from flood flows, the bars cannot revegetate. Also, hardwoods colonize many areas and compete with the near-term conifer regeneration and subsequent recruitment of conifer logs to the stream channel. The size of functional wood increases with channel gradient and with drainage area (Bilby and Ward 1990). In steeper streams, much of the functional wood was recruited prior to logging (Potts and Anderson 1990, Grette 1985). For these reasons, off-channel habitat in the low gradient mainstem may be slow to recover.

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Structure in Streamside Forests

Elements of the streamside environment include the active channel, the associated flood plain, and the zone of influence (after Cummins 1974). The active channel and floodplain are easily recognized and have clear biological and physical limits to their extent. The zone of influence is broader and encompasses the shading, litter fall, and sources of large woody debris (McDade et al. 1990). Riparian forest structural attributes are listed below.

- Live, vigorous trees and large snags as the source of LWD
- Large down logs for habitat formation and sediment retention
- Suspended and embedded wood in the stream channel
- Vertical distribution of foliage
- Many sizes and species of trees in the adjacent forests.

Longitudinal Changes in Ecological Processes

In the upper watershed the nearshore riparian is dominated by brushy early seral trees and shrubs, because the force of the water reworks the narrow valley floor annually. Trees establish very rarely. The narrow stream channels are choked with woody debris that holds what little sediment is stored. The valley wall confines some stream reaches and limits development of forests near the active channel.

As the river widens, terraces begin to build soils and the forest produces large logs. The adjacent forests generate woody debris that is large enough to resist the stream flow (Bilby and Ward 1990); but in large rivers riparian forests are vulnerable to the risk of floods. Rivers move large amounts of material and rework valley floors. The power of these rivers is capable of drastic modification of valley bottom riparian forests.

Streamside forests protect streambanks and channels because they add roughness to the channel, reduce the flow of flood waters, and stabilize soils (Heede 1982). Riparian soils vary widely in both distribution and productivity. Scour and deposition are responsible for some of the spatial variability in site quality. Site capability for tree growth is degraded when fine, nutrient-rich sediments are leached out of coarse-grained gravel bars. Elevated drainage capacity of gravely soils further reduces their nutrient holding capacity. Sediment deposition in the form of silts, and, and gravel from upslope, create sites that are well drained and productive. However, some anaerobic soils are present along streamsides and can result in reduced productivity.

Functions of lateral connections to streams include the input of litter and debris (to drive food chains and provide channel structure) and the hyporheic zone, which functions as both habitat to stream invertebrates and as a fine filtering system for ground water recharge (Stanford and Ward 1988). Forest structure influences the quality of these lateral functions. Forest stands have some effect in altering the channel morphology but they do influence the hyporheic zone, a deep, wide, underground flow. The space beneath the forest processes nutrients through the filtering (e.g., through roots, gravels and sediments) and biological activity of the flora and fauna of the interstitial spaces (Stanford and Ward 1988).

Passive and Active Management

Where stream systems are degraded, the inherent resilience in forests will eventually succeed in recolonizing the area with large dimension conifer forests. Acting passively may not recover the forest functions in time to be of use to declining fish populations, however (CSE 1995). An active approach to riparian management asks, "Can we grow forests that sufficiently supply the proper size and amount of wood?" The alternative is to let the stands proceed through successional pathways toward functional stand structure, perhaps at a much slower rate. A managed forest could feasibly develop beneficial forest structure in a much shorter period than leaving it to the chance circumstances of natural succession. Active management may accelerate development of full canopy cover by planting a mix of desirable species (e.g., red cedar, spruce, Douglas-fir, black cottonwood).

Relationships between forest structure and in-stream channel morphology need to be set up as working hypotheses to be tested. If we are going to manage this apparently unmanageable ecosystem then we need to establish simple empirical models for the different forest and stream types (e.g., McDade et al. 1990, Bilby and Ward 1990, Mitsch and Jorgenson 1989).

An important element of active management is economics. From the forest managers' standpoint, what does riparian recovery cost? We can do better than break-even while creating a residual forest capable of sustaining its own LWD supply, perhaps even beginning to restore some of the more subtle functions of the riparian forest? The Pacific Northwest is blessed with highly productive sites and high value timber species. But even so our regional forest economics are subject to many of the same worldwide influences, interest rates, amount of early investment, growth and recovery rates, and risk of disturbance. The economic feasibility of various silvicultural systems can be evaluated relative to these parameters (Berg 1995).

When viewed at the stand level, it is almost overwhelming to think that restoration might be recommended on thousands of miles of steams in the Pacific Northwest: Hundreds of thousands of acres of degraded streamside forests. Riparian management can improve the function of these streamside forests and restore many of the habitats associated with streams. Large diameter trees, snags, and down logs are the structure most limited in supply in terms of the LWD component of stand structure. This is something we know intuitively, yet it still requires an extensive commitment of research effort.

Basin Scale Silviculture

Basin scale silviculture is the next frontier as we wrestle with definitions, objectives, and the increasingly demanding goals of society (Oliver et al. 1992). Watershed planning and analysis identify the condition of streams and the associated/dependent resources such as salmon populations. Short term projects, as is often the case with restoration projects, limit the ability to implement long term solutions over large areas, such as the recovery of native riparian forest structure and survival of instream populations. Basin level designs should include project implementation and monitor follow through, which will require longer time frames.

Streams and the associated riparian zone are structural elements in the larger landscape that is controlled more by the shape of the land (Trotter 1990, Swanson et al. 1982). Patch size and stand distribution over the landscape interact at a watershed level, and the arrangement of various stand structures may affect landscape functions, such as groundwater (Stanford and Ward 1988) and surface water flow, stream temperatures (Sullivan et al. 1987), or wildlife corridors (Raedeke 1988). Vegetation indicates the condition of a stream or stand (Schoonmaker and Mckee 1988); the species and age indicate time or severity of the last disturbance (Oliver and Larson 1990).

Collins et al. (1994) describe how riparian forests change through time in a watershed, focusing on current stand conditions and their relationships to fish habitat. Washington's standard methodology for conducting watershed analysis (WFPB 1993) identifies three broad classifications of stand types that can be related to the stream shading and debris recruitment functions of the riparian zone. Collins et al. (1994) added a third role of the riparian forest, resisting bank erosion, and categorized forest stand types to a higher resolution than State methodology (WFPB 1993) so restoration recommendations could be more directly interpreted from the data (stand types and ratings are listed in Table 1).

Riparian Management in the Pacific Northwest

Loss of riparian forests has exacerbated channel widening during floods, increased stream temperatures in some reaches, and diminished opportunities for woody debris recruitment. Priority areas are refugia and low gradient mainstems, which potentially are the most productive habitats for steelhead and coho salmon. Not all streams need rehabilitation. Functional riparian zones should be recognized and maintained and yet, in light of all the empirical evidence presented above, a relatively small amount of our collective wisdom is used in the design of recovery systems.

The extensive lower elevation, floodplain forests have been gone from most Pacific Northwest rivers for decades (Sedell and Froggatt 1984). Stream protection efforts have largely focused on salmonid habitat; enhancing parameters such as water quality, shade, LWD, and litter. Many basic biological requirements of other instream species are unknown. Little consideration is given to habitat composition, location, larger scale connectivity, persistence, and variability.

Alder dominated riparian zones of the Pacific Northwest are prime examples of second-growth forests lacking in structural and biological diversity. Past practices have left many streams in dire need of structural enhancement and riparian forest

Table 1. Stand type classes identified in the assessment of riparian forest stands in the Deer Creek basin. (after Collins et al. 1994).

Stand Type	Description	Shade potential	LWD recruitment potential	Bank reinforcement potential
Open	Recently disturbed (landslide, harvest, gravel bar); unvegetated or herbs and grasses	LOW	LOW	LOW
Brush	Vegetated - small diameter (<6") trees and shrubs, herbs	LOW	LOW	LOW
Sapling	6"-12" average DBH (tree diam. @ 4.5 ft. above ground)			
Conifer - C	> 70% conifer	MOD	MOD	MOD
Mixed - M	mixed conifer/deciduous	MOD	MOD	MOD
Hardwood -	> 70% deciduous	MOD	LOW	MOD
Pole	12"-18" average DBH			
Conifer - C	> 70% conifer	HIGH	MOD[a]	MOD[a]
Mixed - M	mixed conifer/deciduous	HIGH	MOD[a]	MOD[a]
Hardwood -	> 70% deciduous	HIGH	MOD	MOD[a]
Mature	>18" average DBH	HIGH	HIGH	HIGH[b]
Conifer - C				
Mixed - M				
Hardwood -				

a. Moderate in western hemlock zone (below 2000 feet elevation), high in silver fir zone (above 2000 feet elevation) where average DBH is 12"-18" in mature and old-growth stands.

b. This function is dependent on the size of the stream. In smaller reaches (especially moderate gradient tributary reaches) mature conifer trees help resist bank erosion by floods. In larger reaches and the low gradient mainstems, large floods may still significantly erode banks even with mature conifer stands established. However, in the low gradient mainstems, banks without large conifer have been more severely eroded than banks with large conifer trees.

restoration. Before the establishment of streamside buffers, conifers were systematically removed from riparian areas and streams were cleaned of woody debris. Frequently, stream channels were used to skid logs, if not splash damned and used to drive logs down. The resultant plant community is generally dominated by red alder and shrubs, such as salmonberry, which in some areas may persist as a plant community until a disturbance or intervention.

In hardwood dominated and debris poor forests, stream channels are often also limited in structural diversity; lacking a mixture of wood-formed pools, known to be important fish habitat. Engineered in-stream structures (e.g., gabions, logs placed with heavy equipment) improve fish habitat only for a short time. A more sustainable, long-lasting, and less expensive method is to establish or release riparian zone conifers. This approach allows the forest to provide recruitment of decay resistant woody debris.

Recovery of Riparian Forests

The economic motivation for restoration is elegantly simple; Northwest salmon streams once supported a thriving industry. Now stocks are declining and the source of fishing revenues is questionable (CSE 1995). Riparian silviculture restores native watershed functions responsible for forming and maintaining stream habitats for a variety of aquatic species, but in a sequence that favors long-term recovery of the seasonal salmon runs.

A second focus of restoration improves habitat related to riparian forests and recruitment of wood from those forests. By planting fast-growing trees with dense root mats (e.g., cottonwood), silviculture speeds recovery of riparian shading and bank stabilization, especially on low terraces. Recovery of decay resistant wood recruitment by various riparian silvicultural treatments are longer term.

The goal is to recover the mature conifer forests that generally dominated riparian forests in the past and provided shade, woody debris to streams, and root strength to erodable banks. Silvicultural treatment can restore the functions of a mature riparian forest using an active approach. Wood delivered to channels augments the effects of riparian vegetation on stabilizing stream banks and bars. Placing wood in channels can improve habitat in key areas (e.g., pools for rearing habitats).

Riparian silviculture requires an inventory of stand and basin riparian conditions. The key to manipulating structural diversity in riparian areas is the establishment of conifers over a long period of time. If understory conifers are present, then they may simply be released. Under planting of conifers may enhance the opportunity to establish multiple cohorts of trees. Thinning to wide spacings also allows regeneration of many species beneath the overstory, even moderately shade intolerant Douglas-fir. Stand dynamics will be particularly important to monitor in riparian areas where aggressive, early seral species like red alder and salmonberry dominate and are likely to out compete riparian conifer plantations.

Target species composition for planting to meet long-term LWD targets is mixed Douglas-fir and western red cedar, although higher elevation stream reaches are usually dominated by the silver fir zone (Franklin and Dyrness 1973) with hemlock and yellow cedar.

Where stream temperatures are elevated and root strength loss has been significant, primarily in the low-gradient mainstem reaches, short-term objectives include establishment of fast-growing cottonwood stands to rapidly recover a dense root mass and to shade the stream channel. As these stands mature and riparian functions are established, gradual conversion through active management to conifer stands can proceed.

Sapling stands of hardwood, or mixed hardwood stands with a conifer understory (e.g., hemlock, cedar), may offer opportunities to recover some of the costs associated with planting conifer species through commercial thinning. Sapling conifer stands can be thinned to increase growth rates and recover large woody debris recruitment.

Restoration of riparian forests should encourage planting of species that both improve bank stability and provide shade. Given a historical channel width of 40 metres and a tree height of 50 metres, the target canopy cover should be 30 percent (Collins et al. 1994). A short term target may be recover 30 to 40 percent canopy cover, accomplished through plantations of fast growing hardwoods (e.g., cottonwood, alder) and decay resistant conifers (e.g., red cedar, Douglas-fir).

There are stream reaches where the unvegetated floodplain extends 200-300 feet from the existing streambank. Planting sites are not always available at the streams edge because of the coarse boulder/

cobble substrate. Here a staged revegetation plan can be developed that attempts to advance the edge of the existing riparian forest incrementally by decade (e.g., plant the first 25-50 feet from the edge of existing forest in the first decade with cottonwood or red cedar; then advance the next increment each decade until the floodplain vegetation is recovered). As the forest recovers, an active silvicultural program (e.g., thinning and planting) may accelerate development of large conifer trees. These reaches are a high priority for recovery.

High priority stands include those in open and brush areas, as well as newly-formed and largely unvegetated terraces in low-gradient mainstem channels. Restoration options for open and brushy areas typically include clearing growing space and establishing mixed Douglas fir and western red cedar, at an estimated cost of $120 to $300 per acre (Table 2). On unvegetated terraces, initiation of cottonwood is the preferred option. Site selection should consider suitability of the selected species to the site, susceptibility to flooding, and proximity to side channel or valley tributary habitats.

Sapling stands of hardwood, or mixed hardwood and understory conifer, are the second priority. These stands may offer opportunities to recover some of the costs of restoration through commercial thinning before planting conifer species. Costs range from $275 to $300 per acre for thinning with no extraction and planting, and from $125 to $150 per acre for commercial thinning and for active management planting. Sapling conifer stands are a lower priority, but they can be thinned to increase growth rates for large woody debris recruitment. Thinning alone costs $150 per acre, but commercial thinning can result in a profit of about $600 per acre (Table 2).

Pole hardwood or mixed stands can be thinned to encourage conifer growth with understory reestablishment through planting of conifer. Costs without extraction range from $125 to $150 per acre, costs with commercial thinning range from $325 to $350 per acre. Pole conifer stands in the western hemlock zone (below 2000 feet elevation) can be thinned from the lower two thirds of the size class (<12" DBH) but this option is not encouraged because the benefits to riparian functions are likely to be small. Piling and pole size conifer stands in the silver fir zone require no action because they are at or near maximum size and are dominantly conifer.

Other Stream and Riparian Restoration Measures

Silvicultural treatment is one method for accelerating development of a mature riparian forest with many of the functions previously mentioned (Rainville et al. 1985). There are also some opportunities for manipulating wood in channels to augment the effects of riparian vegetation on stabilizing stream banks and bars. These opportunities are focused in the low gradient mainstems where large jams can protect newly planted vegetation from floods. Riparian planting should take advantage of existing jams at the upstream ends of bars and newly-formed terraces as a first priority, but jams may be constructed in some areas by cabling together existing woody debris that is scattered across bar tops. Higher terraces with large jams at the upstream end, to protect them from flood flows, are likely to persist in the long-term and should be addressed first. Additionally, the value of off-channel habitats to coho and steelhead indicates that sites providing riparian functions to both the mainstem and off-channels should receive priority.

Habitat restoration with woody debris is most effective in small low-gradient channels (Potts and Anderson 1990). Wood placement is typically used to create pools for rearing habitat, or to protect stream banks from eroding. Placement of woody debris to create pools is typically expensive, and usually improves habitat conditions only within a few meters of the structure. Placement of logs at critical points in-channel can accelerate development of pools and sediment retention. One must acknowledge the dynamic nature of log/pool location and allow for movement of logs through the system (Reeves et al 1995). These approaches are temporary solutions for maintenance of channel structure. Log placement can be accomplished a number of ways, including the use of skyline yarding systems set upon adjacent cutblocks. This technique, called "back-yarding" (Loren Kellogg, Oregon State University; personal communication) has a key advantage of being able to access the channel without heavy equipment and associated site degradation. Yarding corridors can be established at designated points based on critical in-channel conditions in conjunction with standard forest operations.

Beavers play a significant role in the riparian ecosystem, by creating and maintaining off-channel aquatic habitat. The primary roles of beaver in salmonid ecology are (1) physically creating overwintering habitat for coho salmon by dam construction, and (2) helping maintain conifer forests along streams

Table 2. Silvicultural options for the riparian conditions identified in the riparian assessment, with the target of large conifer in the riparian zone for LWD recruitment and shade. All DBH measures are for average of stand, not largest trees. Costs of stock are based on (a) $200/M for cottonwood seedlings and (b) $250/M for cedar and Douglas fir seedlings. Time to objective is based on hemlock, Douglas fir or cedar in the Western hemlock zone (< 2000 feet elevation). Where it is relevant, management options and time to objective are noted for Silver Fir Zone stands.

Stand type	Objective and silvicultural activities	24" DBH	Time to objective 36 " DBH	Cost *Labor*	Cost *Planting Stock*	Log Value	Net *PNV*
Open : low bar in channel	-Active channel; no activity.						
Open: high bar	-Short term objective: stabilize low terrace to provide shade and resist bank erosion. -OPTION 1: plant in spring with alder or cottonwood depending on site conditions; 100-200 trees/ac. High risk of failure due to floods.	30 years to establish cottonwood (24" DBH).	50 years to establish alder (16" DBH).	$80/ac	$40/ac[a]		OPTION 1 -$120/ac
Open: floodplain or upland	-objective: large conifer (LWD recruitment and shade). - OPTION 1: release planting space; plant cottonwood, cedar, D. fir at 400 trees/ac; thin at 20 yr. - OPTION 2: release planting space; plant cedar and Douglas fir at 200 trees /ac; no thin.	cedar - 90 yr. D. fir - 70 yr.	cedar - 150 yr. D. fir - 90 yr.	*-release-* $40/ac *-plant-* $100/ac	$50/ac[b] for 200 tree/ac		OPTION 1 -$190/ac
Brush : conifer (dense, small dimension, low growth potential)	- objective: large conifer (LWD recruitment and shade). - OPTION 1: juvenile spacing (PCT), target of 200 free to grow/ac.	70 yr.	100 yr.	$150/ac			OPTION 1 -$150/ac
Brush: deciduous	- objective: large conifer (LWD recruitment and shade). - OPTION 1: thin deciduous (release growing space) and plant conifer at 200 trees / ac (100 TPA each of Douglas fir and cedar).	100 yr.	130 yr.	*-release-* $150/ac ; *-plant-* $100/ac	-plant- $50/ac		OPTION 1 -$300/ac
Brush: mixed	- objective: large conifer (LWD recruitment and shade). - OPTION 1: release existing conifer by thinning deciduous (target of 200 conifer/ac); plant to target level if necessary.	hemlock, fir - 70 yr. cedar - 90 yr.	hemlock, fir - 100 yr. cedar - 150 years	*-release-* $150/ac; *-plant-* $100/ac	*-plant-* $25/ac		OPTION 1 -$275/ac

Sapling: conifer	- objective: large conifer (LWD recruitment and shade). - OPTION 1: thin to 150-200 tree /ac (no extraction). - OPTION 2: thin to 150-200 tree /ac (commercial thin).	50 yr.	80 years	*-release -* $150/ac; *-comm. thin-* $400/ac (@$200/Mbf logging and haul; 2 Mbf/ ac)		*-comm. thin* $1000/ac (conifer @ $500/MBF; 2 MBF/ac)	OPTION 1 -$150/ac OPTION 2 +$600/ac
Sapling: deciduous	- objective: large conifer (LWD recruitment and shade). - OPTION 1: release - no extraction; plant cedar and fir at 200 trees/ac. - OPTION 2: commercial thin, plant cedar and Douglas fir at 200 trees/ac.	100 yr.	130 yr.	*-release -* $150/ac *-comm. thin-* $400/ac (@$200/MBF logging and haul; 2 MBF/ ac) *-plant-* $100/ac	-plant-$50/ac	*-comm. thin -* $400/ac (hardwood @ $200/ MBF; 2 MBF/ac)	OPTION 1 -$300/ac OPTION 2 -$150/ac
Sapling: mixed	- objective: large conifer (LWD recruitment and shade). - OPTION 1: release - no extraction, plant cedar and/or Douglas fir to 200 trees per acre - OPTION 2: commercial thin deciduous, plant cedar and/or Douglas fir to 200 trees per acre	50 yr.	80 yr.	*-release -* $150/ac *-comm. thin-* $200/ac (@$200/MBF logging and haul; 1 MBF/ acre) *-plant* $100/ac	*-plant-* $25/acre	*-comm. thin -* $200/ac (hardwood @$200/ MBF; 1 MBF/ac)	OPTION 1 -$275/ac OPTION 2 -$125/ac
Pole: conifer	- objective: large conifer (LWD recruitment and shade). - OPTION 1: no action (best option in silver fir zone "old-growth"). - OPTION 2: thin from lower 2/3 of size class (<12" DBH). no extraction. - OPTION 3: commercial thin from lower 2/3 of size class (<12" DBH).	OPTION 1:50 yr.(0 yrs in silver fir zone "old-growth") OPTION 2:30 yr. OPTION 3:30 yr.	OPTION 1:70 yr. (0 yrs in silver fir zone "old-growth") OPTION 2:50 yr. OPTION 3:50 yr.	*-thin-* $200/ac *-comm. thin-* $1000/ac (@$200/MBF logging and haul; 5 MBF/ ac)		*-comm. thin* $2500/ac (conifer @ $500/MBF; 5 MBF/ac)	OPTION 2 -$200/ac OPTION 3 +$1500/ac

Pole: deciduous	- objective: large conifer (LWD recruitment and shade). - OPTION 1: no action (best option in silver fir zone "old-growth"). - OPTION 2: thin, no extraction, plant Douglas fir and cedar to 200 trees/ac. - OPTION 3: commercial thin , plant Douglas fir and cedar to 200 trees/ac.	OPTION 1:> 200 yr. OPTION 2:100 yr. OPTION 3:100 yr.	OPTION 1:> 200 yr. OPTION 2:130 yr. OPTION 3:130 yr.	*-thin-* $200/acre *-comm. thin-* $1000/ac (@$200/MBF logging and haul; 5 MBF/ acre) *-plant-* $100/ac	*-plant-* $50/ac	*-comm. thin -* $1000/ac (conifer @ $200/MBF; 5 MBF/ac)	OPTION 2 -$350/ac OPTION 3 -$150/ac
Pole: mixed	- objective: large conifer (LWD recruitment and shade). - OPTION 1: no action (best option in silver fir zone "old-growth"). - OPTION 2: thin, no extraction, plant Douglas fir and cedar to 200 trees/ac. - OPTION 3: commercial thin , plant Douglas fir and cedar to 200 trees/ac.	OPTION 1:50 yr. OPTION 2:30 yr. OPTION 3:30 yr.	OPTION 1:70 yr. OPTION 2:50 yr. OPTION 3:50 yr.	*-thin-* $200/ac *-comm. thin-* $1000/ac (@$200/ MBF logging and haul; 5 MBF/ ac) *-plant-* $100/a	*-plant-* $25/acre	*-comm. thin-* $1000/ac (conifer @ $200/MBF; 5 MBF/ac)	OPTION 2 -$325/ac OPTION 3 -$125/ac

by cutting down deciduous trees. Managing for a healthy beaver population may accelerate recovery of off-channel habitats in low gradient mainstems, and may reduce costs associated with silvicultural manipulations designed to promote large conifer forests. The risks are that the beaver will either devour plantation stock or create excessively large impoundments. Prescriptions for active management should be sensitive to beavers and their ecosystem function.

A riparian reference stand (RSS) system based on the riparian conditions (species, age, stocking, valley form, channel type, process domain (Floodplain, steep cascade), and EPA hydrogeomorphic unit (bog, hyporheic) could be implemented as a part of a regional monitoring program (Bisson and Raphael 1996). The Washington State watershed analysis monitoring module (WFPB 1993) could accelerate the development of a reference network through voluntary participation of landowners. The reference system compliments experimentally designed treatments for the variety of active management prescriptions (e.g., thinning, planting, pruning). Controlled experiments with working hypotheses feed the adaptive management process, advancing the knowledge of reliable recovery methods (after Hollings 1978).

Conclusions

Native people have used riparian forests for millennia and yet only in the last century have we developed the technology to remove in excess of 80 percent of native forest cover. With the loss of these forests comes the loss of function they had provided. Protection must extend beyond the lower watershed where buffer strips have often left dysfunctional hardwood stands, dominated by short-lived species while the conifers have been harvested, high-graded for the economic values from the riparian forest. Often the legacy of the previous forest is all that remains, as large dimension and rapidly decaying logs. We are in that twilight where we are even losing these legacies, large stumps are often the only structure that remains of the native forest. The next century will yield the results of our active management. We cannot afford to wait passively for the forest and stream ecosystems to recover while instream species that depend on riparian forests dwindle to extinction.

Literature Cited

Agee, J. K. 1988. Successional dynamics in forest riparian zones. In Streamside Management: Forestry and Fishery Interactions. K.J. Raedeke (ed.). Institute of Forest Resources, University of Washington.

Berg, D. R. 1995. Riparian silvicultural system design and assessment. In the Pacific Northwest Cascade Mountains. Ecological Applications. 5,1:87-96.

Bilby, R. E. 1981. Role of organic debris dams in regulating the export of dissolved and particulate matter from a forested watershed. Ecology 62:1234-1243

Bilby, R. E. and J. W. Ward. 1990. Changes in characteristics and function of woody debris with increasing size of streams in Western Washington. Transactions of the American Fisheries Society. 118:368-378

Bisson, P. L. and M. Raphael. 1996. Riparian ecosystem research: Evaluation of different management approaches to protecting and restoring ecological processes in riparian zones. Pacific Northwest Research Station. Olympia, WA.

Bisson, P. A., J. L. Nielson, R. A. Palmason, and L. E. Grove. 1982. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low stream flow. pp 62-73 In N. B. Armantrout (ed). Acquisition and utilization of aquatic habitat inventory information. Proceedings of a symposium held 28-30 October, 1981, Portland, OR. Western Division, American Fisheries Society.

Bisson, P. A., R. E. Bilby, M. D. Bryant, C. A. Dolloff, G. B. Grette, R. A. House, M. L. Murphy, K. V. Koski, and J. R. Sedell 1987. Large woody debris in forested streams of the Pacific Northwest: past and present. Streamside Management: Forestry and Fishery Interactions. Salo, E. O., and T. W. Cundy.(eds.) Institute of Forest Resources, University of Washington, Seattle, WA.

Bryant, M. D. 1983. Role and management of woody debris in west coast salmonid nursery streams. No. Am. Jour. Fish. Mgmt. 3:322-330

Cederholm, C. J. and W. J. Scarlett. 1982. Seasonal immigrations of juvenile salmonids into four small tributaries of the Clearwater River, Washington 1977-1981. pp 98-110 In E. L. Brannon and E. O. Salo (eds.) Proceedings of the Salmon and Trout Migratory Behavior Symposium. School of Fisheries, University of Washington, Seattle, WA.

Centre for the Study of the Environment (CSE). 1995. Status and Future of Salmon in Western Oregon and northern California. University of California, Santa Barbara, CA.

Collins, B. D., T. J. Beechie, L. E. Benda, P. M. Kennard, C. N. Veldhuisen, V. S. Anderson, and D. R. Berg. 1994. Salmonid habitat and watershed assessment and restoration plan for Deer Creek, North Cascades of Washington. Report by the 10,000 years Institute to Stilliguamish Indian Tribe and Washington Department of Ecology. 232 pp.

Cummins, K. W. 1974. Structure and function of stream ecosystems. Bioscience 24(11):631-641.

Franklin, J. F. and C. T. Dyrness. 1969. Natural Vegetation of Oregon and Washington. USDA Forest Service Res. Pap. PNW 80 Pacific Northwest Forest and Range Experiment Station. Portland OR. 216 pp.

Franklin, J. F., D. R. Berg, D. Thornburg, and J. Tappenier. 1996 Alternative silvicultural approaches to timber harvest: Variable retention harvest system. In Forestry for the 21st Century. Kathryn Kohm (ed.) Island Press. Washington D.C.

Franklin, J. F., T. Spies, D. Perry, M. Harmon, and A. Mckee. 1986. Modifying Douglas fir management regimes for nontimber objectives. In Douglas fir: Stand Management for the Future. C. D. Oliver, D. P. Hanley, J. A. Johnson (eds.) Institute of Forest Resources Contribution No. 55. University of Washington, Seattle, WA.

Grant, G. E., F. J. Swanson, and M. G. Wolman. 1990. Pattern and origin of stepped bed morphology in high gradient streams, western Cascades, OR. Geological Society of America Bulletin 102:320-352.

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

Grette, G. 1985. The role of large woody debris in juvenile salmonid rearing habitat in small streams. M.S. thesis, University of Washington, Seattle, WA. 105 pp.

Harmon M. et al. 1986 The ecology of coarse woody debris in temperate ecosystems. Adv. Ecol. Res. 15:133-302.

Heede, B. H. 1985. Interactions between streamside vegetation and stream dynamics pp 54-58 In Proceedings of Riparian Ecosystems and Their Management: Reconciling Conflicting Uses. Tucson, AZ.

Hollings, C. S. 1978. Adaptive Environmental Assessment and Management. London. John Wiley and Sons.

Keller, E. H. and F. J. Swanson. 1979. Effects of Large organic material on channel form and fluvial process. Earth Surface Processes. 4:361-380.

Lienkaemper G. W. and F. J. Swanson. 1987. Dynamics of large woody debris in old growth Douglas fir forests. Can. J. For. Res. 17:150-156.

Mcdade, M. H., F. J. Swanson, W. A. McKee, and J. F. Frank. 1990. Source distances for course woody debris entering small streams in Western Oregon and Washington. Can. Jour. of For. Res. 20(3):326-329.

Megahan, W. F. 1982. Channel sediment storage behind obstructions in the forested drainage basins draining the granitic bedrock of the Idaho Batholith pp 114-121 In F. J. Swanson, R. J. Janda, T. Dunne, and D. N. Swanston (eds.) Sediment budgets and routing in forested drainage basins. USDA For. Serv. Gen. Tech. Rep. PNW-141 Pac. Northwest Forest and Range Exp. Stn. Portland, OR.

Mitsch, W. J., and S. E. Jorgensen. 1989. Ecological Engineering. Wiley and Sons, Inc. NY. 472 pp.

Naiman, R. J., T. J. Beechie, L. E. Benda, D. R. Berg (and others). 1992. Fundamental elements of ecologically healthy watersheds in the Pacific Northwest coastal ecoregion. pp 127-188 In Watershed Management: Balancing Sustainability and Environmental Change. (R. J. Naiman ed.). Springer-Verlag, NY.

Nilsson, C., G. Grelsson, M Johansson, and U. Sperens. 1989. Patterns of species richness along riverbanks. Ecology 70(1).

Oliver, C. D. and B. C. Larson. 1990. Forest Stand Dynamics. McGraw-Hill, Inc. NY. 467 pp.

Oliver, C. D. and T. M. Hinkley. 1987. Species, Stand Structure and Silvicultural Manipulation Patterns for the Streamside zones. In Streamside Management: Forestry and Fishery Interactions. Salo, E. O., and T. W. Cundy (eds.). Institute of Forest Resources, University of Washington, Seattle, WA.

Oliver, C. D., D. R. Berg, D. R. Larsen, and K. L. O'Hara. 1992. Integrating management tools,

ecological knowledge, and silviculture. pp 361-382 In Watershed Management: Balancing Sustainability and environmental change. (R.J. Naiman ed.). Springer-Verlag, NY.

Potts, D. F. and B. K. M. Anderson. 1990. Organic debris and the management of small stream channels. W. Jour. Appl. For. 5(1):25-28.

Raedeke, K. J. 1988. Ecology of large mammals in riparian systems of the Pacific northwest forests. pp. 113-132 In Streamside Management: Forestry and Fishery Interactions. K.J. Raedeke (ed.). Institute of Forest Resources, University of Washington, Seattle, WA.

Rainville R. P., S. C. Rainville, and E. L. Lider. 1985. Riparian silvicultural strategies for fish habitat emphasis. In Proceedings of 1985 Society of American Foresters National Convention. Ft. Collins, CO.

Reeves, G. H., L. E. Benda, K. M. Burnett, P. A. Bisson, and J. R. Sedell. 1995. A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionary significant units of anadromous salmonids in the Pacific Northwest. American Fisheries Society Symposium 17:334-349.

Salo, E. O. and T. W. Cundy. 1987. Streamside Management: Forestry and Fishery Interactions. Institute of Forest Resources, University of Washington, Seattle, WA.

Sedell, J. R. and J. L. Froggatt. 1984. Importance of streamside forests to large rivers: Isolation of the Willamette River, OR, USA from its floodplain by snagging and streamside forest removal. Internationale Vereinigung fur theoretische und Limnololgie, Verhandlungen 22:1828-1834.

Sedell, J. R., P. A. Bisson, F. J. Swanson, and S. V. Gregory. 1989. What we know about large trees that fall into streams and rivers. 153 pp. In From the forest to the sea: a story of fallen trees. C. Maser, R. F. Tarrant, J. M. Trappe and J. F. Franklin. (Eds.). GTR. PNW-GTR-229 Pacific Northwest Forest and Range Experiment Station, Portland, OR.

Schoonmaker, P. and W. A. McKee. 1988. Species composition and diversity during secondary succession of coniferous forests in the western Cascade Mountains. For. Sci. 34(4):960-979.

Stanford, J. A., and J. V. Ward. 1988. The hyporheic habitat of river ecosystems. Nature 335:64-66.

Sullivan, K., T. E. Lisle, C. A. Dolloff, G. E. Grant, and L. M. Reid. 1987. Stream Channels: The link Between Forests and Fishes. In Salo, E. O., and T. W. Cundy. 1987. Streamside Management: Forestry and Fishery Interactions. Institute of Forest Resources, University of Washington, Seattle, WA.

Swanson, F. J. and G. W. Lienkaemper. 1982. Interactions among fluvial processes, forest vegetation and aquatic ecosystems, South Fork, Hoh River, Olympic National Park. In Ecological research in the National Parks of the Pacific Northwest, E. E. Starkey, J. F. Franklin and J. W. Matthews (eds.) Oregon State University. Corvallis, OR.

Swanson, F. J. and D. R. Berg. 1991. Ecological roots of new approaches to forestry. Forest Perspectives. 1(3):6-8.

Swanson, F. L., S. V. Gregory, J. R. Sedell, and A. G. Campbell. 1982. Land and water interactions: The riparian zone. In R.L. Edmonds (ed.) Analysis of coniferous forest ecosystems in the western United States. Hutchinson Ross Publishing Co., Stroudsburg, PA.

Triska, F. J., J. R. Sedell, and S. V. Gregory. 1982. Coniferous forest streams. In R.L. Edmonds (ed) Analysis of coniferous forest ecosystems in the western United States. Hutchinson Ross Publishing Co., Stroudsburg, PA.

Trotter, E. H. 1990. Woody debris, forest-stream succession, and catchment geomorphology. Jour. No. Am. Benthol. Soc. 9(2):141-156.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37:130-137.

Washington Forest Practices Board (WFPB). 1993. Standard Methodology for Conducting Watershed Analysis. Washington State Department of Natural Resources. Olympia, WA.

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The Relationship of Large Woody Debris, Riparian Vegetation, and Landform in Small, Low Gradient Streams

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Riparian and stream restoration too often is attempted without an appreciation for how large woody debris (LWD) dynamics and riparian composition are affected by landform and fluvial processes. A total of twenty one sites were studied within mature and old forests of southern Washington state. Stream channels were delineated by bed morphology into three classes: bedrock, plane-bed, and forced pool-riffle. Geomorphic landform was used to approximate the relative fluvial influence upon the riparian plant community. Landform was divided into three classes based upon height above the active channel (floodplain 0 to 1 m above the channel, low terrace to 1 to 3 m, high terrace > 3 m), with slope as the fourth class. Tree density by species (trees per hectare) characterized the riparian forest while the understory was delineated with Detrended Correspondence Analysis. Valley constraint was defined as the ratio of valley width to active channel width, at 3 m above the active channel.

Large woody debris volume was significantly influenced by valley constraint within forced pool-riffle channels with volume increasing as a power function of decreasing valley constraint ($y=15.854c^{0.1559x}$, $r^2=0.58$). No pattern was observed for bedrock or plane-bed channels. The presence of off-channel habitat for aquatic organisms increased exponentially with decreasing valley constraint for all channel types ($0.843^{(0.085x)}$, $r^2=0.71$).

Fluvial disturbance had a decreasing impact on riparian succession and community composition with increasing elevation above the channel. In the overstory, deciduous species dominated floodplain landform with red alder (*Alnus rubra*) at 62 percent of stems. Conifers were more common on higher landforms. Upland disturbances controlled riparian succession on low and high terraces, while fluvial disturbances controlled floodplain riparian succession.

The diameter of LWD in the channel was related to the diameter at breast height (dbh) of the riparian forest. In old-growth stands (>300 years), LWD diameter was significantly greater ($\alpha=0.05$) than the average forest diameter for all sites. This follows common successional theories for old-growth as the oldest age cohort begins to die out with some of those stems falling into the channel. In younger stands (100 to 300 years), a mixed relationship between LWD and riparian forest diameter reflected LWD sources from previous stands, smaller suppressed stems from the existing stand, and a wide range of diameters contributed through bank erosion.

When planning riparian restoration, the effect of fluvial disturbances and channel migration must be factored into the plan. The floodplain landform (in many cases) can be used to define the channel migration zone, or the area bounded by historical channel movement. The rate of channel migration must be predetermined prior to restoration. If conifer regeneration is desired on floodplain landform (a risky prospect), then planting should be restricted to safe sites: topographic highs, on nurse logs, and behind stable LWD jams. Finally, all restoration projects include an element of luck. Luck (or a period of mild annual floods) may have accounted for successful natural conifer regeneration, and may be needed for the success of your restoration plan.

Literature Cited

Fetherston, K. I., R. J. Naiman, and R. E. Bilby. 1995. Large woody debris, physical processes, and

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riparian forest development in montane river networks of the Pacific Northwest. Geomorphology. 13:133-144.

Hupp, C. R. 1988. Plant ecological aspects of flood geomorphology and paleoflood history. pp. 335-356 In V. R. Baker, R. C. Kochel, and P. C. Patton (eds). Flood geomorphology. John Wiley and Sons, New York, NY.

Montgomery, D. R. and J. M. Buffington. 1993. Channel classification, prediction of channel response, and assessment of channel condition. Washington State Timber-Fish-Wildlife agreement. Report TFW-SH10-93-002. 84 pp.

Rot, B. W. 1995. The interaction of valley constraint, riparian landform, and riparian plant community composition size and age upon channel configuration of small, low gradient streams of the western Cascade Mountains, Washington. M.S. Thesis, University of Washington, Seattle, WA. 67 pp.

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Location and Design Influence Hydrologic Dynamics of Wetland Mitigation Projects

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Hydrologic conditions are critical to the establishment, function, and persistence of wetlands, and are perhaps the most important element influencing the successful development of mitigation projects. Despite the recognized importance of hydrology, it remains one of the least studied attributes of wetlands (Kusler and Kentula 1990). To achieve wetland management objectives that incorporate restoration techniques, it is necessary to develop an understanding of how hydrology influences other structural and functional attributes of wetlands, and to understand how design and siting of projects will influence their hydrologic regimes.

A goal of the U.S. EPA's Wetlands Research Program (Leibowitz et al. 1992) is to assess the extent to which wetland mitigation projects are replacing the structure and function of naturally-occurring wetlands, and to provide technical data to support development of design and siting guidelines for wetland mitigation projects. As one element of an integrated assessment of naturally-occurring wetlands and wetland mitigation projects within the Wetlands Research Program (Magee et al. 1993), a 3-year project was initiated in the fall of 1993 to monitor water levels in approximately 50 freshwater, emergent wetlands (naturally-occurring wetlands and wetland mitigation projects) in the Portland, Oregon, metropolitan area. The study wetlands are located in diverse land use and hydrogeomorphic settings, ranging from small slope wetlands, to riparian corridors of urban streams, to wetlands on the floodplain of a large, highly-regulated river system.

Data from the first year of monitoring demonstrate wide variability in water level dynamics within and among study wetlands, suggesting important differences in the control of short and long-term variability in water levels (i.e., individual storm events and interannual variability). During individual storm events, land use setting appears to be the most important factor affecting changes in wetland water levels. Both naturally-occurring wetlands and wetland mitigation projects in urbanized settings experience large (up to 1 m) changes in water levels, with rapid rise and recession. Stage increases are much smaller in wetlands in rural areas, and recessions are considerably slower than in urban systems. The differences are presumably attributable to extensive landscape modifications in the urban areas (extensive impermeable areas, channelized runoff, etc.) that route water quickly to wetlands and stream channels. The association between land use and hydrology suggests that location constrains levels of hydrologically-related functions (e.g., flood storage), and further suggests that wetlands should be designed so that functions are compatible with extant site conditions. Further, where specific functions are desired, projects should be placed in settings likely to maximize potential levels of the desired functions.

On an annual basis, there are pronounced differences in hydrologic regimes of naturally-occurring wetlands and wetland mitigation

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projects that are closely tied to project design and wetland morphology. Differences are related most notably to the presence and type of water control structure (excavated pond, impoundment) in the wetland. During 1994, wetland mitigation projects had more extensive and persistent standing water than naturally-occurring wetlands, observations consistent with previous research in other geographic areas (e.g., Confer and Niering 1992). Results reflect the fact that most naturally-occurring wetlands are marshes or meadows in which water regimes are unregulated, while most wetland mitigation projects are ponds with structural modifications designed to maintain standing water and regulate water levels. The annual range in water levels (excluding storm events) in 1994 was much larger in naturally-occurring wetlands than in wetland mitigation projects (1.09 vs. 0.56 m) and in some wetland mitigation projects water levels varied by less than 10 cm during the year. Average water depths were much higher in wetland mitigation projects than in naturally-occurring wetlands (0.75 vs. 0.30 m), and the average extent of flooding in wetland mitigation projects was about twice that in naturally-occurring wetlands (52 vs. 26 percent of wetland area). Standing water was present throughout the year on at least part of the wetland in 80 percent of wetland mitigation projects, but only 20 percent of naturally-occurring wetlands.

Results show that design of wetland mitigation projects is resulting in hydrologic regimes that are fundamentally different from those of the naturally-occurring wetlands they were built to replace, and that design of wetland mitigation projects is leading to changes in the relative abundance of wetland types for freshwater emergent wetlands in the Portland area. Given the influence of hydrology on other wetland attributes, changes in wetland type are almost certainly resulting in wetlands with soils, vegetation, and habitat functions that are different from those of naturally-occurring wetlands in the region. Results suggest that, as advocated by Kentula et al. (1992), wetland managers need to recognize the intimate relationships between wetland structure and function; if functional replacement of wetlands is the desired endpoint of management activities, modification of current project designs is necessary.

Literature Cited

Confer, S. R., and W. A. Niering. 1992. Comparison of created and natural freshwater emergent wetlands in Connecticut. (USA) Wetlands Ecology and Management 2:143-156.

Kentula, M. E., R. P Brooks, S. E. Gwin, C. C. Holland, A.D. Sherman, and J. C. Sifneos. 1992. An Approach to Improving Decision Making in Wetland Restoration and Creation. Edited by A.J. Hairston. EPA/600/R-92/150. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Kusler, J. A. and M. E. Kentula. 1990. Executive summary. p. xvii-xxv In J. A. Kusler and M. E. Kentula (eds.) Wetland Creation and Restoration: The Status of the Science. Island Press. Washington, D.C.

Leibowitz, S. G., E. M. Preston, L. Y. Arnaut, N. E. Detenbeck, C. A. Hagley, M. E. Kentula, R. K. Olson, W. D. Sanville, and R. R. Sumner. 1992. Wetlands Research Plan FY 1992-96: An Integrated Risk-Based Approach. EPA/600/R-92/060. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Magee, T. K., S. E. Gwin, R. G. Gibson, C. C. Holland, J. Honea, P. W. Shaffer, J. C. Sifneos, and M. E. Kentula. 1993. Research Plan and Methods Manual for the Oregon Wetlands Study. EPA/600/R-93/072. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

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A Portland, Oregon, Landscape Scale Prescription for Rare Wetland Wildlife

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Abstract: **Basic landscape ecology principles were used to develop a site-specific strategy for maximizing the chances for long-term survival of several wetland-associated wildlife species. This application used existing data on the area requirements of eight locally rare wildlife species with strong associations with wetlands. The eight species were classified into habitat and area size requirement groups. National Wetlands Inventory maps were used to identify potential areas of wetland habitat on a satellite photo. The primary areas of potentially suitable habitat identified for the eight species were the lowlands and islands along the Columbia River which included areas of pasture, former pasture, forest, agricultural land, remnant sloughs and lakes, residential, commercial and industrial areas. Patches of potential habitat ranged in size from over 3.5 square kilometers in size to those smaller than 50 hectares. It was concluded that there was an abundance of potential habitat for these species in this area and that interconnections between patches could be restored or maintained. This general approach identifies specific sites that should be considered for acquisition by entities such as the local regional government, METRO, (for its Greenspaces program), park districts, and state agencies. It also reveals where interconnections between sites should be considered in land use and development decisions by regulatory agencies. These sites could also be used as regional wetland mitigation banks if wetland restoration and enhancement activities would be appropriate to provide or substantially improve habitat quality.**

Introduction

A number of wildlife species strongly associated with wetlands were identified as rare in the Portland, Oregon metropolitan area by the 1990-1992 Metro Greenspaces Inventory (Poracsky et al. 1992), and from other sources such as the Oregon Natural Heritage Program (1993) and the Portland Audubon Society's Checklist of Portland Birds. One of the goals of the Greenspaces Master Plan (METRO 1992) is to ensure the long-term survival of as many indigenous species of plants and wildlife as possible in the Portland urban area. The Master Plan identified three general landscape-scale restoration principles that could be applied throughout the area: (1) consolidate habitat areas; (2) restore contiguous riparian vegetation; and (3) restore connections between watersheds at headwaters. The Plan also included an overall description of each watershed within the study area in terms of its degree of urbanization, types and extent of natural areas remaining, and development pressures. Recommendations on natural area conservation within each watershed were also made, but often were not very site specific.

Without some kind of prioritization and attention to species-specific habitat needs, application of these three principles is unlikely to meet the needs of at least some rare species of concern. The logical next step, therefore, would be to identify the habitat and area requirements of a small number of rare wetland species thought to represent the habitat requirements of most. This habitat information could be used to develop a landscape-scale prescription that could enhance the likelihood of long-term survival of these species in the urban area through a combination of restoration, preservation, enhancement, and creation of the habitats they require. Additional population declines might then be avoided and eventually, population increase and even successful reintroduction of extirpated species might be possible.

There are a number of planning and acquisition level projects by various levels of governments in the Portland area which, in combination, could greatly enhance the chances of long-term survival of these species. This analysis was designed to identify high priority areas for wetland conservation efforts to

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benefit these species, to identify the likelihood of successfully applying the prescriptions, and to suggest research on solving some of the difficult problems these species face in a dynamic urban environment.

Methods

Study Area

This analysis covers the Portland, Oregon urban and urbanizing area, located along the south shore of the Columbia River and including islands and the mouths of the Willamette and Sandy Rivers (Figure 1). The only connection between the Cascades Range on the east and the Coast Range on the west lies adjacent to the Columbia River and in the agricultural lands either many miles to the south of Portland or to the north of Vancouver. Habitats and land uses along the south side of the Columbia River, from east to west, include: (1) the Sandy River Delta, a large delta of former pasture and bottomland hardwoods on the west side and which comprises the western end of the Columbia Gorge National Scenic Area; (2) an area rapidly changing from row crops and pasture to residential, commercial and industrial area still traversed by stream corridors, sloughs, remnant lakes and wetlands, with the heaviest industrial development west of the Portland International Airport and extending to the mouth of the Willamette River; (3) predominately agricultural lands and remnant wetlands on Sauvie Island, a large island separated from the shore by the Multnomah Channel of the Willamette River; and (4) on the mainland the forested peninsula of the Tualatin Mountains, which extend deeply into urban Portland from the Coast Range, including the 2,000-ha Forest Park, the largest urban park in the United States.

The density of development in the central portion of Portland is high and most streams have been put in underground channels. The riparian corridor along the Willamette River is absent in the most intensively developed areas but is present as at least a narrow strip of vegetation in the remainder. Several natural parks are located along the Willamette and on its islands within the Portland area. Riparian corridors along other major rivers and smaller streams are variable as well.

Wildlife Species

Eight rare wetland-dependent wildlife species were selected for this analysis. The main criteria used to select species were that they were rarely encountered during the Metro Greenspaces field surveys (Poracsky et al. 1992), were considered to be rare or sensitive according to the Oregon Natural Heritage Program (1993), were listed as rare on the Portland Audubon Society's Portland Bird Checklist, or were considered to be rare by local wildlife ecologists and birders.

The wetland habitats commonly used by the selected species were identified on the basis of the Geenspaces data, published literature, and observations by local ecologists and birders. References such as Brown (1985) were reviewed, and information on home range/territory size and habitat requirements was summarized. On the basis of habitat use, the eight species were placed into a smaller number of basic habitat-related groups representing combinations of wetland and upland habitats.

Potential Habitat Identification and Mapping

A color poster, derived from 1992 satellite data and published by Satellite Images, Inc. (reproduced by permission of Satellite Images, Inc.), was used as the base map for this analysis because it provided an overview of the entire urban area and its vegetation. The National Wetlands Inventory maps (U.S. Fish and Wildlife Service, various dates) covering the urban area were examined and the largest areas of mapped wetlands, 50 ha or more in size, were marked on an overlay of the satellite image. The satellite image base provided information on the basic vegetation type(s) for each wetland area as well as the character of the upland area surrounding each wetland.

Habitat Availability and Suitability Analysis

Based on size, vegetation, location, and surrounding land use, the general suitability of each potential area of habitat was assessed for each of the eight species. This was compared to the known range of occurrence of each species in the Portland area. The general status of each species and the challenges it faces in the urban environment was described.

Future Research

Suggestions of research projects that should precede reintroduction or population enhancement efforts for some species were made, based on the results of the analysis.

Results

Wildlife Species

The species selected were the northern harrier *(Circus cyaneus* L.), Virginia rail *(Rallus limicola* Vieillot), sora *(Porzana carolina* L.), American bittern *(Botaurus lentiginosus* Rackett), black-crowned night heron *(Nycticorax nycticorax* L.), muskrat *(Ondatra zibethica* L.), western pond turtle *(Clemmys marmorata* Baird and Gerard), and western painted turtle *(Chrysemys picta* Schneider). All of these species have been adversely affected by habitat degradation, including water pollution problems such as algae blooms and oil spills, water level fluctuations due to stormwater runoff, reed canarygrass invasion of emergent areas, habitat fragmentation (especially for less mobile species such as the two turtle species); predation of turtles and nests by introduced bullfrogs *(Rana catesbeiana* Shaw) and other predators; and in the case of the muskrat, potential competition with exotic nutria *(Myocastor coypus* E. Geoffroy St.-Hila'lre).

Habitat Relationships

The basic habitats utilized by the eight species fit into four general groupings as follows: wet and dry meadow and emergent–northern harrier; emergent and open water–Virginia rail, sora, American bittern, muskrat; emergent, open water and forest–black-crowned night heron; open water, emergent and adjacent uplands–western pond and painted turtles (Godfrey 1966, Perry 1982, Brown 1985, Boutin and Berkenholtz 1987, Holland 1994, Storm and Leonard 1995).

According to Brown (1985), northern harrier breeding home ranges vary from 100 to 890 ha, and winter home ranges vary from 12 to 260 ha; published home range sizes for the bittern and black-crowned night heron were not found in the literature but probably intermediate between harrier and the smaller rails; Virginia rail and sora home ranges during nesting are in the < 1 ha range, with sora density reported as up to 30 birds per square kilometer.

Muskrat home ranges are reported as usually <1 ha but up to 2.4 ha, with a minimum habitat per population of 1.6 km of stream (Brown 1985). Boutin and Birkenholz (1987) report home range size of individual muskrats in the < 1 ha to 1 ha range and point out that home range size appears to vary with habitat quality. Densities of muskrats were in the 10-80 animals per ha range in large marshes and were much lower along water courses, ranging from 1 pair per 46 m of shoreline to 3 or 4 pairs per 1609 m (Perry 1982).

The home ranges of individual western pond turtles are reported as less than 1 ha by Bury (1972). Data on home range size of the western painted turtle could not be found. For purposes of this analysis, painted turtle home ranges were assumed to be similar to pond turtles.

On the basis of this review, the eight wildlife species were grouped into three home range size categories differing by an order of magnitude: individual pairs of harriers–over 100 ha; bittern, black-crowned night heron, and muskrat colonies–10 to 100 ha; and individual Virginia rails and soras, and colonies of western pond turtle and western painted turtle–< 1 to 10 ha.

Potential Habitat Identification

The larger wetland areas identified in the National Wetlands Inventory maps are outlined in Figure 1. These are the areas where opportunities might exist to enhance wetland and upland habitats in terms of quantity and quality for the eight species of concern. The largest areas are concentrated in the lowlands along the Columbia River, including around three remaining large lakes: Sturgeon, Vancouver, and Smith-Bybee; on islands and along stream corridors and Sandy River Delta. Forested areas appear dark in this Figure while some croplands appear nearly white and are also indicated by relatively large block size. The urbanized area appears a mottled gray color, although the rural lands north of Vancouver, Washington appear to be more urbanized than they actually are.

The larger areas of wetlands around the lakes consist of a combination of emergent, scrub-shrub, and forested patches adjacent to these large areas of open water. In the case of Sturgeon Lake on Sauvie Island, these wetlands are surrounded by agricultural land which is expected to remain agricultural over the long term as a result of current zoning and Oregon's land

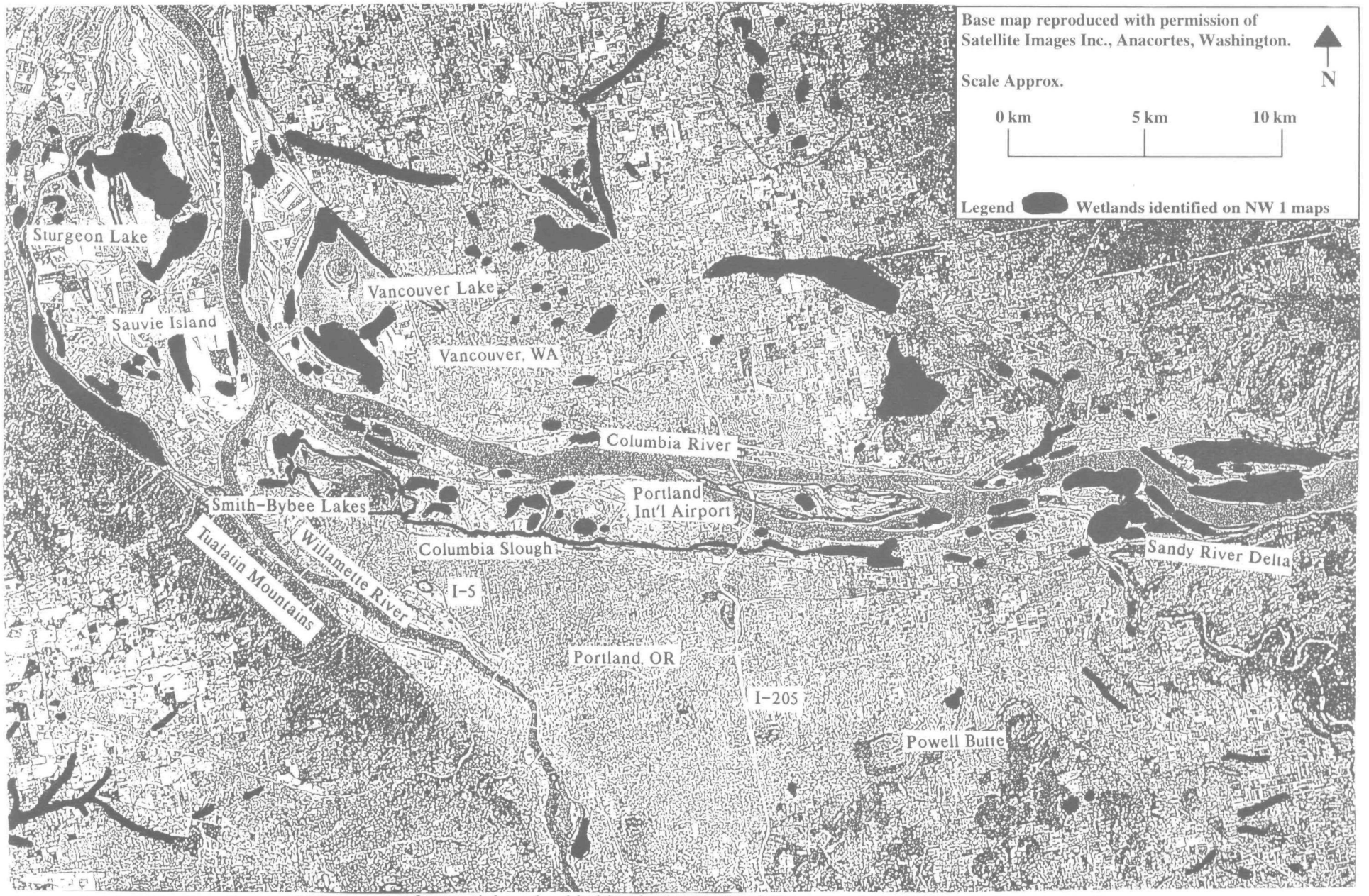

Figure 1. Satellite-based map of large areas of potential wetland habitat in the Portland, Oregon area.

use law. The areas adjacent to Smith-Bybee are intensively urban and mostly industrial, and development has occurred or is rapidly occurring in the area between Smith-Bybee Lakes and the Sandy River Delta to the east. The Columbia Slough, however, and wetland mitigation activities in the area between the airport and Sandy River Delta could maintain a connection between the Delta and Smith-Bybee Lakes, focused on the Columbia Slough.

Habitat Availability and Suitability Analysis

Harriers are regularly observed year-round at Sauvie Island and around Vancouver Lake, where there is an abundance of wet and dry meadows. The area along the Columbia River east of Sauvie Island did provide habitat but rapid development is expected to eliminate this habitat. The only other remaining areas of suitable habitat large enough to support breeding pairs are Powell Butte (a City Park), and the Sandy River Delta, now under U.S. Forest Service ownership. The Management Plan for the Sandy River Delta (USDA Forest Service 1995) calls for rehabilitation and creation of upland and wetland meadows, including several strategies for control of reed canary grass, in the 100-ha or greater size range, which should provide nesting habitat for harriers over the long term.

The bittern, black-crowned night heron, rails, and muskrat all utilize open water/emergent complexes. The night heron also requires forested areas for roosting. These areas are also prevalent on Sauvie Island, in the Smith-Bybee Lakes area, in some patches near the Columbia and along the Willamette River, and in the Sandy River Delta. Emergent wetlands in at least some proportion of all of these areas are heavily invaded by reed canarygrass which does not appear to be as suitable as habitat for these species as other emergents such as cattail *(Typha latifolia* L.) and bulrush *(Scirpus).*

The western painted turtle is fairly widely distributed in the lakes, sloughs, and islands along the south side of the Columbia. The western pond turtle, however, is rarely observed. The relatively large population of introduced bullfrogs combined with habitat degradation is suspected to be a factor in the decline of the pond turtle, along with a loss of suitable undisturbed upland nesting habitat adjacent to remaining wetland areas.

Discussion

Maintenance of wet and dry meadow habitat blocks in the 100-ha size range which are not disturbed during the nesting season by mowing, plowing, and at least somewhat protected from human and dog access would be an appropriate goal for harrier habitat management in all of these areas. A secondary but no less important goal should be to control reed canary grass *(Phalaris arundinacea* L.) as much as possible because prey availability and thus habitat suitability is low owing to the height and density of reed canary grass, according to a USFWS (undated) draft Habitat Evaluation Procedures model for the northern harrier. Late summer mowing could still occur to prevent invasion by trees and shrubs. Upland meadows need to be included, as nesting occurs on the ground in uplands as well as wetlands.

Introduced nutria may be acting as a keystone species in creating and maintaining monocultures of reed canary grass in many urban wetlands in the Portland area, particularly in the floodplains of local streams. This suggestion is based on personal observations of major changes in emergent species composition at created and natural wetlands in the Portland area concurrent with rapid nutria population growth. Reed canary grass control measures might be more successful if nutria populations were also controlled. A valuable experiment would be to test identical methods of physical/chemical control of reed canary grass with and without nutria population control.

Habitat for the two rail species, bittern, night heron, muskrat, and two turtle species overlaps considerably and usually occurs in complexes that include all of the elements needed–open water, emergent vegetation, scrub-shrub, forested wetland, and adjacent uplands. Preservation of the large blocks of habitat needed by an individual harrier pair could also benefit these other species if the harrier habitat includes marshes, lakes, and channels. In addition, many of the areas smaller than 100 ha shown on Figure 1 could provide suitable habitat for populations of these smaller, less wide-ranging wildlife species. The habitat components present and/or which could be restored at individual sites would determine which of the above species would find suitable habitat.

Connections between habitat patches are probably not so important for harriers and other birds because they are highly mobile. Aquatic connections should be present between smaller patches to provide avenues of movement by species such as muskrats and the two turtle species.

Literature Cited

Anonymous. Undated. Draft Habitat Suitability Model for the Northern Harrier. Unpublished draft manuscript provided by U.S. Fish and Wildlife Service. 1 p.

Boutin, S. and D. E. Birkenholz. 1987. Muskrat and Round-tailed Muskrat. pp. 314-325 In M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch (eds). Wild Furbearer Management and Conservation in North America. Ontario Ministry of Natural Resources, Toronto. 1150 pp.

Brown, E. R. (ed.) 1985. Management of wildlife and fish habitats in forests of western Oregon and Washington. Pacific Northwest Region, Forest Service, USDA, Publication No. R6-F&WL-192-1985. 2 Vols.

Bury, R. B. 1972. Habits and home range of the Pacific pond turtle *(Clemmys marmorata)* in a stream community. Ph.D. Dissertation, Univ. Calif. Berkeley (not seen, cited in Holland 1994).

Godfrey, W. E. 1966. The Birds of Canada. Nat. Mus. Can. Bull. No. 203, Biol. Series 73. 428 pp.

Holland, D. C. 1994. The western pond turtle: habitat and history - Final report. Prepared for U.S. Department of Energy, Bonneville Power Admin., Portland, OR. 279 p. plus appendices.

Metropolitan Service District (METRO). 1992. Metropolitan Greenspaces Program Master Plan. METRO, Portland, OR.

Oregon Natural Heritage Program. 1993. Rare, threatened and endangered plants and animals of Oregon. Oregon Natural Heritage Program, Portland, OR. 79 pp.

Perry, H. R., Jr. 1982. Muskrats. pp. 282-325 In Chapman, J. A. and G. A. Feldhamer (eds). Wild Mammals of North America, Biology, Management, and Economics. John Hopkins Univ. Press, Baltimore. 1147 pp.

Poracsky, J., P. L. Sharp, and E. Lev. 1992. Metropolitan Greenspaces Program Data Analysis: Field Based Biological Data Final Report. Prepared for METRO, Portland, OR. 142 pp.

Portland Audubon Society. Checklist of Portland Birds. Portland, OR.

Satellite Images, Inc. 1992. Portland-Mt. St. Helens from Space (Poster). Satellite Images, Inc., P.O. Box 1257, Anacortes, WA 98221.

Storm, R. M. and W. P. Leonard. 1995. Reptiles of Washington and Oregon. Seattle Audubon Society, Seattle, WA. 176 pp.

U.S. Fish and Wildlife Service. various dates. National Wetland Inventory U.S.G.S. 7.5-minute Quadrangle Maps: Portland, Mt. Tabor, Linnton, Sauvie Island, Sandy, Damascus, Beaverton, and Lake Oswego, Oregon; Vancouver, Camas, and Washougal, WA.

USDA Forest Service. 1995. FEIS for the Sandy River Delta Plan. USDA Forest Service, Columbia River Gorge National Scenic Area, Hood River, OR.

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Criteria for the Restoration and Creation of Wetland Habitats of Lentic-Breeding Amphibians of the Pacific Northwest

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Abstract: **I describe watershed features and wetland design guidelines to encourage colonization and self-sustaining populations of lentic-breeding amphibians at wetland restoration and creation sites. I describe breeding, feeding and refuge habitat. I also characterize migration corridors between these habitat components and describe requirements for dispersal habitat. I provide criteria based on a review of conservation biology and landscape ecology, amphibian natural history literature, and on my own studies of lentic-breeding amphibians in the Puget Sound Basin. Specifically, I suggest area, lengths, widths and quality of refuge and movement habitats. I also describe a hierarchy of explicit wetland attributes, including current velocity, minimum water depths and water level fluctuation, and open water and vegetation requirements to encourage amphibian breeding. Finally, I present criteria for wetland size, orientation, configuration and buffer condition to further maximize spawning success.**

Introduction

Success of wetland mitigation depends on replacing the range of physical and biologic functions that have been lost. Most restoration to date has been directed towards creating appropriate hydrology (Kusler and Brooks 1988) and native vegetation (Galatowitsch and van der Valk 1994), the most fundamental and crucial wetland characteristics (Mitsch and Gosselink 1993). Methods for improving waterfowl production have also been addressed, but primarily for wetlands in the prairie pothole and midwest region of the United States (Weller 1990). Techniques for restoring habitat for a greater diversity of wildlife, and in other geographic locations, have only recently been reported (Hammer 1992, Payne 1992, Kent 1994a, Kent 1994b, Wheeler et al. 1995); yet a bias remains for developing avifaunal habitat (Galatowitsch and van der Valk 1994, Ward et al. 1995). Minimal attention is focused on amphibians or other non-avian aquatic wildlife, and on landscape requirements for such taxa.

Paradoxically, with reported advances in providing wildlife functions at restoration wetlands, concern about amphibian extinctions and declines has grown (Corn 1994, Stebbins and Cohen 1995, Olson and Leonard 1997, among others). The dominant amphibian assemblages of concern in many of these writings are those using wetlands and other lentic habitats for breeding. Scientists believe there is no single reason for amphibian losses, but rather that there are a multitude of factors potentially involved. In the Pacific Northwest, factors include pathogenic fungi (Blaustein et al. 1994a), Ultraviolet B radiation (Blaustein et al. 1994b, 1995b), agricultural runoff (Boyer 1993, Boyer and Grue 1995) and the introduction of exotic animals (Hayes and Jennings 1986).

Habitat loss and deterioration, however, is considered by many to be one of the foremost reasons for wetland-breeding amphibian declines (Stebbins and Cohen 1995). In watersheds undergoing development (e.g., urbanization) and other disturbance (e.g., logging, grazing), upland habitats essential for land-dwelling stages may be severely altered (Minton 1968, Richter and Azous 1995, Olson and Leonard 1997, Delis et. al. 1996). Specifically, feeding and refuge (e.g., aestivation and hibernation) patches may be destroyed or fragmented. Consequently, movement between these patches and breeding sites (e.g., wetlands) may be hindered, separating individuals from these environments and from each other, and thereby increasing the risk of extinction (Blaustein et al. 1994c, Sjögren-Gulve and Ray 1996).

Wetlands themselves may be filled, drained and otherwise changed by land-use conversions that modify their hydrology, water quality and vegetation (Booth 1991, Booth and Reinelt 1993, Taylor 1993, Booth and Jackson 1994, Holland et. al. 1995), destroying amphibian breeding sites. For example, there is evidence that in watersheds with

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proportionately greater development amphibians occur in fewer wetlands and the quality of breeding habitat remaining may be poorer than in wetlands less affected by runoff (Minton 1968, Richter and Azous 1995, Azous and Richter 1995). Collectively, these activities are hypothesized to account for decreasing ranges and dwindling numbers of the Oregon spotted frog (*Rana pretiosa*), (Orchard 1992, Richter and Azous 1995, Azous and Richter 1995, McAllister and Leonard 1990, 1997), western toad (*Bufo boreas*) (McAllister personal communication, Leonard, personal communication) and possibly other lentic-breeding species (Olson and Leonard 1997).

Amphibian declines attributable to habitat losses, however, can be stabilized and reversed by watershed protection and wetland restoration if factors responsible for successful population survival and breeding biology are identified and provided. I am unaware, however, of any scientific literature that specifically targets amphibians for wetland restoration at a collective watershed-wetland scale. Nor am I familiar with papers that systematically suggest and quantify habitat characteristics that may be used as wetland restoration guidelines. Consequently, the objectives of this paper are to review the literature and identify the diverse biogeographic and wetland habitat requirements of amphibians, and also to quantify wetland site-selection and design criteria targeting Puget Sound Basin amphibians. Hopefully, these restoration criteria may apply to behaviorally similar species elsewhere in temperate climates.

Methods

I develop watershed habitat recommendations from a review of conservation biology (e.g., Saunders et al. 1987, Saunders and Hobbs 1991, Forman 1995), population ecology (e.g., Gill 1978a, 1978b; Sjögren 1994, Sjögren-Gulve and Ray 1996), natural history (e.g., Nussbaum et al. 1983, Blaustein et al. 1995a), and herptile biology (e.g., Duellman and Trueb 1986, Stebbins and Cohen 1995) literature of relevance to lentic-breeding amphibians. I specifically searched for empirical data on habitat use and microclimate associations, to determine conditions that maximize wetland colonization and continued use by amphibians.

I also review spawning habitat use by wetland-breeding amphibians (e.g., Nussbaum et al. 1983, Blaustein et al. 1995a, among others), to determine requirements for successful egg development. Moreover, I suggest design criteria based on 1988 through 1996 searches of amphibian breeding habitat at 23 wetlands in the Puget Sound Basin (Azous and Richter 1995, Richter and Azous 1995, Richter and Leonard unpublished data, Richter and Roughgarden unpublished manuscript), and on pitfall trap data at 19 Puget Sound wetlands censused from 1988 through 1995 (Richter and Azous 1995). These surveys sampled up to seven species at a site including the Pacific Northwestern salamander, *Ambystoma gracile* Baird; long-toed salamander, A. *macrodactylum* Baird; rough-skinned newt, *Taricha granulosa* Skilton; red-legged frog, *Rana aurora* Baird & Girard; bullfrog, *Rana catesbeiana* Shaw, western toad, *Bufo boreas* Baird & Girard; and Pacific treefrog, *Hyla regilla* Baird & Girard.

Design criteria also are based on empirical evidence from field experiments of habitat and egg manipulations in *A. gracile* and *R. aurora* (Richter and Roughgarden unpublished manuscript, Richter unpublished data), and on oviposition literature of Pacific Northwestern and other temperate-latitude North American and European wetland-breeding species.

Finally, I assume that recurrently identified amphibian breeding habitat features are beneficial to spawning and egg survival (i.e., reproductive fitness), and that characteristics that substantially differ from identified features are detrimental to colonization and self-sustaining populations at restored and created wetlands.

Results

The historic literature has generally described wetland-breeding amphibian habitat associations during either aquatic or land-dwelling stages. The recent literature augments this knowledge by emphasizing that amphibians respond to habitats at several integrated landscape scales, ranging from specific wetland features favored for breeding to broader watershed attributes. The former include oviposition sites and the latter include feeding and refuge patches in addition to the connecting habitat between them (Brown 1997, Sjögren-Gulve and Ray 1996). Also, recent population biology and conservation theories of amphibian distributions are incorporating metapopulation and source-sink models to explain the spatial and temporal distribution of viable, self-sustaining populations. Fundamental to these concepts is the notion that animals use more or less discrete, suitable feeding and refuge patches within their home range, and further require

favorable migration habitat between home range patches, as well as unhindered dispersal habitat leading from natal wetlands to new breeding sites.

Site-Selection Criteria

Feeding and Refuge Habitats

Feeding and refuge habitats are critical to the land-dwelling life stages of wetland-breeding amphibians. In general, many Pacific Northwest species spawn in wetlands during only a few weeks or months and spend most of the year in terrestrial (upland) feeding and refuge habitats (See Blaustein et. al. 1995a, Olson and Leonard 1997 for an overview of breeding chronology and aquatic and terrestrial habitats used by amphibians associated with lentic habitats). Some low-elevation species or individuals of normally migrating species (*R. aurora, Taricha granulosa*), as well as most high-altitude species (*R. cascadae, A. macrodactylum*), however, may remain at wetlands in which they breed.

The size and characteristics of feeding and refuge habitats away from wetlands (i.e., home range) are functions of climatic condition and habitat structure. Amphibians, in general, prefer cool, humid conditions (Duellman and Trueb 1986, Kleeberger and Werner 1983) with *T. granulosa* and *A. gracile*, often reaching their highest abundance in relatively cool forests that are not extremely wet (Aubry and Hall 1991). I have trapped the greatest number of amphibians within stands and conditions of cool, stable climate, well-drained soils and high ground humidity (Richter personal observation). Such circumstances maintain moisture in woody debris, leaf litter and duff, and are favored as feeding and refuge sites because they facilitate amphibian respiration and provide more invertebrate food than drier sites without logs and other organic debris. Nevertheless, some wetland-breeding species including *T. granulosa*, some Oregon spotted frogs (*R. pretiosa*), *B. boreas*, and Cascades frog (*Rana cascadea*) may be found in warm, dry, exposed locations, especially early in spring or at high altitudes.

There is a strong correlation of terrestrial-breeding amphibian distribution (Plethodontidae) with large woody debris, dead and decaying wood and organic matter, and other habitat conditions favorable to thermoregulation, foraging, and resting (Welsh and Lind 1988, Walls et al. 1992, Dupuis et al. 1995). Because of similar physiology and food preference, it may be inferred that land-dwelling stages of many lentic-breeding species prefer similar conditions. In fact, Corn and Bury (1991) found higher densities of *T. granulosa,* and Aubry and Hall (1991) larger numbers of *A. gracile* in old-growth stands generally having greater volumes of downed wood than mature and younger stands (Spies and Cline 1988). In contrast, Lehmkuhl et al. (1991) found no relationship between stand age and amphibian richness, but did find amphibians least abundant in clearcuts. Clearly, mature and older forests have more numerous crevices below logs, extensive root zones and unoccupied rodent and other small mammal burrows that may be used as temporary refuge during migration and dispersal (Loredo et al. 1996), feeding (Williams 1970) and for aestivation and hibernation than younger immature forests and timber plantations.

Little empirical data is available for home range sizes (e.g. post-breeding through winter hibernation) of Pacific Northwest species with the exception of *A. gracile*, individually marked and followed by Stringer (Stringer personal communication). She identified many recaptured individuals within 200 m of her surveyed wetland. Home ranges for species in other regions indicate that home ranges are small for most ambystoma salamanders and somewhat larger for ranid frogs and toads (Table 1). Some individual common toads of Europe, *Bufo bufo* L., may even move considerable distances to new feeding and refuge sites several times a summer (Beebee 1983).

Without additional empirical studies presumably it is best to select wetland restoration sites adjacent to, or as close to, forested habitats as possible. Larger forested areas with bigger diameter logs and other coarse woody debris adjacent to some clearings will provide habitats to species with smaller or more diverse home range requirements.

Habitat Contiguity

Dispersal Habitat. The passive colonization and self-sustainable occupation of restored or created wetlands depends on the ability of amphibians to disperse from favorable source sites. Dispersal routes from source populations are also critical when populations are eliminated by stochastic processes such as drought (Pounds and Crump 1994), disease (Bradford 1991), and pollution (Richter personal observation), or when populations produce insufficient offspring to occupy a site permanently (Gill 1978a,b; Sinsch 1992). Finally, amphibians at restoration wetlands may benefit as members of a metapopulation. A metapopulation encompassing

Table 1. Home Range and Dispersal Distances of Selected Amphibians.

Species	Home Range (m^2)[a] Mean	Dispersal Distances (m)[b]	Location	References
SALAMANDERS				
NORTHWESTERN SALAMANDER *Ambystoma gracile*	(<200)	{1,000}	Washington	Stringer pers. comm.
JEFFERSON SALAMANDER *Ambystoma jeffersonii*	------- [1-181] ----- ----- -----	(250) {625} (252) {1,610} {152}	Kentucky Indiana New York Michigan	Douglas & Monroe 1981 Williams 1970 Bishop 1941 in Williams 1970 Wacasey 1961 in Williams 1970
SPOTTED SALAMANDER *Ambystoma maculatum*	[1-39] ------ {12-14}	[0-125] (64) [157-250] [6-220] (150)	Indiana Michigan Kentucky	Williams 1970 Kleeberger & Werner 1983 Douglas & Monroe 1981
MARBLED SALAMANDER *Ambystoma opacum*	[13-226]	[0-450] (194)	Indiana	Williams 1970
MOLE SALAMANDER *Ambystoma talpoideum*	[<1-23]	[81-261]	South Carolina	Semlitsch 1981
SMALL-MOUTHED SALAMANDER *Ambystoma texanum*	[1-5]	[0- 125] (52)	Indiana	Williams 1970
FROGS				
PACIFIC TREEFROG *Hyla regilla*	-----	[500- 914]	California	Brattstrom & Warren 1955
NORTHERN LEOPARD FROG *Rana pipiens*	[68-503]	[34-300]	Michigan	Dole 1965
POOL FROG *Rana lessonae*	-----	{15,000}	Austria/Hungary	Tunner 1991 in Dingle 1996
WOOD FROG *Rana sylvatica*	{60} -----	----- {2,530} (1,126)	Minnesota Virginia	Bellis 1962 Berven & Grudzien 1990
TOADS				
AMERICAN TOAD *Bufo americanus*	----- [50-150]	400 -----	Minnesota Minnesota	Ewert 1969 in Christein & Taylor 1978 Bellis 1959
COMMON TOAD *Bufo bufo*	[50-150] -----	3,000 [70-1,600]	Switzerland Germany	Heusser 1968 Sinsch 1988a
NATTERJACK TOAD *Bufo calamita*	-----	1,400 [100-200]	England	Beebee 1983

[a]Minimum polygon: [Range], (mean), {linear distance}.
[b][Range], (mean), {maximum}.

several wetlands may maintain healthy populations that otherwise could go extinct from inbreeding depression (Sjögren 1991, 1994, Pechmann and Wilbur 1994). In turn, the success of colonization and the movement between breeding, feeding and hibernation sites, is considered to be dependent on the vagility of individual species, the distance between habitat patches, and the environmental characteristics of the intervening landscape.

Given their small size and physiological constraints amphibians are less mobile than other terrestrial vertebrates. Moreover, amphibian dispersal is taxa and age specific. Urodeles regularly disperse up to 1,000 m and anurans sometimes greater than 1,000 m from breeding sites (Table 1). Juveniles are generally more vagile than adults (Williams 1970, Gill 1978b) with metamorphs and young moving to new wetlands. Most adults are philopatric, faithfully returning to breeding wetlands during successive years (Twitty 1958, 1959, 1966; Oldham 1966; Shoop, 1968; Gill 1978a; Breden 1987; Sinsch 1988a, 1992; Berven & Grudzien 1990; Denton and Beebee 1993; Sinsch and Seidel 1995). Some adults even return to breeding sites several years after their obliteration (Heusser 1960, Shoop 1968).

All habitat traits being equal, dispersal to and colonization of a restoration wetland decreases as a negative exponential of distance (Wolfenbarger 1949, Breden 1987, Berven and Grudzien 1990) from source populations. Colonization is also a function of the probability that breeding amphibians actually find the mitigation site which, in turn, is the product of the combined probability that the site is reachable and that dispersers travel towards the site from source wetlands. This collective probability is described by the function $P = E\ [1- \exp\ (- \pi\ pd^2)]$, where E is the percent of an amphibian population emigrating from source wetlands, p is the density of randomly distributed wetlands within a given area, and d is a species' maximum dispersal distance during a single generation (Travis 1994).

Dispersal distances from amphibian recaptures and sightings are presented in Table 1. Theoretical probabilities for successful dispersal under ideal conditions can be calculated from the equation above using Table 1 data and other required landscape and species data. Clearly, the greater and closer the number of source wetlands within the normal dispersal distance of a targeted species, the greater the potential for successful colonization. However, until dispersal distances for all Pacific Northwest wetland-breeding species are known, a 1,000 m maximum distance between source and mitigation wetlands is suggested in undisturbed forested landscapes. This recommendation is based on dispersal distances and gene flow calculations in other species (Gittins et al. 1980, Berven and Grudzien 1990, Gibbs 1993, Beebee 1996). Note however, that successful colonization is significantly reduced below optimum dispersal distances by roads, fences, housing, farmlands, clearcuts and other barriers.

Corridors. Corridors theoretically channel dispersing individuals to newly created and restored wetlands, provide migration routes between wetland breeding, feeding and hibernation patches, and may increase suitable habitat size for amphibians. Routes used by amphibians may vary widely with animal thermal tolerances, often depending on acclimation to specific latitudes, altitudes and other regional or localized climatic conditions (Hutchison and Dupre' 1992 and others referenced therein, Duellman and Trueb 1986). Corridors should provide cool, moist microclimates because amphibians' highly permeable integuments require that most species have access to moist environments to maintain or restore lost body water and to favor adequate cuticular respiration (Shoemaker et al.1992).

Amphibians frequently migrate within distinct habitats between wetlands and uplands during successive years. Semlitsch (1981) found that the Mole Salamander *A. talpoideum*, selected wooded rather than open grassy areas. Kleeberger and Werner (1983) found that the Spotted Salamander *A. maculatum* and Gittins et al. (1980) that the Marbled Salamander A. *opacum* Gravenhorst, and *B. bufo* prefer moist soils. Shoop (1968) found that although *A. opacum* limited migration to a narrow 10-30 m-wide corridor near wetlands, it nevertheless crossed open fields between wetlands and forests. Beneski et al. (1986) found no preference in *A. macrodactylum* for habitat quality during migration.

Most amphibians migrate and disperse during wet conditions and don't seem to distinguish between areas with or without cover during these times. Under drier conditions, however, amphibian physiology, and these studies suggest that animals will be less stressed, and hence prefer to move within cooler, moister forested and vegetated areas. Nevertheless, they will traverse roads and meadows and, when necessary, tolerate intervening unsuitable habitats for limited lengths of time.

Ambystoma spp. frequently exhibit identical migration routes between years (*A. opacum* Douglas and Monroe 1981, *A. maculatum, A. opacum*

Stenhouse 1985; A. *maculatum* Shoop 1965, 1968, Shoop and Doty 1972; *A maculatum* Kleeberger and Werner 1983; *A. macrodactylum* Beneski et al. 1986). Some species, including the eastern narrow-mouthed toad (*Gastrophryne carolinensis* Holbrook), and the striped newt (*Notopthalmus perstriatus* Rafinesque) enter wetlands to breed from all directions, but generally disperse to upland sites along select routes (Dodd and Charest 1988). Conversely, others (*B. bufo)* migrate to wetlands in a straight line, but move away toward upland sites along more undefined paths (Sinsch 1988a). Consequently, identifying and protecting routes between wetlands, and between wetlands and upland feeding and hibernation sites, is important.

Roads and increasing traffic volumes are significant contributors to amphibian population declines (Langton 1989, Fahrig et al. 1995). Enabling unhindered movement of animals with distinct migration routes across highways and roads through specially constructed tunnels and underpasses has decreased mortality and facilitated dispersal by many species (Langton 1989). Unhindered migration in fragmented habitats also reduces homozygosity resulting from reproductive isolation. Populations surrounded by roads exhibit higher levels of genetic distancing from reduced gene flow than those connected by woods and other favorable corridors (Reh 1989). Metal culverts, tunnels and fences, however, may confuse amphibians using magnetoperception (e.g., *B. bufo*) for orientation and should be avoided (Sinsch 1989).

For species that do not exhibit clear migration or dispersal routes, undeveloped flood zones, riparian strips and other shrublands and forests adjacent to rivers, streams and creeks are ideal corridors. These provide cool, moist microclimate conducive to amphibian transit (Hurlbert 1969, Kleeberger and Werner 1983, Sinsch 1988a) and may already receive regulatory protection. Moreover, floodplains and riparian habitats may deter upland predators normally not making regular use of wet habitats (Forman 1983). From riparian corridors, forested draws and other vegetated depressions may direct amphibians upslope to feeding and refuge habitat.

Although dispersal distances in amphibians may exceed 1,000 m (Table 1); individuals of most taxa remain within 1,000 m of wetlands. Hence, corridors between wetlands, or between suitable home-range habitats, do not need to exceed this distance.

Corridor widths should provide a core area of large diameter logs and other coarse wood to provide refuge habitat - particularly cover to protect from desiccation during summer droughts (Dupuis, et al. 1995), cover to minimize predation, and cover to limit potential agonistic behavior between species, sexes and age classes. Unfortunately, refuge requirements for minimizing predation, competition and agonistic behavior are not known.

A 30-m core corridor width is suggested to take competition, predation and other potential density dependent regulating mechanisms into consideration. To maintain soil moisture, retain cool microclimates, and provide woody debris for amphibians establishing temporary refuges during migration and dispersal, buffers should flank the corridor. Moreover, to protect vegetation in the core from the effects of wind and evapotranspiration, buffer widths of 2-3 times the height of adjacent vegetation are suggested as a rule-of-thumb in high-density stands (Fritschen and Edmonds 1976) and 5 times the vegetation height in low-density stands (Reifsnyder 1955). Buffer widths of 60-120 m to maintain soil moisture and ground temperature, 120-180 m to stabilize air temperature and humidity, and 240 m to negate vegetation responses have been identified to insulate old-growth Douglas-fir (*Pseudotsuga menziesii*) forests from the climatic effects of clear-cut edges (Chen et al. 1990, Chen et al. 1992). Consequently, to protect amphibians in corridors from changes in soil moisture and air temperatures, total corridor widths of 150 to 270 and 270 to 390 m respectively are required. To retain intact and self-sustaining vegetation in the core a total corridor width of 510 m would be required.

These widths minimize structural and functional changes to a core but may not be essential to migrating and dispersing amphibians. These animals will move through sub-optimum vegetation and climatic conditions of narrower corridors not protected by wide buffers. Moreover, depth-of-edge microclimatic influences are also dependent on vegetation beyond the buffer's edge. Large shrubs, and trees adjacent to the buffer's outer edge can reduce buffer-width requirements (Wales 1972, Ranney et al. 1981, Forman 1995) without disturbing climatic conditions within the core. As a compromise, corridor widths of 150 m are suggested with reduced widths proportional to the insulative properties of vegetation adjacent to core's buffer. Not surprisingly, meadow and grass communities are poor habitats, exposing amphibians to greater climatic extremes and predators and should not be used as corridors or count as protective buffers to corridors.

Watershed Conditions to Protect and Maintain Restoration Wetlands

To provide the full range of biological functions of consequence to amphibians the restoration wetland should be located within a watershed basin in which imperviousness (the sum of roofs, sidewalks, parking lots and roads) does not exceed 10 to 15 percent (Schueler 1994, Richter and Azous 1996), unless specific methods are used to minimize hydrological and water-quality impacts on wetlands. In depressional wetlands, greater imperviousness leads to elevated discharge and increases wetland water level fluctuations. In slope wetlands, more runoff from impervious surfaces increases current velocities. Both diminish suitable amphibian breeding, feeding and rearing habitat (see hydrology section). Moreover, wetland invertebrates and plants also decrease in richness and abundance with greater water-level fluctuations and concomitant pollution loads (Schueler 1994, Ludwa 1994, Hicks 1995), further reducing amphibian habitat quality.

Wetland Design Criteria

Breeding Habitat

Hydrodynamics and Hydrology. Hydrodynamic and hydrologic modifiers of importance to amphibians include current velocity and the duration, depth and frequency of flooding. Collectively, these factors directly influence spawning and egg survivorship, and indirectly affect breeding success by determining vegetation zonation and the outcome of competition and predator-prey relationships.

Current Velocity. Lentic breeding amphibians spawn only in vernal ponds, depressional wetlands, or in slow-moving or quiescent water of riverine backwaters and slope wetlands (Savage 1961, Nussbaum et al. 1983, Blaustein et al. 1995a, Olson and Leonard 1997). Increased discharge to riverine and slope wetlands can increase current velocity preventing breeding, reducing the success of fertilization, dislodging eggs from oviposition sites, or physically damaging eggs with suspended silt, sediment and large floating debris (Lind et al. 1996, Richter personal observation). Velocities exceeding 5 cm/second preclude breeding by both *R. aurora* and *A. gracile* (Richter and Roughgarden unpublished manuscript), suggesting that currents through mitigation wetlands should not exceed this threshold. Flow regimes with current velocities approaching 2 cm/second may be optimum in that slight currents are beneficial in flushing silt and suspended solids, and in delivering oxygenated water to eggs. Modest currents resist freezing, thereby providing open water for early spring spawning. However currents can also reduce temperatures, prolonging time to hatching and metamorphosis.

Duration of Water. All native lentic-breeding Pacific Northwest amphibians use permanently flooded wetlands. Successive years of inundation are essential only to *A. gracile,* some *A. macrodactylum* populations that have more than a 1-year larval period, neotenic *T. granulosa*, and high-elevation species that overwinter in wetlands. Semi-permanence is beneficial to many species because it precludes the establishment of fish, (e.g., Salmonidae), amphibian (e.g., *R. catesbeiana, A. gracile*) and invertebrate (e.g., crayfish, dragonflies) predators. Consequently, semi-permanent marshes and swamps make ideal mitigation wetlands for most Puget Sound Basin species, whereas permanent wetlands may be targeted for *A. gracile. Rana catesbeiana* and *A. gracile* may be a threat to other amphibians in small and structurally simple permanent wetlands. Competitive exclusion between these predators and other amphibians has yet to be established in large, structurally complex wetlands (Richter and Azous 1995, Richter personal observation). Several closely situated wetlands with differing hydroperiods, or one two-celled wetland in which one cell is vernal and the other remains permanently flooded, would provide breeding and spawning habitat for the full compliment of native species.

Water Depth. Pacific Northwest amphibians exhibit distinct minimum and maximum water depths beyond which oviposition normally does not occur (Table 2). From these and other surveys (Cooke 1975, Seale 1982, Waldman 1982) it can be inferred that oviposition of most temperate amphibian species is at depths between 10 and 100 cm. Under exceptional circumstances, spawn of most species may be found in very shallow (e.g., Licht 1969) water such as flooded ruts and roadside ditches (Richter personal observation) or very deep in lakes (e.g., 200-300 cm below the surface [Crisafulli personal communication]), presumably because oviposition sites at preferred water depths are unavailable. In these unique situations however, successful egg and larval development through metamorphosis may be rare.

Breeding amphibians may select medium-depth water because of optimum temperatures that

Table 2. Oviposition Water Depths of Selected Pacific Northwest Amphibians

Species	Water Depth (cm)[a]	Depth Below Surface (cm)[a]	Location	References
SALAMANDERS				
NORTHWESTERN SALAMANDER *Ambystoma gracile*	[15-61] [0-122] ----- 200-300[b]	 (7) ----- 200-300[b]	Washington Washington ----- Washington	Slater 1936 Richter & Roughgarden unpublished manuscript Crisafulli personal communication
LONG-TOED SALAMANDER *Ambystoma macrodactylum*	[7-22] [3-39] (11)	12-65 tracks substrate[c]	California Washington	Anderson 1967 Richter & Leonard personal observation.
FROGS				
RED-LEGGED FROG *Rana aurora*	[30-152] [61-122] [26-47] (28) ----- -----	max 152 tracks substrate[c] ----- [15-38]	British Columbia Washington ----- Oregon	Licht 1969 Richter & Roughgarden unpublished manuscript Storm 1960
CASCADES FROG *Rana cascadae*	[5-18]	substrate but touching surface	Oregon	Sype 1974
BULLFROG (INTRODUCED) *Rana catesbeiana*	[42-54]	surface to subsurface	Washington	Richter personal observation
SOUTHERN LEOPARD FROG *Rana sphenocephala*	(32) & (40)	(4) & (15)	South Carolina	Caldwell 1986
PACIFIC TREEFROG *Hyla regilla*	[5-45] (20)		Washington	Richter personal observation
TOADS				
WESTERN TOAD *Bufo boreas*	[5-100]		Oregon	Olson 1988

[a] [Range], (mean).
[b] Personal observation.
[c] Eggs typically a fixed distance above the bottom.

accelerate egg development (Duellman and Trueb 1986), and possibly because such depths are unavailable to some predators. Medium-depth water may also indicate that wetlands are only semi-permanently flooded, thereby eliminating the threat of predation, yet retaining sufficient water through metamorphosis.

To encourage spawning by Puget Sound Basin amphibians, mitigation wetlands with depths of 10-50 cm should be available from December through May. Terraced shores of 10-20, 30-40 and 50-60 cm depths, or alternately, a gradual sideslope gradient of 10-horizontal to 1-vertical or shallower slope, to a total depth of 50 cm, should provide spawning habitat for all species. Commonly used 3-vertical to 1-horizontal or steeper sideslopes provide little optimum preferred depth and should not be graded at proposed breeding sites.

Water Level Fluctuation. Water-level fluctuation, especially maxima and minima, are vitally important to wetland biota (Weller 1990). Extremes may directly harm amphibians or decrease habitat value through simplified floral communities (Azous and Richter 1995), reduced food abundance and variety (Munn and Brusven 1991, Ludwa 1994), and altered spatial relationships among physical elements (Reinelt and Horner 1995).

Many temperate latitude amphibians may minimize exposure of eggs to fluctuating depths and temperatures by spawning not only in mid-depth water but also by initially or permanently submerging eggs below the surface (Table 2).

Controlled experiments with several caudates reveal distinct and narrow oviposition preferences within the water column. In aquarium studies using simulated vegetation, Miaud (1995), for example, found that three species of *Triturus* spawn within 10 cm of the surface. My field experiment using wooden dowels as surrogate vegetation, (Richter and Roughgarden unpublished manuscript) show that every *A. gracile* breeding female during a given night attaches its eggs at almost identical depths (i.e., 7.06 cm ±1.27 cm , N=19) below the surface.

Amphibian egg development depends on permanent or partial submergence. Therefore, water levels should be stabilized from spawning through hatching, which for most Puget Sound Basin species is from mid-December through mid-May. Although mean water level fluctuations exceeding 20 cm have been correlated to decreased amphibian richness in wetlands (Richter and Azous 1995, Azous and Richter 1995), our experiments suggest that extended drops of more than 7 cm from oviposition through hatching may kill *A. gracile* eggs. Moreover, eggs of *A. macrodactylum* and *H. regilla* spawned in shallow water are harmed by stranding and desiccation on shore (Richter personal observation).

Water level declines may also leave eggs spawned within the water column closer to the surface, thereby increasing exposure to freezing and to greater temperature fluctuations. Finally, short-duration water fluctuations may not kill eggs outright but may leave eggs above the surface so that their weight pulls them from attachment sites and they sink to the substrate when water levels rise (e.g., *A. gracile*, early development stage of *R. aurora*) or float to the surface (e.g., late development stage of *R. aurora*) (Richter and Roughgarden unpublished manuscript). At the surface, currents and wind may route free-floating eggs to shallow water and other unfavorable environments, whereas on the substrate they may be prone to sedimentation and colder water where eggs develop slower or fail to develop at all (Richter and Roughgarden unpublished manuscript).

Water Quality. Amphibians are found in water of widely varying chemical composition (Cooke and Frazer 1976). Ildos and Ancona (1994) found chemical water quality in 42 wetlands well within the range of amphibians' tolerability. Richter (unpublished data) also found water chemistry not to be directly limiting amphibian distribution and spawning when comparing pH, conductivity, alkalinity, TSS, chlorophyll a, ammonia-nitrogen, nitrate+nitrite-nitrogen, soluble reactive phosphorus, total phosphorus P, fecal coliforms, enterococci, microtox, and metals (cadmium Cd, copper Cu, lead Pb, zinc Zn) in 19 wetlands (Reinelt and Horner 1990) against concentrations known to be harmful (Power et al. 1989). Within these same wetlands, however, Azous (1991), found a significant negative correlation between amphibian richness and water column conductivity. Moreover, Platin (1994) and Platin and Richter (1995) found *R. aurora* embryo mortality positively correlated to a principal water quality component comprising conductivity, Ca, Mg, and pH, and negatively correlated to a second principal component including total P, total suspended solids, Pb, Zn, Al, total organic content and dissolved oxygen. Interestingly, *A. gracile* egg mortality under similar conditions showed no correlation with these two principal components but instead correlated with total petroleum hydrocarbons and fecal coliforms. These and

other studies (Strijbosch 1979, Beattie and Tyler-Jones 1992, Rowe et al. 1992, Sadinski and Dunson 1992, Rowe and Dunson 1993, 1995) suggest that some species' distribution and breeding success may be predicted locally by water quality, most notably conductivity, pH, Al, total cations, NO_2, chemical oxygen demand and dissolved organic carbon.

Because heavy metals and other pollutants adhere to suspended solids, removing sediments before they enter mitigation wetlands may protect water quality. The settling rate of suspended solids is a function of vegetation and current velocity, which in turn is determined by slope. Consequently, moderately vegetated or totally submerged hydrophytes at emergent wetland gradients of roughly 0.0012 to 0.0006 will provide slope conditions to remove depositional sediments in 15- 61 cm depth waters of 2.4- to 6.1-m-wide channels (Adamus et al. 1987).

Methods to reduce suspended solids and pollutants at mitigation wetlands can also include capturing runoff in detention facilities and wet ponds, or by channeling water through biofiltration swales prior to wetland discharge (Horner 1988). Swales should be less than 10 cm deep or have current velocities exceeding 5 cm/s to deter breeding amphibians from placing their eggs in danger. Short, 50-cm-high fences are also suggested around stormwater ponds with widely fluctuating water depths to deter access by native amphibians and around facilities with stable water to prevent colonization by introduced *R. catesbeiana* (Richter personal observation).

Other than outright death from toxic spills and sediment deposition, direct relationships between water quality, amphibian distribution, and egg survivorship remain complex, and may be a reason for the absence of water-quality criteria for amphibians (Boyer and Grue 1995). Consequently, until causality between water-quality factors and amphibian success are assessed under controlled conditions (both in isolation and in synergy with each other, and under diverse hydrological conditions) it may be prudent to meet or exceed water-quality standards similar to those suggested to protect human health by federal, state and local jurisdictions.

Vegetation. Amphibian presence at wetlands is correlated with vegetation that provides shelter, calling locations, spawning sites and feeding possibilities (Ildos and Ancona 1994). Vegetation classes, cover, structure and species may all significantly affect breeding and larval success. Pacific Northwest caudates attach their eggs directly to vegetation within the water column (Slater 1936, Anderson 1967, Richter and Roughgarden unpublished manuscript, Richter and Leonard, unpublished data), whereas anurans anchor eggs to vegetation either below or near the surface (e.g., *R. aurora, B. boreas*), or occasionally spawn free-floating eggs (*R. pretiosa*; Licht 1969).

Vegetation Classes. Aquatic-bed and emergent vegetation classes are significant in accounting for amphibian richness and breeding at wetlands, whereas the total number of habitat classes per Cowardin et al. (1979) is unimportant in the Puget Sound Basin (Richter and Azous 1995). Vegetation type is one of the most important characteristics determining the distribution of seven amphibians in Italy (Ildos and Ancona 1994). Strijbosch (1979) found egg masses of all nine species surveyed occurred disproportionately within particular hydrophyte communities in the Netherlands. Interestingly, Strijbosch found a clear correlation between certain hydrophyte communities and the chemical/physical water-quality data. This confounded direct vegetational relationships but highlighted the complex interactions of site criteria that guide amphibian habitat use.

Cover and Interspersion. Although not well quantified, vegetation cover may be one of the most important factors determining amphibian breeding distributions. Most amphibians generally avoid both open water and densely vegetated sites, instead selecting habitats with an interspersion of both features (Strijbosch 1979, Ildos and Ancona 1994, Richter personal observation). Comparisons of vegetation cover within ¼ -m^2 circular plots surrounding *A. gracile, Rana aurora* and *R. catesbeiana* eggs to cover within ¼ -m^2 randomly chosen sites, suggests that both dense (95-100 percent) and light (0-5 percent) cover is avoided (Richter personal observation). This may be to reduce predation in exposed or highly vegetated areas. In contrast, some species may exhibit different preferences in different parts of their range. For example, *B. bufo* was the only species among nine amphibians spawning in open water in the Netherlands (Strijbosch 1979), but was found spawning among vegetation in England (Beebee 1983).

Nevertheless, these findings suggest that most species may prefer interspersed open water and vegetation for oviposition. Therefore, a 50:50, 25:75, or 75:25 ratio of open water to vegetation is suggested to encourage spawning.

Plant Species. Amphibians oviposit on a variety of plants, although within particular wetlands only certain species may be used. Among the caudates, Miaud (1995) found the warty newt (*Triturus cristatus*) attaching its eggs entirely on one plant species (e.g., *Nasturtium*) whereas eggs of the palmate newt (*T. helveticus helveticus* Razoumowski) were found on four species. Strijbosch (1979) found *T. cristatus*, the alpine newt (*T. alpestris* Laurenti), and the smooth newt (*T. vulgaris* L.), ovipositing primarily on reeds (*Phragmites* spp.). Richter and Roughgarden (unpublished manuscript), however, found eggs of *A. gracile* attached to a minimum of 28 species of herbs, shrubs and trees in 11 wetlands. Richter and Leonard (unpublished data) found *A. macrodactylum* spawning on minimum of eight plant species in just one wetland and Richter (unpublished data) found its eggs on at least four different plant species at a second wetland.

Anura also use a diversity of species for oviposition. Licht (1969) found *R. aurora* eggs attached to submerged stalks of *Typha*, *Carex* and *Potamogeton*, but Richter and Roughgarden (unpublished manuscript) rarely found *R. aurora* using *Typha* spp., despite its abundance. Rather, they observed wide use of less common *Oenanthe*, *Polygonum*, *Juncus*, *Scirpus*, and *Carex*. From these surveys and studies it can be inferred that plant species per se is unimportant for oviposition. This frees mitigation ecologists to select any plant species that grows under the specific hydrologic conditions required by breeding amphibians.

Species Structure. Experimental evidence suggests that vegetation structure, particularly plant shape and stem diameter, are the oviposition criteria most important to caudates. The role of vegetation structure in anuran oviposition remains undetermined. In laboratory tests a linear shape is selected over an arborescent shape by ovipositing *T. helveticus*, *T. cristatus* and *T. alpestris* (Miaud 1995). Moreover, the thinnest PVC tubing (i.e., simulated vegetation) of 25 μ was preferred by the smallest species, *T. helveticus*, whereas medium tubing of 50 μ was preferred over thicker tubing of 100 μ by *T. alpestris* and *T. cristatus*.

Wetland surveys and controlled field studies of several Pacific Northwest salamanders also confirm that distinct stem widths are preferred by ovipositing caudates. Field surveys of *A. gracile* egg masses indicated that the stem diameters of the 28 plant species used ranged between 1 and 8 mm, and for all species averaged 3 to 4 mm. Moreover, under controlled field studies using 3, 6, and 9 mm wooden dowels as surrogate hydrophytes, an overwhelming preference for 3-mm dowels over 6-mm diameter dowels was found, with 9 mm dowels almost totally avoided (Richter and Roughgarden unpublished). Interestingly, the smallest caudate in the Puget Sound Basin, *A. macrodactylum*, oviposits primarily on very thin (i.e., 1 to 2 mm diameter vegetation [Richter unpublished data]) supporting a direct relationship between salamander size and oviposition substrate size.

It is clear that Pacific Northwest amphibians use the most commonly available thinstem hydrophytes and small twigs and branches available in benign hydrodynamic and hydrologic spawning conditions. Stems need to be grasped readily by females, hold eggs firmly, and allow eggs to float within the water column to fluctuating depths and currents. A partial list of recommended shallow-marsh hydrophytes used by Pacific Northwest amphibians is provided in Table 3.

Hardhack, (*Spiraea douglasii*), willow (*Salix* spp.) and other shrubs and trees with rigid woody stems are not prescribed. Finally, mitigation should not use aggressive smartweeds (*Polygonum* spp.) and cattail (*Typha latifolia*). Reed canarygrass (*Phalaris arundinacea*), yellow flag (*Iris pseudacorus*), loosestrife (*Lythrum salicaria*, *Lysimachia vulgaris*) and other exotics. If they are found at mitigation wetlands, they should be excluded through selective vegetation management.

Interim oviposition sites may be provided at newly restored and created wetlands by adding thin, dead branches and twigs from hardhack, red alder (*Alnus rubra*), western redcedar (*Thuja plicata*) western hemlock (*Tsuga heterophylla*) and other species at spawning locations.

Wetland Layout

Size. Given that all other habitat features are equal, wetland size is unrelated to amphibian richness (Sjögren 1991, Ildos and Ancona 1994, Richter and Azous 1995). Hence, there is no minimum area required by breeding amphibians (Cooke and Frazer 1976, Andren and Nilson 1985, Denton 1991, Richter and Azous 1995). Smaller wetlands may be used more than larger ones by some species (Cooke and Frazer 1976, Moler and Franz 1987) because larger, and consequently often permanent wetlands are suitable for predators such as fish, reptiles and amphibians requiring permanent water. *Ambystoma macrodactylum* may be especially abundant in small wetlands (e.g., 201 different individuals captured over 25 nights in a 130-m^2 pond; Leonard and

Table 3. Puget Sound Lowland Shallow Marsh Hydrophytes Used as Amphibian Oviposition Sites.

Amphibian Species	Additional Material Used	Scientific Name	Common Name
	1 to 2 mm Diameter, Thin-stemmed Emergents		
LONG-TOED SALAMANDER *Ambystoma macrodactylum*	Roothairs, rootlets & thin twigs.	POACAE *Alopecurus aequalis* *A. geniculatus*	GRASS FAMILY Shortawn foxtail Water foxtail
PACIFIC TREEFROG *Hyla regilla*	Some roothairs, rootlets & thin twigs.	*Agrostis aequivalvis* *Calamagrostis canadensis*	Alaska bentgrass Bluejoint reedgrass
OREGON SPOTTED FROG *Rana pretiosa*	Some variable-diameter emergents.	*Glyceria grandis* *G. elata* *G. occidentalis*	American mannagrass Tall mannagrass Western mannagrass
		CYPERACEAE *Carex obnupta* *Eleocharis palustris* *E. ovata* *E.* acicularis	SEDGEFAMILY Slough sedge Common spikerush Ovate spikerush Needle spikerush
		JUNCACEAE *Juncus acuminatus* *J. bufonius*	RUSH FAMILY Tapered rush Toad rush
	3 to 6 mm Diameter, Medium-stemmed Emergents		
NORTHWESTERN SALAMANDER *Ambystoma macrodactylum*		CYPERACEAE. *Carex athrostachya* *C. utriculata*	SEDGE FAMILY Slender-beaked sedge Beaked sedge
RED-LEGGED FROG *Rana aurora*	Some variable-diameter emergents.	CYPERACEAE *Carex* obnupta	RUSH FAMILY Slough sedge
		MENYANTHACEAE *Menyanthes trifoliata*	BUCK-BEAN FAMILY Buckbean
		APIACEAE *Oenanthe sarmentosa*	PARSLEY FAMILY Water-parsley
		POLYGONACEAE *Polygonum amphibium* *P. hydropiperoides* *P. punctatum*	BUCKWHEAT FAMILY Water smartweed Waterpepper Dotted smartweed
		POTAMOGETONACEAE *Potamogeton natans* *P. emersum* *P. gramineus*	PONDWEED FAMILY Floating-leaf pond weed Emersed pondweed Grass-leaved pond weed
		SPARGANIACEAE *Sparganium emersum* *S. eurycarpum*	BUR-REED FAMILY Simple-stem Bur-reed Broadfruit Bur-reed
	Variable Diameter Emergents		
ROUGH-SKINNED NEWT *Taricha granulosa*	Leafy submerged plants.	CYPERACEAE *Scirpus cyperinus* *S. microcarpus*	SEDGE FAMILY Woolgrass Small-fruited Bullrush
WESTERN TOAD *Bufo boreas*	Grasses & leafy grass-like plants.		

Richter unpublished data). Nevertheless, there may be instances where some species avoid very small pools (e.g., *Triturus* <100 m^2 [Cooke and Frazer 1976]). Other species with strong territorial males (e.g., introduced *R. catesbeiana*, and green frog, *R. clamitans* Latreille), may be less numerous in smaller than in larger wetlands.

Because water depth and seasonal availability, interspersion of open water and vegetation, and specific vegetation structure are all important breeding criteria, wetlands should be big enough for these attributes to coexist.

Configuration. Amphibians favor shorelines for oviposition because they spawn in specific vegetation at particular depths. Moreover, food and shelter for larvae is more frequently found along vegetated shorelines. Consequently, configuration (e.g., length of shoreline in relation to area) is an important characteristic that should maximize spawning, feeding and cover habitats. Additional shoreline habitat can be created by establishing long lobes and scalloped edges.

Orientation. Egg development is a function of water temperature (Herreid and Kinney 1967, Brown 1975, 1977). Clutch numbers increase with temperature (Albers and Prouty 1987, Sjögren et. al. 1988, Sjögren and Larsson 1988), with warmer northern shores exhibiting the highest numbers of eggs among spring-breeding species (Stenhouse 1985, Sinsch 1988b).

Given that all other habitat features are similar, Pacific Northwest amphibians (as well as those in other higher-latitude regions) show a clear spawning preference for northern quadrants of wetlands, although the importance of orientation decreases with wetland size (Richter and Roughgarden unpublished manuscript). In controlled field experiments where *A. gracile* could choose to oviposit eggs along both north and south shores and furthermore select between northwestern and northeastern shorelines, 98 percent of eggs were spawned along the north shore, and 68 percent of these were attached to sites along the northwestern shore (Richter and Roughgarden unpublished manuscript). Presumably the breeding preference for the northwestern wetland locations is a consequence of higher water temperatures from solar radiation.

Consequently, mitigation wetlands should provide a gradual shallow slope along the northern shore. Oval wetlands with a long axis from southwest to northeast will optimize preferred breeding habitat along the northwestern side of wetlands. Podloucky (1989) suggests 500 to 1,000 m^2 ponds with, east-west to north-south ratios of 2:1 to maximize sunny areas along the northern shore.

Optimizing water depth through wetland bathymetry (i.e., gradual side-slope with optimum water depth and vegetation plantings etc.) along the northern shore and increasing solar radiation through vegetation management can further raise springtime water temperatures.

Buffers. Buffers are essential components of wetlands (Wilard and Hiller 1990), providing separation between wetlands and developed environments, moderating water level fluctuations, and trapping sediments and insoluble pollutants (Castelle et al. 1992). In highly urbanized areas, wetland buffers also may be the only terrestrial habitat remaining for amphibians (Richter personal observation); adjacent upland feeding, aestivation and hibernation sites may be inaccessible or destroyed. In the Pacific Northwest, *R. pretiosa* and *R. cascadae* (Dumas 1966) remain within several meters of water throughout the year, whereas in other regions, the Columbia spotted frog *R. luetraventris*, *R. catesbeiana*, *R. clamitans* (Richter personal observations) and other Ranids, (Beebee 1996) remain at the wetland buffer throughout most of their lives.

Buffers may be especially important in providing cover to females and to metamorphs. Female *R. aurora*, *A. gracile* (Richter personal observation) and A. *macrodactylum* (Beneski et. al. 1986, Leonard and Richter unpublished data) generally wait in buffers near wetlands until environmental and biological conditions favor spawning. They then enter wetlands during one or a few nights to spawn, and quickly retreat to the cover of adjacent buffers or more distant vegetated upslope habitats.

Metamorphs of many species may also benefit from wetland buffers. They are important to the tiger salamander, *Ambystoma tigrinum* Green, seeking shelter in rodent burrows during the first days after emigrating from natal ponds (Loredo et al. 1996). Surveys of wetland buffers and upland habitats during summer droughts, suggests that metamorphs of *H. regilla*, *B. boreas*, *R. aurora*, and *T. granulosa* may spend several weeks in buffers and disperse to upland sites only after rains or when the soil and vegetation are moist (Richter personal observation). Vulnerable metamorphs and juveniles may have greater cover, moisture, and food within wetland buffers than in upland sites.

Guidance on buffer width for overall wetland protection is available (Brown and Schaefer 1987, Kusler and Kentula 1990, Castelle et al. 1992, Castelle at al.1994), although not based on

empirical studies of the unique microclimatic and other biological requirements of amphibians. Extrapolations from Rudolph and Dickson (1990) suggest that 30-95 m buffers retain the full complement of amphibians adjacent to streams. The Washington Department of Wildlife (1992) suggests 50 m buffers for *R. pretiosa* in western Washington and 30 m for *R. luteiventris* in eastern Washington, although empirical justification for either width is unavailable.

In the absence of experimental data it seems prudent to maintain a treed, riparian zone. A buffer equal to two to three tree heights circumscribing the mitigation wetland may be optimum for litter and downed wood provision and for sustaining a moderate microclimate for migrating breeding adults and newly metamorphosed juveniles. prior to terrestrial dispersal. The 60-240 m protective zone recommended for migration corridors (see previous section) can be applied to wetlands and may be wider than the widths based on tree heights. Whether a zone of this width is necessary to provide staging habitat for breeding and food and cover for metamorphs cannot be known until individual species requirements are established.

Discussion

Wetland-breeding amphibians exhibit complex life cycles that include ontogenetic changes in morphology, physiology and behaviors. Typically each stage includes carnivorous terrestrial adults, sedentary aquatic spawn, motile aquatic larvae and transitional metamorphs–all constrained by the distribution of suitable habitat. Wetland restoration and creation goals must consider habitats to provide for all life-stages in order to attract and sustain amphibians at wetlands. Restoration must focus first on optimizing a number of concurrent landscape elements that include the spatial arrangement between wetland breeding, upland feeding and refuge patches, and their interconnecting corridors through the landscape matrix. Restoration should then include wetland design features that optimize, oviposition and egg and larval survival.

Theoretical models for landscape structure of fragmented populations are available (Harris and Scheck 1991, Soulé and Gilpin 1991, Forman 1995). Explicit empirical landscape elements, however, are seldom considered in mitigation (Fahrig and Merriam 1985), although a few have been developed for birds (Saunders and Hobbs 1991), mammals (Saunders and Hobbs 1991) and amphibians (Sjögren-Gulve and Ray 1996). Whereas general qualitative habitat needs have been described for many amphibians, we know relatively little about the population biology of amphibians and how they use the landscape. Few if any empirically based quantitative data exist on landscape attributes, including the effective distances between wetlands, corridor dimensions or the required area and characteristics of intervening feeding and refuge habitats. Moreover, conservation biologists continue to debate the merits of small multiple habitat patches as opposed to single larger patches and the benefits of corridors (Mann and Plummer 1995). In the interim, I assume that creation and restoration connected to or near natural wetlands will increase success. Future studies must address such critical aspects of successful wetland restoration.

General habitat-use patterns are species specific, and may vary between geographic regions and years (Aubry and Brookes 1991 and references therein, Brown 1997). Foremost life history adaptations of Pacific Northwest lentic-breeding species indicate late winter and early spring migrations, explosive and synchronous breeding, strategic selection of oviposition sites, rapid egg and larval development, and an initiation of metamorphosis prior to wetland drying (Wilbur and Collins 1973; Brown 1975, 1977; Nussbaum et al. 1983; Blaustein et al. 1995a, Olson and Leonard 1997, Richter and Roughgarden unpublished manuscript). The literature review and field studies reported here also suggest that oviposition sites are the result of selective pressures enabling females to spawn successfully, and eggs to develop rapidly and safely under conditions of relatively distinct weather, wetland hydrology, water quality, vegetation and predation (see Collins 1975, Hairston 1987 and references therein). These findings have significant management implications for the restoration and creation of wetlands that may be applied to encourage amphibian reproduction.

I am confident the data and conservative suggestions within this paper can be applied to many wetland-breeding species because, as this paper suggests, specific landscape and breeding habitat needs of Pacific Northwest species (as well as other temperate latitude, lentic-breeding species) have more common features than differences. By providing habitat for the species exhibiting the widest habitat needs, species with more restricted needs can be accommodated. For example, all amphibians in the Pacific Northwest can successfully breed in permanent wetlands, but not all breed in shallow temporary wetlands. Moreover, by providing the hydrologic requirements for *A. gracile*, all other

species' hydrologic needs are also likely to be met. Similarly, by providing the upland needs for *B. boreas* and *R. aurora* the refuge habitat of most other amphibians may be met. Clearly, additional species-specific research will find amphibians under a wider range of habitat conditions (i.e., exhibiting wide tolerances) because of their "plasticity" and adaptability. Nevertheless, as an increasing number of controlled studies show, amphibians exhibit clear and sometimes narrow habitat preferences under which survival excels. It is for these conditions that restoration wetlands should be built. Studies that focus on the entire range of observations, such as amphibian homing rather than post-metamorphic dispersal, may greatly overestimate the potential for wetland colonization and population stability of a species. Hence, for wetland restoration I recommend species-specific optimum conditions whenever they have been identified. Suggested recommendations should further be adjusted to reflect local climatic conditions and the breeding and spawning habitat preferences of endemic populations.

The historical absence of carefully designed empirical methods for identifying movement corridors, refuge habitat and spawning requirements makes it difficult to differentiate discrete optimum conditions from those used merely because of availability. Clearly, the range of circumstances over which species are found and breed in nature may not reflect optimum environmental factors (see above). They may, in fact, differ from that observed under controlled experiments and represent conditions under which species may not maintain themselves. To determine optimum conditions, site selection and designs criteria can be selected with explicit quantitative characteristics in mind. These can then be monitored at the restoration wetland for success and thereby facilitate comparisons with natural conditions to improve amphibian restoration.

Suggestions that benefit amphibians favor other wildlife and vegetation. For example, breeding birds (Brown, and Dinsmore1986) and mammals (Kent 1994b) are more abundant when distance to adjacent wetlands is less and density of wetlands increases. Native plant species richness also increases with proximity to the nearest seed source (Reinartz and Warne 1993), with wetlands of significantly more species per unit area in locations where other wetlands are close (Møller and Rørdam 1985).

Finally, siting mitigation wetlands to benefit amphibians by the suggested criteria will preserve upland habitat, contribute undeveloped areas of open space, and provide places for passive recreation. Maintaining urbanizing areas in forest cover also increases stormwater infiltration, promoting groundwater recharge and flood control. These important landscape features, which are disappearing from watersheds under increasing development pressures, should be protected and maintained.

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Literature Cited

Adamus, P. R., E. J. Clairain, Jr., R. D. Smith, and R. E. Young. 1987. Wetland Evaluation Technique (WET), Volume II: Methodology. U.S. Army Corps of Engineers Waterways Experiment Station, Operational Draft Technical Report Y-87, Vicksburg, MS.

Albers, P. H. and R. M. Prouty. 1987. Survival of spotted salamander eggs in temporary woodland ponds of coastal Maryland. Environmental Pollution 46:45-61.

Anderson, J. D. 1967. A comparison of the life histories of coastal and montane populations of *Ambystoma macrodactylum* in California. The American Midland Naturalist 77:323-355.

Andren, C. and G. Nilson. 1985. Breeding pool characteristics and reproduction in an island population of natterjack toads, *Bufo calamita* Laur., at the Swedish west coast. Amphibia-Reptilia 6:137-142.

Aubry K. B. and M. H. Brookes (contributing editors). 1991. Wildlife and Vegetation of Unmanaged Douglas-fir forests. U.S. Forest Service, Pacific Northwest Research Station, Portland, OR. General Technical Report PNW-GTR-285.

Aubry, K. B, and P. A. Hall. 1991. Terrestrial amphibian communities in the southern Washington Coast Range. pp. 327-338 In K.B. Aubry and M.H. Brookes (contr. eds.) Wildlife and Vegetation of Unmanaged Douglas-fir forests. U.S. Forest Service, Pacific Northwest Research Station, Portland, OR. General Technical Report PNW-GTR-285.

Azous , A. 1991. An analysis of urbanization effects on wetland biological communities. M.S. Thesis, University of Washington, Seattle, WA.

Azous, A. and K. O. Richter. 1995. Amphibian and plant community responses to changing hydrology in urbanizing wetlands. pp. 156-162 In E. Robichaud (ed.) Puget Sound Research '95 Proceedings, Volume 1. Puget Sound Water Quality Authority, Olympia, WA.

Beattie, R. C. and R. Tyler-Jones. 1992. The effects of low pH and aluminum on breeding success in the frog *Rana temporaria*. Journal of Herpetology 26:353-360.

Beebee, T. C. J. 1983. The Natterjack Toad. Oxford University Press, Oxford, England.

Beebee, T. C. J. 1996. Ecology and Conservation of Amphibians. Chapman & Hall, London, England.

Bellis, E. D. 1959. A study of the movement of American toads in a Minnesota bog. Copeia 1959: 173-174.

Bellis, E. D. 1962. Cover value and escape habitats of the wood frog in a Minnesota bog. Herpetologica 17: 228-231.

Beneski, J. T. Jr., E.J. Zalisko and J. H. Larsen Jr. 1986. Demography and migratory patterns of the eastern long-toed salamander, *Ambystoma macrodactylum columbianum*. Copeia 1986:398-408.

Berven, K. A. and T. A. Grudzien. 1990. Dispersal in the wood frog (*Rana sylvcatica*): Implications for genetic population structure. Evolution 44:2047-2056.

Bishop, S. C. 1941. The salamanders of New York. NY State Mus. Bull. No. 324. NY.

Blaustein, A. R., B. Edmond, J. M. Kiesecker, J. J. Beatty, and D. G. Hokit. 1995b. Ambient ultraviolet radiation causes mortality in salamander eggs. Ecological Applications 5:740-743.

Blaustein, A. R., P. D. Hoffman, D. G. Hokit, J. M. Kiesecker, S. C. Walls, and J.B. Hays. 1994b. UV repair and resistance to solar UV-B in amphibian eggs: A link to population declines? Proceedings National Academy of Sciences 9:1791-1795.

Blaustein, A. R., D. G. Hokit, R. K. O'Hara, and R.A. Holt. 1994a Pathogenic fungus contributes to amphibian losses in the Pacific Northwest. Biological Conservation 67:251-254.

Blaustein, A. R., D. W. Wake, and W. P. Sousa. 1994c. Amphibian declines: Judging stability, persistence, and susceptibility of populations to local and global extinctions. Conservation Biology 8:60-71.

Blaustein, A. R., J. J. Beatty, D. H. Olson, and R. M. Storm. 1995a. The Biology of Amphibians and Reptiles in old-growth forests in the Pacific Northwest. U.S. Forest Service, Corvallis, OR. General Technical Report PNW-GTR-337.

Booth, D. B. 1991. Urbanization and the natural drainage system - impacts, solutions, and prognoses. Northwest Environmental Journal 7:93-118.

Booth, D. B. and C. R. Jackson. 1994. Urbanization of aquatic systems - degradation thresholds and limits of mitigation. pp. 425-434. In R.A. Martson and V. R. Hasfurther (eds.) Effects of Human-Induced Changes on Hydrologic Systems, Proceedings; Annual Summer Symposium of the American Water Resources Association, Jackson Hole, WY.

Booth, D. B. and L. E. Reinelt. 1993. Consequences of urbanization on aquatic systems - measured effects, degradation thresholds, and corrective strategies. pp. 545-550 In Watershed '93, A National Conference on Watershed Management, Alexandria, VA.

Boyer, R. 1993. Evaluation of water quality in relation to frogs at Klamath Basin National Wildlife Refuges. M.S. Thesis. University of Washington, Seattle, WA.

Boyer, R. and C. E. Grue. 1995. The need for water quality criteria for frogs. Environmental Health Perspectives 103:352-357.

Bradford, D. F. 1991. Mass mortality and extinction in a high-elevation population of *Rana muscosa*. Journal of Herpetology 25:174-177.

Brattstrom, B. H. and J. W. Warren. 1955. Observations of the ecology and behavior of the Pacific Treefrog, *Hyla regilla*. Copeia 1955:181-191.

Breden, F. 1987. The effect of post-metamorphic dispersal on the population genetic structure of Fowler's toad, *Bufo woodhousei fowleri*. Copeia 1987:386-395.

Brown, C. 1997. Habitat structure and occupancy patterns of the montane frog. *Rana cascadae*, in the Cascade Range, Oregon, at multiple scales: Implications for population dynamics in patchy environments. M.S. Thesis, Oregon State University, Corvallis, OR.

Brown, H. A. 1975. Embryonic temperature adaptations of the Pacific treefrog, *Hyla regilla*. Comp. Biochem. and Physiol. 51A:863-873.

Brown, H. A. 1977. Temperature and embryonic development of the northwestern salamander, *Ambystoma gracile*. pp. 209-217 In R.D. Andrews III, R.L. Carr, F. Gibson, B.Z. Long, R.A. Soltero and K.C. Svedberg. Proceedings of the Symposium on Terrestrial and Aquatic Ecological Studies of the Northwest. Eastern Washington State College Press, Cheney, WA.

Brown, M. T. and J. M. Schaefer. 1987. Buffer Zones for Water, Wetlands, and Wildlife: A Final Report on the Evaluation of the Applicability of Upland Buffers for the Wetlands of the Wekiva Basin. Center for Wetlands, University of Florida, Gainsville, FL.

Brown, M. and J. J. Dinsmore 1986. Implication of marsh size and isolation for marsh management. J. Wildl. Manage. 50:392-397.

Caldwell, J. P. 1986. Selection of egg deposition sites: A seasonal shift in the Southern Leopard Frog. Copeia 1986: 249-253.

Castelle, A. J., C. Conolly, M. Emers, E. D. Metz, S. Meyer, M. Witter, S. Mauermann, T. Erickson, and S. S. Cooke. 1992. Wetland Buffers: Use and Effectiveness. Adolfson Associates, Inc., Shorelands and Coastal Zone Management Program, Washington Department of Ecology. Olympia, WA. Pub. No. 92-10.

Castelle, A. J., A. W. Johnson, and C. Conolly. 1994. Wetland and stream buffer size requirements-a review. Journ. Environmental Quality. 23:878-882.

Chen, J., J. F. Franklin, and T. A. Spies. 1990. Microclimatic pattern and biological response at the clear-cut edges of old-growth Douglas-fir stands. Northwest Environmental Journal 6:424-425.

Chen, J., J. F. Franklin, and T. A. Spies. 1992. Vegetation responses to edge environments in old-growth Douglas-fir forests. Ecological Applications 2:387-396.

Christein, D. and D. H. Taylor 1978. Population dynamics in breeding aggregations of the American Toad, *Bufo americanus* (Amphibia, Anura, Bufonidae) Journal of Herpetology 12:17-24.

Collins, J. P. 1975. A comparative study of the life history strategies in a community of frogs. Ph. D. Dissertation, University of Michigan, Ann Arbor, MI.

Cooke, A. S. 1975. Spawn site selection and colony size of the frog (*Rana temporaria*) and the toad (*Bufo bufo*). Journal of Zoology (London) 175:29-38.

Cooke, A. S. and J. F. D. Frazer. 1976. Characteristics of newt breeding sites. Journal of Zoology (London) 178:223-236.

Corn, P. S. 1994. What we know and don't know about amphibian declines in the west. pp. 59-67. In W. W. Covington and L. F. DeBano (tech. coords.) Sustainable Ecological Systems: Implementing an Ecological Approach to Land Management, U.S. Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. General Technical Report RM-247.

Corn, P. S. and R. B. Bury. 1991. Terrestrial amphibian communities in the Oregon Coast Range pp. 305-317 In K. B. Aubry and M.H. Brookes (contr. eds.) Wildlife and Vegetation of Unmanaged Douglas-fir forests. U.S. Forest Service, Pacific Northwest Research Station, Portland, OR. General Technical Report PNW-GTR-285.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. Classification of Wetlands and Deepwater Habitats of the United States. U.S. Fish and Wildlife Service, Washington DC. FWS/OBS-79/31.

Delis, P. R., H. R. Mushinsky, and E. D. McCoy. 1996. Decline of some west-central Florida anuran populations in response to habitat degradation. Biodiversity and Conservation 5:1579-1595.

Denton, J. S. 1991. The distribution and breeding sites of newts in Cumbria, England. Herpetological Journal 1:549-554.

Denton, J. S. and T. J. C. Beebee. 1993. Density-related features of natterjack toad (*Bufo calamita*) populations in Britain. Journal of Zoology (London) 229:105-119.

Dingle, H., 1996. Migration: The Biology of Life On the Move. Oxford University Press, Oxford, England.

Dodd, C. K. Jr. and B. G. Charest. 1988. The herpetofaunal community of temporary ponds in north Florida sandhills: Species composition, temporal use, and management implications. pp. 11-22 In R. C. Szaro, K. E. Severson, and D. R. Patton (tech. coords.) Management of Amphibians, Reptiles, and Small Mammals in North America. Proceedings of a Symposium Flagstaff, AZ. U. S. Forest Service General Technical Report RM-166.

Dole, J. W. 1965. Spatial relations in natural populations of the Leopard Frog, Rana pipiens Schreber, in northern Michigan. Am. Mid Nat. 74:464-478.

Douglas, M. E. and B. L. Monroe Jr. 1981. A comparative study of topographical orientation in *Ambystoma* (Amphibia: Caudata). Copeia 1981:460-463.

Duellman, W.E. and L. Trueb. 1986. Biology of Amphibians. McGraw-Hill, Inc., New York, NY.

Dumas, P. C. 1966. Studies of the *Rana* species complex in the Pacific Northwest. Copeia 1966:60-74.

Dupuis, L. A., J. N. M. Smith, and F. Bunnell. 1995. Relation of terrestrial-breeding amphibian abundance to tree-stand age. Cons. Biol. 9:645-653.

Ewert, M. A. 1969. Seasonal movements of the toads *Bufo americanus* and *Bufo cognatus* in Northwestern Minnesota. Ph.D. Dissertation. University of Minnesota, Minneapolis, MN.

Fahrig, L., J. H. Pedlar, S. E. Pope, P. D. Taylor, and S. F. Wegner. 1995. Effect of road traffic on amphibian density. Biol. Cons. 73:177-182.

Fahrig, L. and G. Merriam. 1985. Habitat patch connectivity and population survival. Ecology: 66:1762-1768.

Forman, R. T. T. 1983 Corridors in landscapes: Their ecological structure and function. Ecology (CSSR)2:375-387.

Forman, R. T. T. 1995. Land Mosaics: The Ecology of Landscapes and Regions. Cambridge University Press, New York, NY.

Fritschen, L. J. and R. Edmonds. 1976. Dispersion of florescent particles into and within a Douglas-fir forest. pp 280-301 In R. J. Engelmann and G. A. Sehmel (eds.) Atmosphere-Surface exchange of particulate and gaseous pollutants. ERDA Symposium Series-38. NTIS CONF-74-0921.

Galatowitsch, S. M. and A. van der Valk. 1994. Restoring Prairie Wetlands: An Ecological Approach. Iowa State University Press, Ames, IA.

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

Gill, D. E. 1978a. Effective population size and interdemic migration rates in a metapopulation of the red-spotted newt, *Notopthalmus viridescens* (Rafinesque). Evolution 32:839-849.

Gill, D. E. 1978b. The metapopulation ecology of the red-spotted newt, *Notopthalmus viridescens* (Rafinesque). Ecological Monographs 48:145-166.

Gittins, S. P., A. G. Parker, and F. M. Slater 1980. Population characteristics of the common toad (*Bufo bufo*) visiting a breeding site in Mid-Wales. Journal of Animal Ecology 49:161-173.

Hammer, D. A. 1992. Creating Freshwater Wetlands. Lewis Publishers, Chelsea. MN.

Hairston, N. G. 1987. Community Ecology and Salamander Guilds. Cambridge University Press, Cambridge, MA.

Harris, L. D. and J. Scheck. 1991. From implications to applications: The dispersal corridor principle applied to the conservation of biological diversity. pp. 189-220 In D. A. Saunders and R. J. Hobbs (eds.) Nature Conservation 2: The Role of Corridors. Surrey Beatty & Sons Pty Limited, Australia.

Hayes, M. P. and M. R. Jennings. 1986. Decline of ranid frog species in western north America: Are bullfrogs (*Rana catesbeiana*) responsible? Journal of Herpetology 20:490-509.

Herreid, C. F. and S. Kinney. 1967. Temperature and devilment of the wood frog, *Rana sylvatica*, in Alaska. Ecology 48:579-590.

Heusser, H. 1960. Über die beziehungen der Erdkröte (*Bufo bufo* L.) zu ihrem Laichplatz II. Behavior 16:93-109.

Heusser, H. 1968. Die Lebensweise der Erdkröte, *Bufo bufo* L.: Wanderungen und Sommerquartiere. Rev. Suis. Zool. 76:444-517.

Hicks, A. 1995. Impervious surface area and benthic macroinvertebrate response as an index of impact from urbanization on freshwater wetlands. M.S. Thesis, University of Massachusetts, Amherst, MA.

Holland, C. C., J. Honea, S. E. Gwin, and M. E. Kentula. 1995. Wetland degradation and loss in the rapidly urbanizing area of Portland, OR. Wetlands 15:336-345.

Horner, R. R. 1988. Biofiltration Systems for Storm Runoff Water Quality Control. Prepared for Washington State Department of Ecology, Municipality of Metropolitan Seattle, King County,

and Cities of Bellevue, Mountlake Terrace and Redmond, WA.

Hurlbert, S. H. 1969. The breeding migrations and interhabitat wandering of the vermilion-spotted newt *Notopthalmus viridescens* (Rafinesque). Ecological Monographs 39:465-488.

Hutchison, V. H. and R. K. Dupre'. 1992. Thermoregulation. pp. 206-249 In M. E. Feder and W. W. Burggren (eds.) Environmental Physiology of Amphibians. The University of Chicago Press, Chicago, IL.

Ildos, A. S. and N. Ancona. 1994. Analysis of amphibian habitat preferences in a farmland area (Po Plain, northern Italy. Amphibia-Reptilia 15:307-316.

Jameson, D. L. 1957. Population structure and homing responses in the Pacific tree frog. Copeia 1957:221-228.

Kent, D. M. 1994a. Designing wetlands for wildlife. pp. 283-306 In Donald M Kent (eds.) Applied Wetlands Science and Technology. CRC Press, Inc., Boca Raton, FL.

Kent, D. M. 1994b. Managing wetlands for wildlife. pp. 307-329 In Donald M Kent (eds.) Applied Wetlands Science and Technology. CRC Press, Inc., Boca Raton, FL.

Kleeberger, S. R. and J. K. Werner. 1983. Post-breeding migration and summer movement of *Ambystoma opacum*. Journal of Herpetology 17:176-177.

Kusler, J. A. and G. Brooks (eds.). 1988. Proceedings of the National Wetland Symposium: Wetland Hydrology. Association of State Wetland Managers, Inc., Berne, NY.

Kusler, J. A. and M. E. Kentula (eds.) 1990. Wetland Creation and Restoration: The Status of the Science. Island Press, Washington, D.C.

Langton, T. S. E., ed. 1989. Amphibians and Roads. ACO Polymer Products Ltd. Shefford, Bedfordshire, England.

Lehmkuhl, J. F., L. F. Ruggiero and P. A. Hall. 1991. Landscape-scale patterns of forest fragmentation and wildlife richness and abundance in the southern Washington Cascade range. pp. 425-442 In K. B. Aubry and M. H. Brookes (contributing editors) Wildlife and Vegetation of Unmanaged Douglas-fir forests. U. S. Forest Service, Pacific Northwest Research Station, Portland, OR. General Technical Report PNW-GTR-285.

Licht, L. E. 1969. Comparative breeding behavior of the red-legged frog (*Rana aurora aurora*) and the western spotted frog (*Rana pretiosa pretiosa*) in southwestern British Columbia. Canadian Journal of Zoology 47:1287-1299.

Licht, L. E. 1971. Breeding habits and embryonic thermal requirements of the frog, *Rana pretiosa pretiosa*, in the pacific northwest. Ecology 52:116-124.

Lind, A. J., H. H. Welsh Jr., and R. A. Wilson. 1996. The effect of a dam on breeding habitat and egg survival of the foothill yellow-legged frog (*Rana boylii*) in northwestern California. Herpetological Review 27:62-67.

Loredo, I., D. VanVuren, and M. L. Morrison. 1996. Habitat use and migration behavior of the California tiger salamander. Jour. Herp. 30:282-285.

Ludwa, K. A. 1994. Urbanization effects on palustrine wetlands: empirical water quality models and development of a macroinvertebrate community-based biological index. M. S. Thesis, University of Washington, Seattle, WA.

Mann, C. C. and M. L. Plummer. 1995. Are wildlife corridors the right path? Science 270:1428-1430.

McAllister, K. R. and B. P. Leonard. 1990. Past Distribution and Current Status of the Spotted Frog (*Rana pretiosa*) in Western Washington. 1989 Progress Report. Washington State Department of Wildlife, Olympia, WA.

McAllister, K. R. and B. P. Leonard. 1997. Washington State Status Report for the Oregon Spotted Frog. Washington Department of Fish and Wildlife, Olympia, WA.

Miaud, C. 1995. Oviposition site selection in three species of European newts (Salamandridae) genus *Triturus*. Amphibia-Reptilia 16:265-272.

Minton, S. A., Jr. 1968. The fate of amphibians and reptiles in a suburban area. Journal of Herpetology 2:113-116.

Mitsch, W. J. and J. G. Gosselink. 1993. Wetlands. 2nd Edition. Van Nostrand Reinhold Company, Inc., New York, NY.

Moler, P. E. and R. Franz. 1987. Wildlife values of small, isolated wetlands in the southeastern coastal plain. pp. 234-241 In R. R. Odom, K. A. Riddleberger, and J. C. Ozier (eds.) Georgia Department of Natural Resources, Atlanta, GA.

Møller, T. R. and C. P. Rørdam 1985. Species numbers of vascular plants in relation to area, isolation and age of ponds in Denmark. Oikos 45:8-16.

Munn, M. D. and M. A. Brusven. 1991. Benthic macroinvertebrate communities in nonregulated and regulated waters of the Clearwater River, ID. Regulated Rivers: Research & Management 6:1-11.

Nussbaum, R. A., D. B. Edmond Jr., and R. M. Storm. 1983. Amphibians and Reptiles of the Pacific Northwest. University of Idaho Press, Moscow, ID.

Oldham, R. S. 1966. Spring movements in the American Toad, *Bufo americanus*. Can. J. Zool. 44:63-100.

Olson, D. H. 1988. The ecological and behavioral dynamics of breeding in three sympatric anuran amphibians. Ph.D. Dissertation. Oregon State University.

Olson, D. H. and W. P. Leonard. 1997. Amphibian inventory and monitoring: A standardized approach for the Pacific Northwest. pp. 1-21 In D. H. Olson, W. P. Leonard and R. B. Bury (eds.). Sampling Amphibians in Lentic Habitats. Northwest Fauna 4, Olympia, WA.

Orchard, S. A. 1992. Amphibian population declines in British Columbia. pp. 10-13 In C. A. Bishop and K. E. Pettit (eds.) Declines in Canadian Amphibian Populations: Designing a National Monitoring Strategy. Occasional Paper 76. Canadian Fish and Wildlife Service. Ottawa, Canada.

Payne, N. F. 1992. Techniques for Wildlife Habitat Management of Wetlands. Biological Resource Management Series, McGraw-Hill, New York, NY.

Pechmann, J. H. K. and H. M. Wilbur. 1994. Putting declining amphibian populations in perspective: Natural fluctuations and human impacts. Herpetologica 50:65-84.

Platin, T. J. 1994. Wetland amphibians *Ambystoma gracile* and *Rana aurora* as bioindicators of stress associated with watershed urbanization. M.S. Thesis, University of Washington, Seattle, WA.

Platin, T. J. and K. O. Richter. 1995. Amphibians as bioindicators of stress associated with watershed development. pp. 163-170 In E. Robichaud (ed.) Puget Sound Research '95 Proceedings, Volume 1, Puget Sound Water Quality Authority, Olympia, WA.

Podloucky, R. 1989. Protection of amphibians on roads- examples and experiences from Lower Saxony. pp. 15-27 In T. S. E. Langton (ed.) Amphibians and Roads. ACO Polymer Products Ltd. Shefford, Bedfordshire, England.

Power, T., K. L. Clark, A. Harfenist, and D. B. Peakall. 1989. A Review and Evaluation of the Amphibian Toxicological Literature. Technical Report No. 61, Canadian Wildlife Service, Headquarters, Ottawa, Canada.

Pounds, J. A. and M. L. Crump. 1994. Amphibian declines and climate disturbance: the case of the golden toad and the harlequin frog. Conservation Biology 8:72-85.

Ranney, J. W., M. C. Bruner, and J. B. Levenson. 1981. The importance of edge in the structure and dynamics of forest islands. pp. 67-95 In R.L. Burgess and D. M. Sharpe (eds.) Forest Island Dynamics in Man-Dominated Landscapes. Springer-Verlag, New York, NY.

Reh, W. 1989. Investigations into the influences of roads on the genetic structure of populations of the common frog *Rana temporaria*. pp. 101-103 In T.E.S. Langton (ed.) Amphibians and Roads. ACO Polymer Products Ltd., Shefford, Bedfordshire, England.

Reifsnyder, W. E. 1955. Wind profiles in a small isolated forest stand. Forest Science: 1:289-297.

Reinarts, J. A. and E. L. Warne. 1993. Development of vegetation in small created wetlands in southeastern Wisconsin. Wetlands 13:153-164.

Reinelt, L. E. and R. R. Horner. 1990. Characteristics of the hydrology and water quality of palustrine wetlands affected by urban stormwater. King County Resource Planning, Seattle, WA.

Reinelt, L. E. and R. R. Horner. 1995. Pollutant removal from stormwater runoff by palustrine wetlands based on comprehensive budgets. Ecological Engineering 4:77-79.

Richter, K. O. and A. L. Azous. 1995. Amphibian occurrence and wetland characteristics in the Puget Sound Basin. Wetlands 15:305-312.

Richter, K. O. and A. L. Azous. 1996. Amphibian distribution, abundance and habitat use. pp. 5.1-5.11 In Wetlands and Urbanization, Implications for the Future. Puget Sound Wetlands and Stormwater Management Research Program, King County, Seattle, WA.

Rowe, C. L. and W. A. Dunson. 1993. Relationships among abiotic parameters and breeding effort by

three amphibians in temporary wetlands of central Pennsylvania. Wetlands 13:237-246.

Rowe, C. L. and W. A. Dunson. 1995. Impacts of hydroperiod on growth and survival of larval amphibians in temporary ponds of Central Pennsylvania. Oecologia 102:397-403.

Rowe, C. L., W. J. Sadinski, and W. A. Dunson. 1992. Effects of acute and chronic acidification on three larval amphibians that breed in temporary ponds. Archives of Environmental Contamination and Toxicology 23: 339-350.

Rudolph, D. C. and J. G. Dickson. 1990. Streamside zone width and amphibian and reptile abundance. The Southwestern Naturalist 35:472-476.

Sadinski, W. J and W. A. Dunson. 1992. A multilevel study of the effects of low pH on amphibians of temporary ponds. Journal of Herpetology 26:413-422.

Saunders, D. A., G. W. Arnold, A. A. Burbridge and A. J. M. Hopkins. 1987. Nature Conservation: The Role of Remnants of Native Vegetation. Surrey Beatty & Sons Pty Ltd. NSW, Australia.

Saunders, D. A. and R. J. Hobbs (eds.). 1991. Nature Conservation 2: The Role of Corridors. Surrey Beatty & Sons Pty Limited, NSW, Australia.

Savage, R. M. 1961. The Ecology and Life History of the Common Frog. Pitman, London, England.

Schueler, T. 1994. The importance of imperviousness. Watershed Protection Techniques 1:100-111.

Seale, D. 1982. Physical factors influencing oviposition by the woodfrog, *Rana sylvatica*, in Pennsylvania. Copeia 1982:627-635.

Semlitsch, R. D. 1981. Terrestrial activity and summer home range of the mole salamander (*Ambystoma talpoideum*). Canadian Journal of Zoology 59:315-322.

Shoemaker, V, H., S. S. Hilman, S. D. Hillyard, D. C. Jackson, L. L. McClanahan, P. P. Withers, and M. L. Wygoda. 1992. Exchange of water, ions, and respiratory gases in terrestrial amphibians. pp. 125-150 In M. E. Feder and W. W. Burggren (eds.) Environmental Physiology of Amphibians. The University of Chicago Press, Chicago, IL.

Shoop, C. R. 1965. Orientation of *Ambystoma maculatum*: Movements to and from breeding ponds. Science 149:558-559.

Shoop, C. R. 1968. Migratory orientation of *Ambystoma maculatum*: Movements near breeding ponds and displacements of migrating individuals. Biological Bulletin 135:230-238.

Shoop, C. R. and T. L. Doty. 1972. Migratory orientation by marbled salamanders (*Ambystoma opacum*) near a breeding area. Behavioral Biology 7:131-136.

Sinsch U. 1988a. Seasonal changes in the migratory behavior of the toad *Bufo bufo*: direction and magnitude of movements. Oecologia 76:390-398.

Sinsch U. 1988b. Temporal spacing of breeding activity in the natterjack toad *Bufo calamita*. Oecologia 76:399-407.

Sinsch, U. 1989. Migratory behaviour of the common toad *Bufo bufo* and the natterjack toad *Bufo calamita*. pp. 113-124 In Amphibians and Roads T.E.S. Langton (ed.) ACO Polymer Products Ltd. Shefford, Bedfordshire, England.

Sinsch, U. 1992. Structure and dynamic of a natterjack toad metapopulation (*Bufo calamita*). Oecologia 90:489-499.

Sinsch U. and D. Seidle. 1995. Dynamics of local and temporal breeding assemblages in a *Bufo calamita* metapopulation. Australian Journal of Ecology 20:351-361.

Sjögren, G. P. 1991. Extinction and isolation gradients in metapopulations: The case of the pool frog (*Rana lessonae*). Biological Journal of the Linnean Society (London) 42:135-147.

Sjögren, G. P. 1994. Distribution and extinction patterns within a northern metapopulation of the pool frog *Rana lessonae*. Ecology 75:1357-1367.

Sjögren, P. and K. Larsson. 1988. Thermal requirements for breeding in *Rana lessonae*, *R. arvalis* and *R. temporaria* at 60^0 N. Ph.D. Thesis Uppsala University, Sweden.

Sjögren, G. P., J. Elmberg, and S. A. Berglind. 1988. Thermal preference in the pool frog *Rana lessonae*: impact on the reproductive behavior of a northern fringe population. Holarctic Ecology. 11:178-184.

Sjögren-Gulve, P. and C. Ray 1996. Using logistic regression to model metapopulation dynamics: Large-scale forestry extirpates the Pool Frog. pp. 111-137 In Metapopulations and Wildlife Conservation D. R. McCullough (ed.) Island Press, Washington, D.C.

Slater, J. R. 1936. Notes on *Ambystoma gracile* Baird and *Ambystoma macrodactylum* Baird. Copeia 1936:234-236.

Soulé, M. E. and M. E. Gilpin. 1991. The theory of wildlife corridor capability. pp. 3-8 In D. A. Saunders and R. J. Hobbs (eds.) Nature Conservation 2: The Role of Corridors. Surrey Beatty & Sons Pty Limited, Australia.

Spies, T. A. and S. P. Cline. 1988. Coarse woody debris in forests and plantations of coastal Oregon pp. 5-24 In C. Maser, R. F. Tarrant, J. M. Trappe, and J. F. Franklin (tech. eds.) From The Forest to the Sea: A Story of Fallen Trees. U.S. Forest Service, Portland, OR. Gen. Tech. Rep. PNW-GTR-229.

Stebbins, R. C. and N. W. Cohen. 1995. A Natural History of Amphibians. Princeton University Press, Princeton, NJ.

Stenhouse, S. J. 1985. Migratory orientation and homing in *Ambystoma maculatum* and *Ambystoma opacum*. Copeia 1985:631-637.

Storm, R. M. 1960. Notes on the breeding biology of the Red-Legged Frog (*Rana aurora aurora*). Herpetologica 16:251-259.

Strijbosch, H. 1979. Habitat selection of amphibians during their aquatic phase. Oikos 33:363-372.

Sype, W. E. 1974. Breeding habitats, embryonic thermal requirements and embryonic and larval development of the Cascade frog, *Rana cascadae* Slater. Ph.D. Thesis. Oregon State University, Corvallis, OR.

Taylor, B. L. 1993. The influence of wetland and watershed morphological characteristics on wetland hydrology and relationships to wetland vegetation communities. M.S. Thesis University of Washington, Seattle, WA.

Travis, J. 1994. Calibrating our expectations in studying amphibian populations. Herpetologica 50:104-108.

Tunner, H. G. 1991. Migratory behaviour of *Rana lessonae* and its hybridogenetic associate *Rana esculenta* of Lake Neusiedl (Austria, Hungary). Proc. 6th Ord. Gen. Meeting Eur. Soc. Herp. Budapest, p. 91.

Twitty, V. C. 1958. Experiments on homing behavior and speciation in *Taricha*. pp. 415-458 In Vertebrate Speciation. ed. W.F. Blair. University of Texas Press, Austin, TX.

Twitty, V. C. 1959. Migration and speciation in newts. Science 130:1735-1743.

Twitty, V. C. 1966. Of Scientists and Salamanders. W.H. Freeman and Co., San Francisco, CA.

Wacasey, J. W. 1961. An ecological study of two sympatric species of salamanders, *Ambystoma maculatum* and *Ambystoma jeffersonianum* in southern Michigan. Ph.D. Dissertation, Michigan State University, East Lansing, MI.

Waldman, B. 1982. Adaptive significance of communal oviposition in wood frogs (*Rana sylvatica*). Behavioral Ecology and Sociobiology 10:169-174.

Wales, B. A. 1972. Vegetation analysis of northern and southern edges in a mature oak-hickory forest. Ecological Monographs 42:451-471.

Walls, S. C., A. R. Blaustein, and J. J. Beatty. 1992. Amphibian biodiversity of the Pacific Northwest with special reference to old-growth stands. Northwest Environmental Journal 8:53-69.

Ward, D. E., G. J. Hirons, and M. J. Self. 1995. Planning for the restoration of peat wetlands for birds. pp. 207-222 In B. D Wheeler, S. C. Shaw, W. J. Fojt, and R. Al. Robertson (eds.) Restoration of Temperate Wetlands. John Wiley & Sons, New York, NY.

Washington State Department of Wildlife. 1992. Buffer Needs of Wetland Wildlife: Final Draft. State of Washington Department of Wildlife, Habitat Management Division, Olympia, WA.

Weller, M. W. 1990. Waterfowl management techniques for wetland enhancement, restoration and creation useful in mitigation procedures. pp. 517-528 In J.A. Kusler and M. E. Kentula (eds.) Wetland Creation and Restoration: The Status of the Science. Island Press, Washington, DC.

Welsh, H. H. Jr. and A. J. Lind. 1988. Old growth forests and the distribution of the terrestrial herpetofauna. pp. 439-458 In R. C. Szaro R. C. Szaro, K. E. Severson, and D. R. Patton (tech. coords.) Management of Amphibians, Reptiles, and Small Mammals in North America. Proceedings of a Symposium Flagstaff, AZ. U.S. Forest Service Gen Tech Rep RM-166.

Wheeler, B. D., W. J. Fojt, and R. R. Allan. (eds.) 1995. Restoration of Temperate Wetlands. John Wiley & Sons, New York, NY.

Wilard, D. E. and A. K. Hiller. 1990. Wetland dynamics: Considerations for restored and created wetlands. pp. 459-466 In J.A. Kusler and M. E.

Kentula (eds.) Wetland Creation and Restoration: The Status of the Science. Island Press, Washington, DC.

Wilbur, H. M. and J. P. Collins. 1973. Ecological aspects of amphibian metamorphosis. Science 182:1305-1314.

Williams, P. K. 1970. Seasonal movements and population dynamics of four sympatric mole salamanders, Genus *Ambystoma*. Ph.D. Dissertation, Indiana University, Bloomington, IN.

Wolfenbarger, D. O. 1949. Dispersion of small organisms. American Midland Naturalist 35:1-152.

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Mitigating Impacts from Ferry Terminals on Eelgrass *(Zostera marina* L.)

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Abstract: **The purposes of this study were to understand effects of ferry terminals and ferry operations on eelgrass *(Zostera marina* L.) habitat in Puget Sound and to design appropriate measures to avoid, minimize and compensate for associated impacts. Dramatic increases in population and ferry traffic in western Washington have resulted in the need to expand existing terminals. Our studies showed that eelgrass habitats near ferry terminals were affected by light reduction, and other initial and long-term disturbances associated with terminal construction and maintenance, propeller wash, and bioturbation by macroinvertebrates (i.e., seastars and Dungeness crab). Measures to avoid and minimize impacts are being evaluated, including concrete piles, incorporating transparent plastic materials into the dock structure, reflective material under the dock, grating over offloading ramps, and reorienting slips to minimize propeller wash. Restoration is also proposed for portions of habitats previously destroyed by ferry operations.**

Introduction

Background and Purpose

Eelgrass *(Zostera marina* L.) habitats occur along an estimated 50 percent of the shoreline of Puget Sound (Thom and Hallum 1991). This ecosystem is rich in fish and invertebrate species, is important to migratory waterfowl (Baldwin and Lovvorn 1994) and functions as habitats for reproduction, rearing and feeding of commercially and recreationally important fisheries species (Phillips 1984). Although difficult to quantify, losses of eelgrass have occurred in Puget Sound because of physical alterations and development of shorelines (Thom and Hallum 1991). Docks, which necessarily cross over the eelgrass habitats, pose a threat to eelgrass through reduction of light and a variety of other factors. Dock shading has been shown to affect seagrasses in Washington State (Pentilla and Doty, 1990) and New England (Burdick and Short 1995). Because commerce and public transportation requires docks, and because trade and use of waterways for transportation is growing, the demand to expand existing docks as well as construct new docks will continue to impact eelgrass as well as other coastal habitats. For sustainable development to occur sound, scientifically-based methods to mitigate impacts of docks on shallow coastal habitats need to be developed.

The purpose of our study was to understand the impacts of ferry terminals on eelgrass habitats in Puget Sound, and develop measures to mitigate the impacts. Ferries provide a major mode of transportation for the public and commerce. In this paper we present our approach to addressing impacts of ferry terminals on eelgrass, and a conceptual model that identifies key impacts of ferry terminals on eelgrass in Puget Sound and describes how these impacts can be avoided, minimized and compensated. This model is based on impacts to eelgrass and fisheries resources observed at one ferry terminal (Clinton) that is slated for expansion, and on directed studies of light, currents and propeller wash. Studies described herein have also been completed at the Port Townsend and Edmonds terminals in Puget Sound.

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Methods

The investigations consisted of eelgrass habitat mapping, sampling of standing stock, in situ irradiance, fish and invertebrate observations, quantification of propeller wash impacts, and testing of light enhancement technologies.

The Clinton Ferry Terminal

The Clinton Ferry Terminal is located on the southeast end of Whidbey Island, Washington (Figure 1). The present creosote wood terminal is approximately 30 m wide and extends over the depth distribution of eelgrass. Vessels in excess of 70 m in length serve the terminal. During the day, dockings and sailings occur on the order of every 15 to 30 minutes.

Video Surveys

Eelgrass within approximately 500 m of either side of the terminal was mapped in July 1994 using an video camera towed from a boat. Video recordings of the bottom were made while the boat traveled along 26 transects that crisscrossed the study area over the maximum depth range eelgrass would be expected to occur. A Global Positioning System (GPS) mounted on board the boat automatically transferred exact station position to the videotape. These tapes were viewed to score eelgrass cover at approximately 2-m intervals along these transects according to the following cover classes: absent (0 percent), low-moderate (1-50 percent), and moderate to high (50-100 percent). These data were input to a Geographic Information System (GIS) database using GPS data and mapped. In addition, a vertical color aerial photograph, taken showing the general outline of eelgrass, was used to delineate the broad eelgrass distribution.

Eelgrass Standing Stock

Based on the video-generated maps, we established three transects through the eelgrass habitat. The transects ran parallel to the shoreline and were positioned at the inner, mid and outer portions of the habitat. The continuous, 180-m long transects spanned from 50 m north of the terminal to 50 m south of the terminal, and included the 30 m section under the terminal. On 29-30 August 1994, using SCUBA, we surveyed points at 5-m intervals along the each transect; recording depth, time, eelgrass shoot density (in 0.25 m^2 quadrants) and substrata type.

Irradiance

To document the effect of the dock on light during summer, mid-day conditions, we recorded photosynthetically active radiation (PAR) in air using a 4π quantum sensor (LICOR) at 1-m intervals along a transect which ran from 30 m north to 30 m south of the terminal. We also recorded PAR at each of the points where eelgrass was sampled along the transects during midday (1200-1400 hrs). In addition, we recorded PAR in the middle of the habitat and in air using sensors mounted in the habitat and on the dock, respectively. These measurements were taken at the same time as the eelgrass sampling.

Fish and Invertebrate Observations

Fish and macroinvertebrates present within each of the eelgrass quadrats were noted along with those in the general vicinity of the transect lines and the terminal.

Propeller Wash

Our observations indicated that propeller wash affected at least the outer portion of the eelgrass habitat. To examine this further, we recorded current speeds within 30 cm of the bottom during several ferry dockings and departures 20 September and 3 October 1994, and on 8 June 1995. We used a pressure-driven meter during the 1994 visits and a propeller-driven meter in 1995. The current meters were positioned immediately next to the terminal, on the south side, in a location where eelgrass was apparently absent due to sediment disturbances from propeller wash. We also made observations of currents at other positions further from the area obviously impacted by propeller wash. During the 3 October trip, we recorded irradiance concurrently with current speeds using the LICOR sensor positioned next to the current meter. Finally, we made observations and marked on maps of the study area the maximum extent of the visible surface plumes during several ferry departures and dockings.

Light Enhancement Technologies

Because light would be reduced by the expanded terminal, we investigated simple light-enhancement technologies including artificial (halogen lamp) lighting, concrete blocks with transparent glass centers (approximately 200 cm^2), and placement of reflective material. To evaluate whether quartz halogen

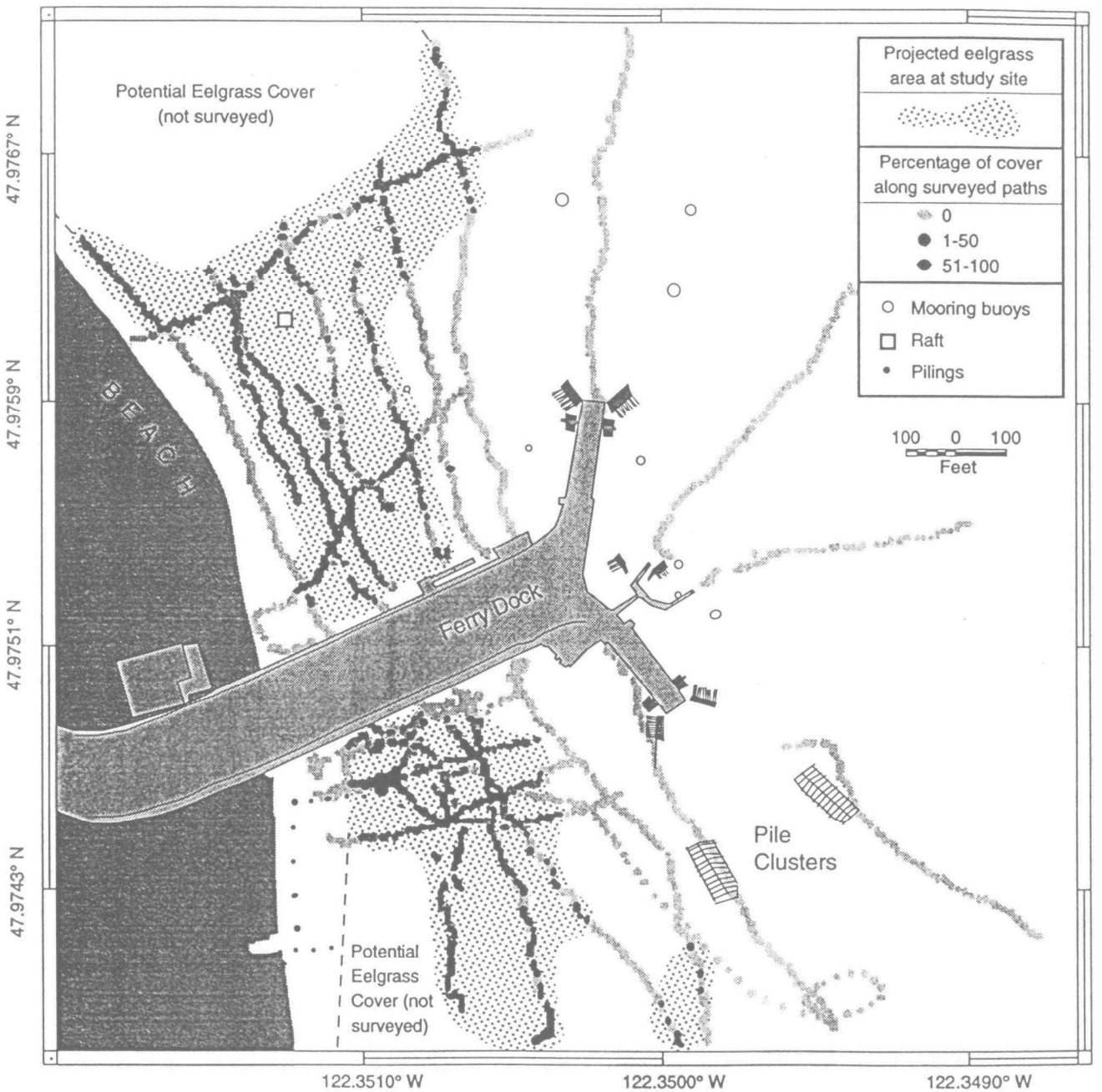

Figure 1. Clinton Terminal Showing Existing Eelgrass Distribution.

lamps could support photosynthesis, using oxygen flux methods we measured photosynthetic rates of three 10-cm long leaf sections placed in glass jars (1L) containing ambient seawater. Ten replicate jars were placed in tanks with flowing seawater at ambient sea temperature, and the tank was covered to block out ambient light. Two quartz halogen lamps (500 watts each) were positioned over the jars in the tank about 50 cm above the jars. The jars were submerged in approximately 40 cm of water. A separate set of 10 jars were incubated in ambient sunlight in a flowing seawater tank. Two blank jars (i.e., containing water but no seagrass) were incubated along with each treatment. Incubations were 3 hrs long. The experiment was run on 29 March 1995 and repeated on 30 March 1995.

To evaluate the amount of PAR transmitted through the concrete blocks containing glass centers, the PAR sensor was placed immediately below the lower edge of the block. Black plastic sheeting was used to exclude all light except that passing through the glass insert. Ambient PAR was concurrently recorded within 1 m of the block frequently throughout the day.

We evaluated the effect of reflective material on increasing light under docks by attaching a 122 x 244 cm sheet of plywood covered with aluminum foil to the underside of the wooden boat dock at Battelle's Marine Science Laboratory. The sheet was approximately in the center of the 12-m wide dock. Ambient PAR was measured on the south side of the dock, and under the dock at a point approximately 1 m below the foil and below an adjacent area of the dock. Light reaching each of these points was primarily light reflected off of the water surface.

Results

Habitat Characteristics

The ferry terminal will be expanded to the south of the existing terminal (Figure 2). The habitats that occur in this area include moderately sloping coarse sand from above extreme higher high water (EHHW) down to about mean lower low water (MLLW). Thereafter, the slope decreases and the substrata consists primarily of medium to fine sands (Figure 1). The terminal is situated along a portion of the beach that is somewhat steeper than areas within at least 1 km north and south of the terminal.

Vegetation (eelgrass and seaweeds)

Aquatic vegetation in the terminal expansion area consisted primarily of eelgrass, benthic macroalgae *(Ulva* spp., *Laminaria* sp.) and benthic diatoms (Table 1). *Ulva* was intermingled with the eelgrass, and *Laminaria* occurred primarily at depths greater than -3 m. Both seaweeds were very patchy in their distribution.

The distribution of eelgrass is illustrated in Figures 1 and 2. Eelgrass was primarily distributed from MLLW down to -3.6 m MLLW. Within this depth range, areas with little or no eelgrass occurred in a band approximately 5m wide on either side of the terminal. Shoot density ranged from 20 to over 800 m^{-2} within the main distribution of the habitat. The apparent zone affected by propeller wash at the outer edge of the habitat was also devoid of eelgrass.

Irradiance

Although light decreased dramatically under the terminal, it remained relatively high for a distance of 5 m under the south side of the terminal, and was also relatively high at 2 m under the north side of the terminal (Figure 3). Bottom light measurements made at midday during a sunny period in August show the dramatic decrease in PAR with depth, and that eelgrass occurred in greatest densities where PAR was greatest (Figure 4). Areas under the dock are distinguishable in Figure 4 as points with very low light relative to depth.

Fish and Invertebrate Observations

Twelve fish and 17 macroinvertebrate taxa were observed during the diving studies (Table 1). Although the eelgrass habitat held more species from each group (14 species of macroinvertebrates, 12 species of fish), the under-dock habitat did contain substantial numbers of species. Eelgrass-associated species such as tubesnout and piling-associated species such as bay mussel were restricted to their preferred habitats.

We observed high seastar *(Pycnopodia helianthodes)* densities (i.e., up to 15 m^{-2}) in some areas under or immediately adjacent to the dock. Seastars commonly feed on barnacles and mussels growing attached to pilings. Extensive piles of mussel shell and barnacle tests on the bottom under the dock were evidence that this was occurring at Clinton. Although

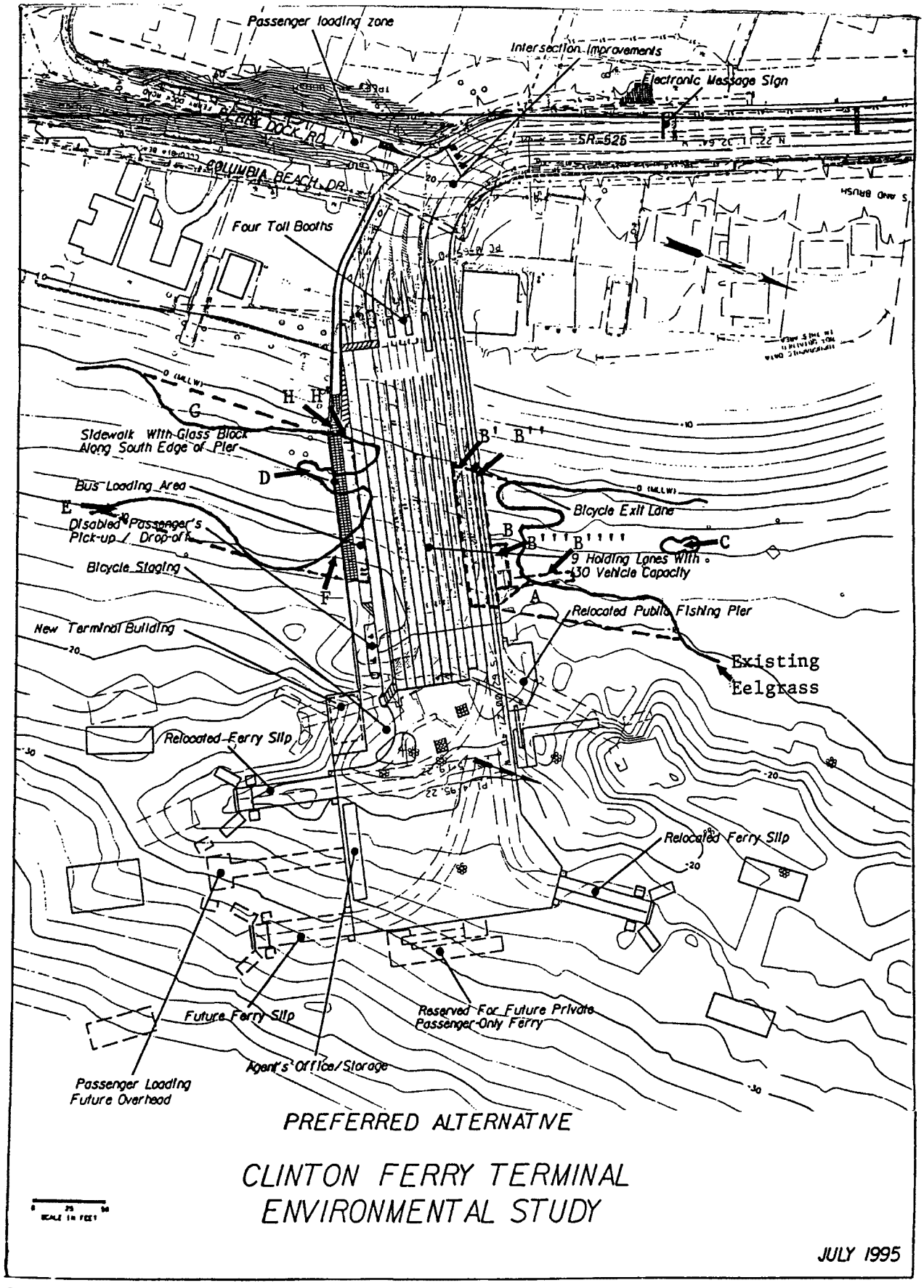

Figure 2. Preferred Construction Plan Showing Existing Eelgrass and Transplanting Areas (A-H).

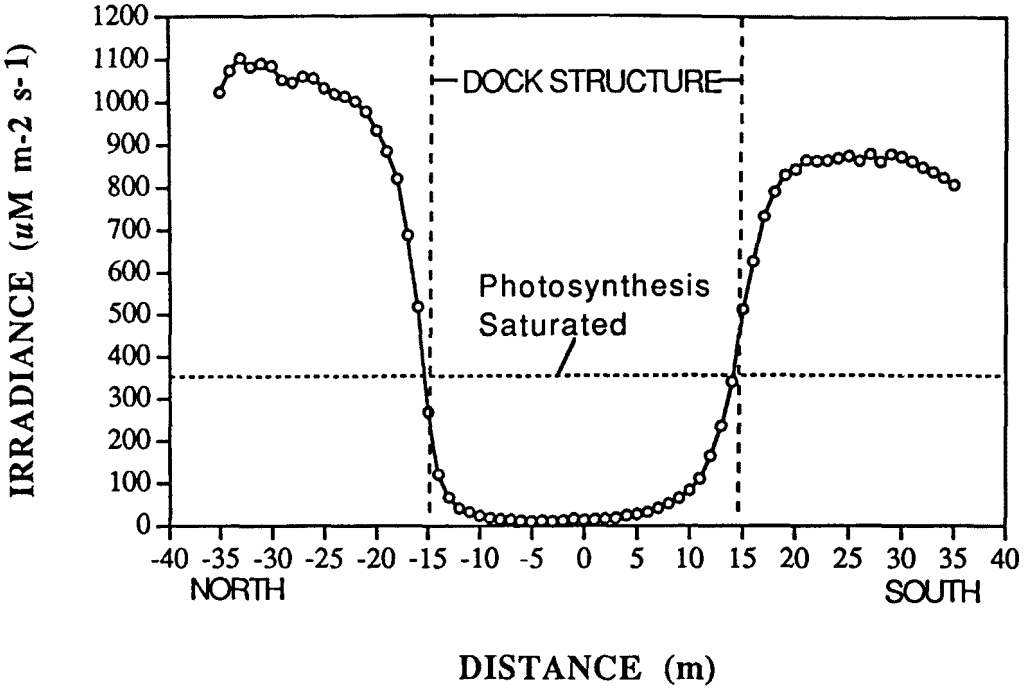

Figure 3. PAR in Air Under the Dock and Next to the Dock.

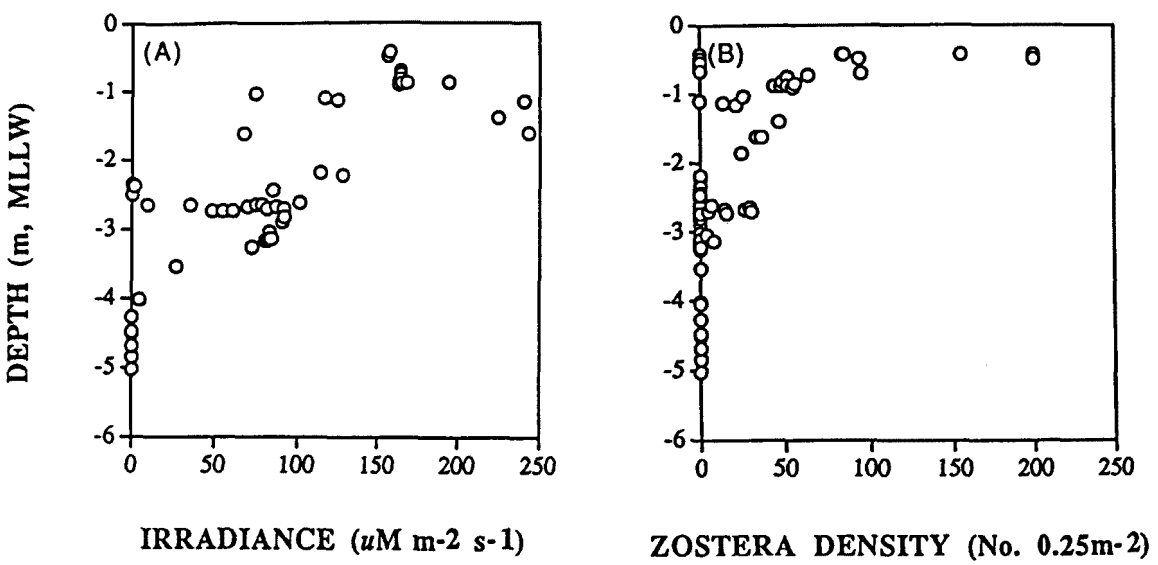

Figure 4. Data from Clinton Terminal showing variation with Depth of (A) Light and (B) Eelgrass Density.

Table 1. Taxa of fish, macroinvertebrates and macrophytes observed at the Clinton Terminal in late summer 1994.

COMMON NAME	SCIENTIFIC NAME	IN OR NEAR EELGRASS	UNDER DOCK
Invertebrates			
barnacle	*Balanus spp.*		•
bay mussel	*Mytilus spp.*		•
brooding anemone	*Epiactus prolifera*	•	
coon-striped shrimp	*Pandalus danae*	•	•
Dungeness crab	*Cancer magister*	•	•
heart cockle	*Clinocardium nuttallii*	•	•
helmet crab	*Telmessus cheiragonus*	•	
horse clam	*Tresus capax*	•	•
kelp crab	*Pugettia producta*	•	•
leather star	*Dermasterias imbricata*	•	•
nudibranch	*Melibe leonina*	•	•
nudibranch	*Dirona aurantia*	•	•
plumose anemone	*Metridium senile*		•
purple star	*Pisaster ochraceus*	•	•
red rock crab	*Cancer productus*	•	
snail (chink shell)	*Lacuna spp.*	•	
sunflower star	*Pycnopodia helianthodes*	•	•
Benthic Plants			
diatoms	unidentified species	•	
eelgrass	*Zostera marina*	•	
gracilaria	*Gracilaria pacifica*	•	
sea lettuce	*Ulva spp.*	•	
sugar wrack	*Laminaria saccarina*	•	
Vertebrates			
copper rockfish	*Sebastes caurinus*	•	
crescent gunnel	*Pholis laeta*	•	•
flatfish	various unidentified species	•	•
kelp greenling	*Hexagrammos decagrammus*	•	
penpoint gunnel	*Apodichthys flavidus*	•	
pile perch	*Rhacochilus vacca*	•	•
saddleback gunnel	*Pholis ornata*	•	•
sanddab	*Citharichthys spp.*	•	•
sculpins	various unidentified species	•	•
shiner perch	*Cymatogaster aggregata*	•	•
striped perch	*Embiotocca lateralis*	•	•
tubesnout	*Aulorhynchus flavidus*	•	

unquantified, Dungeness crab were at least as dense as seastars in many areas near the terminal.

Propeller Wash

Measurements at Clinton showed that the plume created during ferry arrival and departure extended up to 80m from the dockside end of the vessel. There was considerable variability in plume extent and angle depending on currents, wave action, wind and other factors. However, the angle of the main axis of the plume typically extended slightly shoreward of the main axis of the docking vessel. Plume events occurred on average every 15-20 minutes during the day, and lasted for 5-10 minutes. Hence, there are only relatively short periods of time when the plume is not present.

Materials in the observable plume consisted of fine sediments, organic debris and air bubbles. Sediment disturbance appeared to be greatest during periods of low tides, especially during spring tide series in the lunar cycle. At this time the propeller of the vessel is closest to the channel bottom of the slip. Bubbles were always present during plume events. Propeller wash accelerated current speeds from near zero to over 1 m s^{-1} (Figure 5). Maximum rates recorded during a docking were 3.5 m s^{-1}. Accelerations were accompanied by reductions in light (Figure 6).

Light Enhancement Experiments

The quartz halogen lamps supported photosynthesis of the eelgrass leaf sections. Mean productivity under the lights and ambient light were 2.08 and 0.74 mgO_2/g dry wt/hr, respectively. Calculations of the light attenuation caused by water and distance between lights attached to the underside of docks (at about 6 m MLLW) and the depth where eelgrass normally grows (0- to -3 m MLLW) at this site indicated that a prohibitively large number of lights (and expenditure of energy) would be needed to support eelgrass photosynthesis under the docks.

Approximately 60 percent of the ambient PAR was transmitted through the concrete block containing a glass center. The foil attached under the dock reflected substantially more PAR than did the wooden dock (Figure 7).

Discussion

Interpretation of Field Studies

Shading by the dock is undoubtedly a major factor causing the loss of eelgrass near the Clinton terminal, but we believe that propeller wash, bioturbation and other physical disturbances may be contributing to the loss. The irradiance measurements clearly showed that light reaches very low levels under the terminal. However, lack of eelgrass in a 5-m wide band around the terminal suggests that other factors are active. Terminals of this vintage (mid-1950's) were constructed by hydraulically inserting wood piles into the sediment. This process, which involves water jet liquefaction the sediment, completely eliminates eelgrass and likely drastically modifies sediment conditions such as organic content and redox profile. Eelgrass, which primarily spreads by rhizome growth in the region, may take decades to recover from this types of disturbance. Annual maintenance of wood terminals is required and these activities (e.g., barge grounding and anchoring, propeller scars from tugs and work boats) may also disturb eelgrass.

During eelgrass surveys, the divers noted seastars foraging extensively for bivalves at the edge of the eelgrass habitat adjacent to the dock. Enhanced populations of both seastars and crab may have ramifications on eelgrass in the vicinity of the dock. This behavior has been documented by us in other regions of Puget Sound. We have observed that their foraging activity disrupts eelgrass, and could retard recruitment of eelgrass. In addition, Dungeness crab bury in sediments as a predator defense mechanism. This burrowing activity may also disrupt newly recruiting eelgrass seedlings. Where population density is great as at Clinton, burrowing may be a significant factor inhibiting recruitment of eelgrass. Dungeness crab larvae are known to settle in shell piles. The shell offers shelter from predation as well as enhanced food resources for the young crab (Dumbauld et al. 1993). Enhanced crab abundances may be due to the availability of prime habitat for crab larvae.

The net effect of the propeller wash is to scour and redistribute sediments and associated biota and to lower irradiance. Redistribution of sediments is evident as a characteristic disturbed "ring" adjacent to the slip channel. This ring is barren of eelgrass. The reduced irradiance reaching the bottom may have effects on the growth rate and survival of eelgrass. In general, eelgrass exists as deep as light requirements

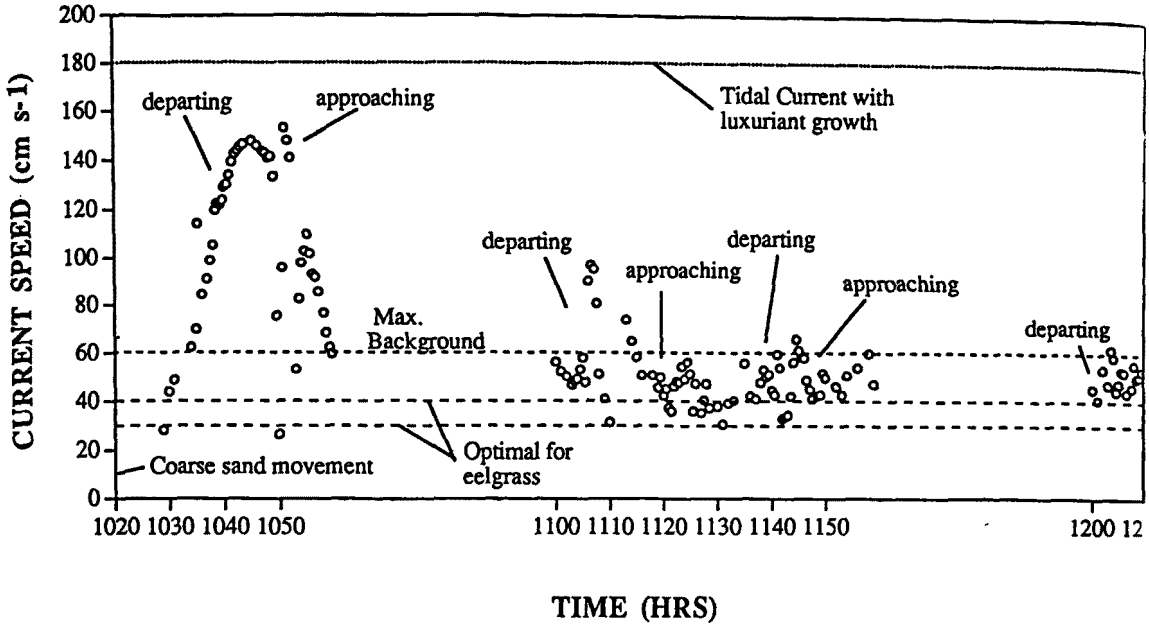

Figure 5. Current speeds in propeller wash plume. Current meter was positioned in Area F (see Figure 2).

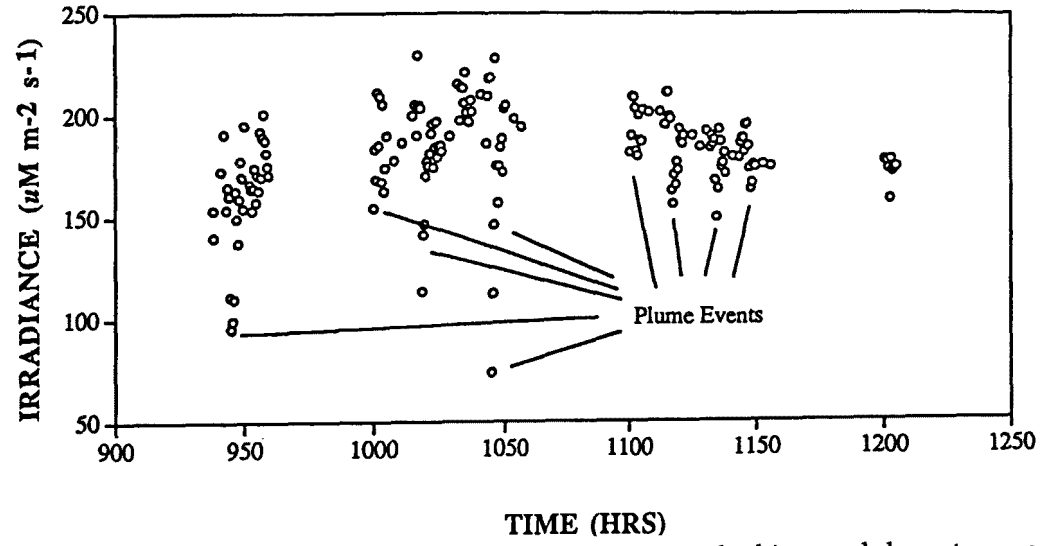

Figure 6. Bottom irradiance at Area F during ferry docking and departures on October 3, 1994 (see Figure 2).

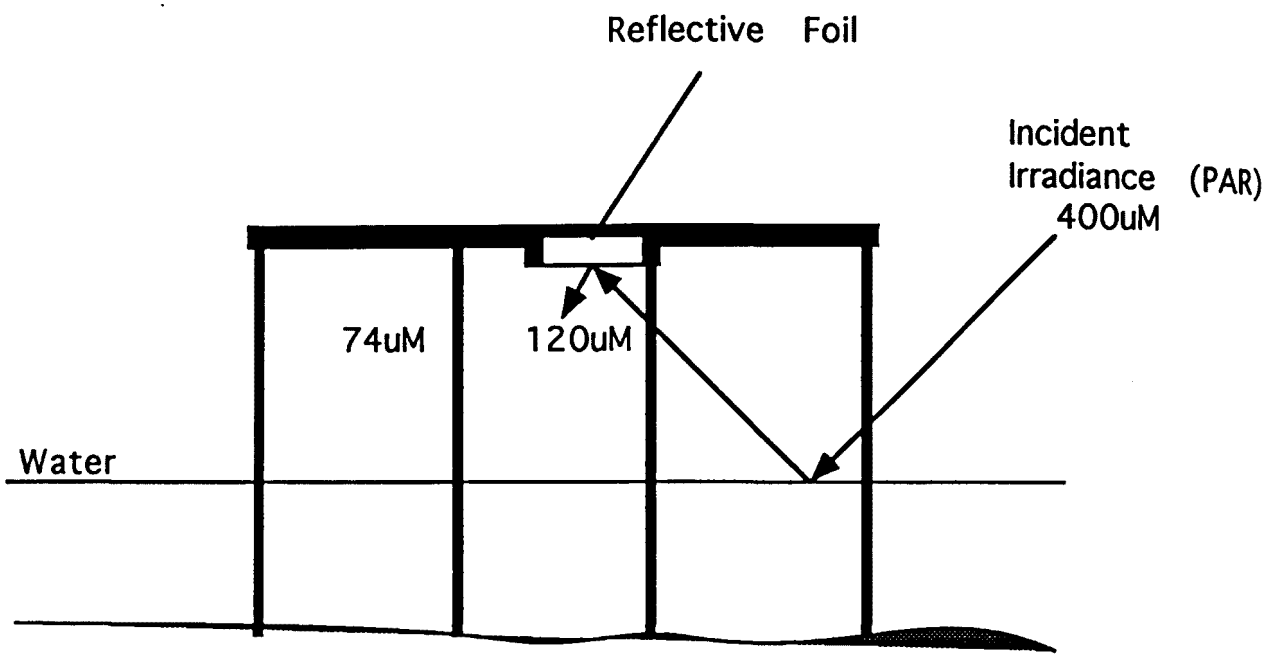

Figure 7. Average PAR Reflected Off Aluminum Foil and Wood Surface of Dock.

and suitable substrata will allow. At the outer (deepest) edge of the habitat, eelgrass growth is probably at a threshold and is highly dependent on some critical level of light reaching the plants. In areas where light is reduced, such as in the propeller wash plume, eelgrass photosynthesis may be inhibited enough to reduce growth below this critical threshold for survival. At present, we believe that the frequently-reduced irradiance associated with the plume has caused a reduction in the seaward extent of the eelgrass habitat near the Clinton terminal. Observations made by us at Clinton, other terminals (Edmonds, Port Townsend), and examination of aerial photographs of several other terminals, indicate that propeller wash is likely a significant factor affecting eelgrass distribution near the terminals.

Conceptual Model of the Dock System Impacts and Mitigation Measures

The Clinton Ferry Terminal Construction Project will involve replacement of the existing timber pile structure with a concrete pile structure, and expansion of the holding area to accommodate increased ferry traffic (Figure 2). The dock will be widened by approximately 19 m and lengthened by approximately 44 m. A new south slip, steel wing walls, floating dolphins, towers and headframe will also be constructed.

The present condition of the eelgrass habitat at Clinton is due to historical (not presently active) as well as current processes (Figure 8). Disturbance to the habitat is partitioned into two phases: construction, and maintenance and operation.

Terminal expansion will have short-term direct effects and longer term impacts on eelgrass. Initial construction activities are predicted to have some limited effect on eelgrass, although these activities will be largely conducted away from existing beds. The new terminal deck will cover 320 m^2 of eelgrass presently located on the south side of the terminal. The proposed mitigation measures are directed at eliminating any longer-term effects.

With potential impacts to eelgrass identified through a series of meetings with state and federal resource agencies, the Washington State Department of Transportation undertook a program to identify necessary actions to avoid, minimize and compensate for these impacts. Impacts expected under the original design plan for the terminal have been either avoided or minimized. For example, ferry propeller wash impacts have been avoided by moving the slips further offshore. Light impacts have been minimized through incorporation of light transmitting structures (concrete blocks with glass centers) in the terminal deck, and lengthening the terminal. In addition, highly reflective paint (i.e., the type used for painting white lines on roads) will be used under the terminal to enhance the albedo. Lengthening the terminal reduced the width of the terminal at the point where it crosses the eelgrass habitat. Maintenance activities have been reduced dramatically through the use of concrete piles and decking as opposed to timber (which was proposed in the original plan). Use of concrete pilings will result in placement of 1/3 fewer pilings than presently exist. This will reduce the amount of space for piling communities to develop and may support fewer seastars and dungeness crab. This may result in less bioturbation effects on eelgrass. Fewer pilings will also allow more light to penetrate under the terminal.

We believe that, with appropriate modifications in the terminal expansion, eelgrass can be restored in many of the areas where eelgrass has been eliminated by past or on-going disturbances. These areas are identified by letters A-H in Figure 2. Probability of success varies among the areas. For example, areas A, B, C, D, E, F, and G are considered areas where disturbances can be essentially eliminated and eelgrass transplants have a moderate to high probability of being successfully established. Other areas are less likely to succeed due to their experimental nature (e.g., area H under glass blocks).

The overall mitigation goal for the Clinton Ferry terminal Expansion project is no *net loss of eelgrass on the project site.* This goal will be achieved by:

1. Reduction in impacts as much as possible in the design and construction of the project (as described above);
2. Avoidance of future impacts associated with operations at this terminal by
3. Locating all anchors or pilings for any floating dolphins associated with the terminal outside of eelgrass areas, b. avoid moorage of barges over eelgrass areas; and,
4. Reduce local habitat fragmentation through restoration of previously disturbed eelgrass patches through transplantation of eelgrass into these areas. Approximately 320 m^2 of eelgrass will be directly impacted by the terminal expansion. Total restoration area (A-H in Figure 2) is

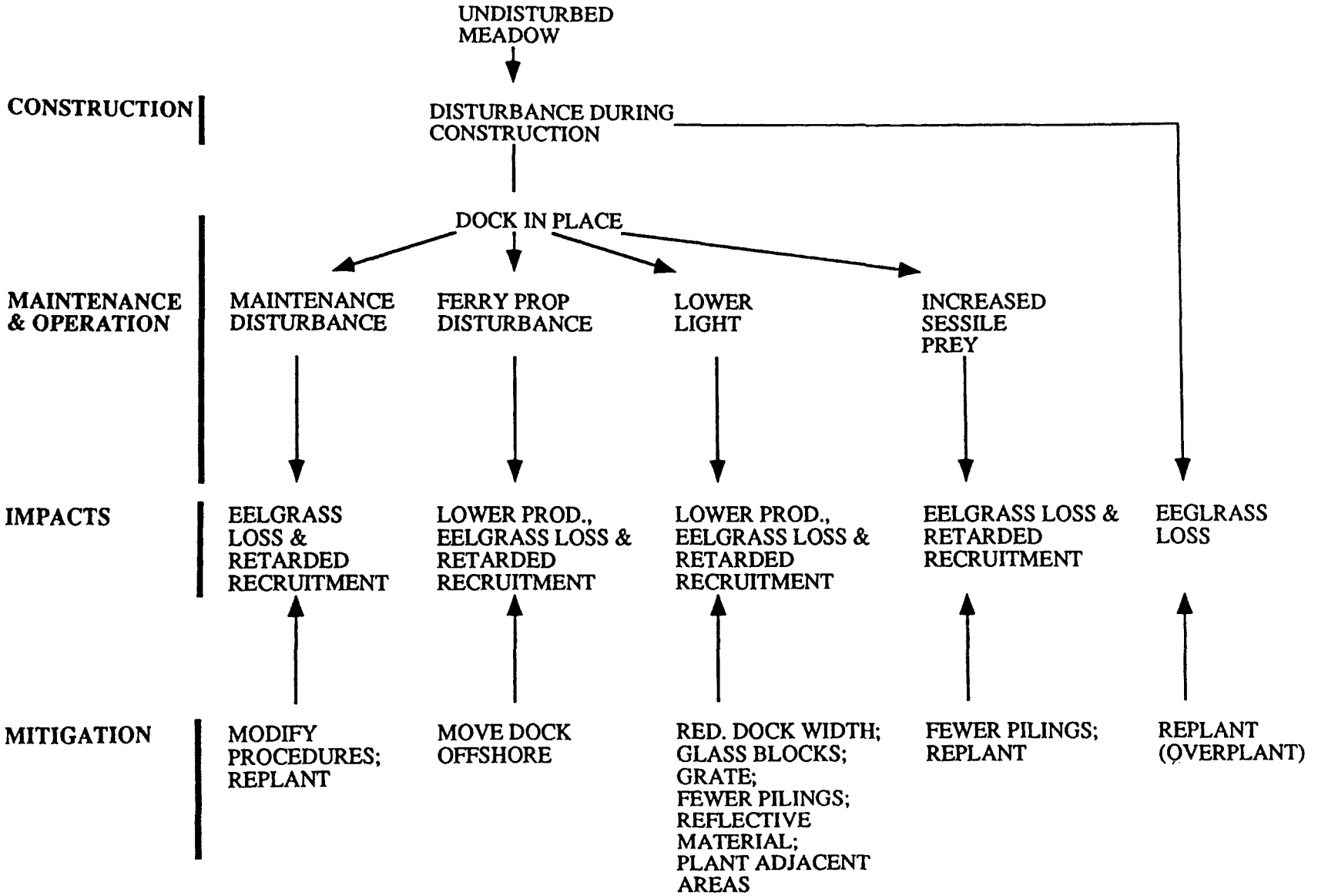

Figure 8. Model of Impacts and Mitigation Measures at Clinton Terminal.

approximately 3,100 m^2, which would result in an approximately 10:1 replacement of the area directly impacted by the dock expansion. The additional area is required to offset poor transplant survival and growth in some areas (Fonseca 1990).

The National Research Council (NRC 1992) has recommended that restoration and mitigation projects should incorporate experiments to help test new technologies and evaluate methods that may help in future projects. Thus, an additional goal of the project is to *evaluate some new concepts in mitigating shading effects of overwater structures.* The use of glass blocks in the passenger walkways and reflective paint under the terminal deck represent ideas that, if proven feasible and successful through post-construction monitoring, could then be incorporated into future terminal expansions elsewhere.

Acknowledgments

This research was supported by the Washington State Department of Transportation. We gratefully acknowledge the excellent logistical support and cooperation provided by WSDOT staff, and the Clinton terminal managers and workers. J. Norris conducted the video surveys and B. Feist produced the GIS-based map of the eelgrass habitat. The map in Figure 1 was produced by the Transportation Northwest Regional Center at the University of Washington.

Literature Cited

Baldwin, J. R. and J. R. Lovvorn. 1994. Habitats and tidal accessibility of the marine foods of dabbling ducks and brant in Boundary Bay, British Columbia. Marine Biology 120:627-638.

Burdick, D. M. and F. T. Short. 1995. The effects of boat docks on eelgrass beds in Massachusetts coastal waters. Submitted to Waquoit Bay National Estuarine Research Reserve and Massachusetts Coastal Zone Management, Jackson Estuarine Laboratory, University of New Hampshire, Durham, NH.

Dumbauld, B. R., D. A. Armstrong and T. L. McDonald. 1993. Use of oyster shell to enhance intertidal habitat and mitigate loss of Dungeness crab *(Cancer magister)* caused by dredging. Canadian Journal of Fisheries and Aquatic Sciences 50:381-390.

Fonseca, M. 1990. Regional analysis of the creation and restoration of seagrass systems. pp. 171-189 In Jon A. Kusler and Mary E. Kentula, eds. Wetland creation and restoration the status of the science. Island Press.

National Research Council (NRC). 1992. Restoration of aquatic ecosystems. National Academy Press, Washington, D.C.

Pentilla, D. and D. Doty. 1990. Results of 1989 eelgrass shading studies in Puget Sound. Unpublished progress report. Washington State Department of Fisheries, Olympia, WA.

Phillips, R. C. 1984. The ecology of eelgrass meadows in the Pacific Northwest: a community profile. U.S. Fish and Wildlife Service FWS/OBS-84/24.

Thom, R. M. and L. Hallum. 1991. Historical changes in the distribution of tidal marshes, eelgrass meadows and kelp forests in Puget Sound. pp. 302-313 In Puget Sound Research '91 Proceedings. Puget Sound Water Quality Authority, Olympia, WA.

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Creating Tidal Marshes on Dredged Materials: Design Features and Biological Implications

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Abstract: **The use of dredged materials to create tidal marshes is not new, but there is a lack of discussion on design parameters necessary for constructing tidal marshes that are similar structurally and functionally to natural tidal marshes. Our study was designed to: (1) investigate the similarity and differences in selected physical and biological components of natural tidal marshes compared to tidal marshes created on dredged materials; and (2) identify important design features to consider in designing future tidal marsh restoration projects using dredge materials to increase the likelihood of achieving functional equivalence. Four tidal marshes were selected for study in San Francisco Bay, California, two that had been constructed using dredged materials (Muzzi Marsh and Faber Tract) and two natural tidal marshes (Corte Madera Ecological Reserve and Laumeister Tract). There were significant differences in the percent cover by tidal marsh vegetation (primarily *Spartina foliosa* and *Salicornia virginica*) between the natural tidal marshes and constructed tidal marshes. These differences appeared to be related to the poor development of a tidal slough channel network at the constructed tidal marshes. At the two constructed tidal marshes, plant heights of *Spartina* and *S. virginica* were greater where there was a well-developed slough channel system, compared to areas where marsh surface elevation has apparently prevented the formation of slough channels. The major conclusion of the study, is that the initial elevation of the dredged materials when tidal activity is restored, is a major determinate of slough channel density and morphology in constructed marshes. The presence of a well-developed tidal slough channel system is important if the constructed marsh is to achieve functional equivalence with natural tidal marshes.**

Introduction

San Francisco Bay (Bay) is an important natural resource, containing a variety of habitats ranging from deep water to shallow intertidal mudflats and tidal marshes. Because of the geomorphological structure of the Bay, maritime trade has become an important component of the economy of the communities surrounding the Bay. However, historical land use practices have greatly reduced some of the valuable natural features of the Bay, especially the acreage of tidal marshes along the margins of the Bay.

The historical loss of tidal marshes around the Bay margins has been well documented (e.g., Josselyn 1983). Since the middle of the last century, one-third of the Bay has been lost due to filling and 90 percent of the Bay's wetlands (tidal marshes and other types of wetlands) converted to other uses (National Research Council 1992). Prior to the gold rush, the acreage of tidal marshes (2,200 km^2) was twice the area of the Bay's open water (National Research Council 1992). Approximately 35 percent of the tidal marshes were located in the Bay and the remaining 65 percent in the Sacramento River-San Joaquin River delta (Delta) (National Research Council 1992). As a result of diking, filling, and anthropogenic sedimentation, only about 5 percent of the historical tidal marsh acreage remains (Atwater 1979).

Sedimentation of navigational channels and harbors has established a need to dredge to maintain and expand the maritime trade in the Bay. The disposal of dredged materials has been controversial in recent years due to the perceived impacts to the environment resulting from the deposition of dredged materials,

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especially in the Bay. The use of dredged materials to create habitat, including tidal marshes, is not new and was the subject of the U.S. Army Corps of Engineers (Corps) Dredged Material Research Program (Environmental Laboratory 1978). This program concluded that the use of dredged material as a substrate for tidal marsh development was a feasible alternative.

Several tidal marsh restoration projects established on dredged materials have been completed in the Bay and Delta region, and several projects are in the planning stage. Hayward Salt Pond No. 3 in the south Bay, Muzzi Marsh in the north Bay, and Donlan Island and Venice Cut in the San Joaquin River were all established using dredged materials and have been subject to long-term monitoring programs (National Research Council 1994). Faber Tract is another tidal marsh restoration on dredged materials, but there has been only limited monitoring of this project, which occurred early in its project life.

Determination of success of these and other restoration projects is a matter of some controversy. The four projects constructed in the Bay with long-term monitoring (Salt Pond No. 3, Muzzi Marsh, Donlan Island, and Venice Cut) are considered to be successful based on perceived attainment of project goals and objectives (England and Nakaji 1990, Landin et al. 1989, Landin 1985, 1990). Landin (1985) reviewed the results of multi-year monitoring of a number of wetlands constructed with dredged materials throughout the United States, including California, and her conclusions regarding the success of these projects were generally favorable.

Determination of success of tidal marshes constructed on dredged materials has been hampered by the lack of well defined goals and objectives and inadequate monitoring. For example, Salt Pond No. 3 is dominated primarily by *Salicornia virginica* and no slough channels have formed naturally. Some may conclude that the presence of relatively dense *S. virginica* means that the project was a success. However, because *Spartina foliosa* was originally planted, the implied purpose of the project (establish a *Spartina* marsh) was not attained and the resulting constructed tidal marsh does not approach functional equivalency of natural *Spartina* marshes.

Minimal information has been developed on designing tidal marsh systems to maximize the likelihood of achieving functional equivalence in constructed tidal marshes. Questions remain about the time frame for attaining functional equivalency in constructed tidal marshes, and even if functional equivalency can be attained. Numerous studies have documented the development of various structural and functional attributes of constructed tidal marshes but no clear pattern has emerged from these studies (e.g., Cammen et al. 1974, Seneca et al. 1976, Craft et al. 1988, Zedler 1988, LaSalle et al. 1991, Havens et al. 1995, Simenstad and Thom 1996). In fact, Havens et al. (1995) concluded that *"[T]he question of if and how long does it take for a constructed marsh to achieve the same level of function as similar natural marshes is unresolved."*

The lack of specific guidance relative to designing and constructing tidal marsh systems to achieve functional equivalency, or to approach functional equivalency in the shortest time possible, is troubling. We felt that the tidal marshes constructed on dredged materials in the Bay offered an opportunity to identify key components of the tidal marsh system that should be considered in the design of future tidal marshes constructed on dredged materials. Therefore, this study was designed to: (1) investigate the similarity and differences in selected components of the physical and biological structure of natural tidal marshes compared to tidal marshes created on dredged materials and; (2) discuss important design features to consider in designing tidal marsh restoration projects using dredged materials to increase the likelihood of success.

Description of Study Sites

Two pairs of sites were studied: (1) Muzzi Marsh and the Corte Madera Ecological Reserve (CMER) in the north Bay (MHHW - 4.34 ft. NGVD); and (2) Faber Tract and Laumeister Marsh in the south Bay (MHHW - 3.13 ft. NGVD) (

1). Muzzi Marsh and CMER were once part of a historical tidal marsh system that extended along the edge of Corte Madera Bay. While CMER remained undiked, Muzzi Marsh was diked in the 1950's for use as a future industrial site. Approximately 750,000 cubic yards of dredged material were placed within the diked portion of the property (Faber 1983, 1988, 1990), with the landward portion of the site at a higher elevation (about 1.3 feet higher than MHHW) than the bayward portion (about 1.1 to 2.1 feet below MHHW). A training dike separated the upper and lower portions. Tidal activity was restored by breaching the bayward dike in 1976.

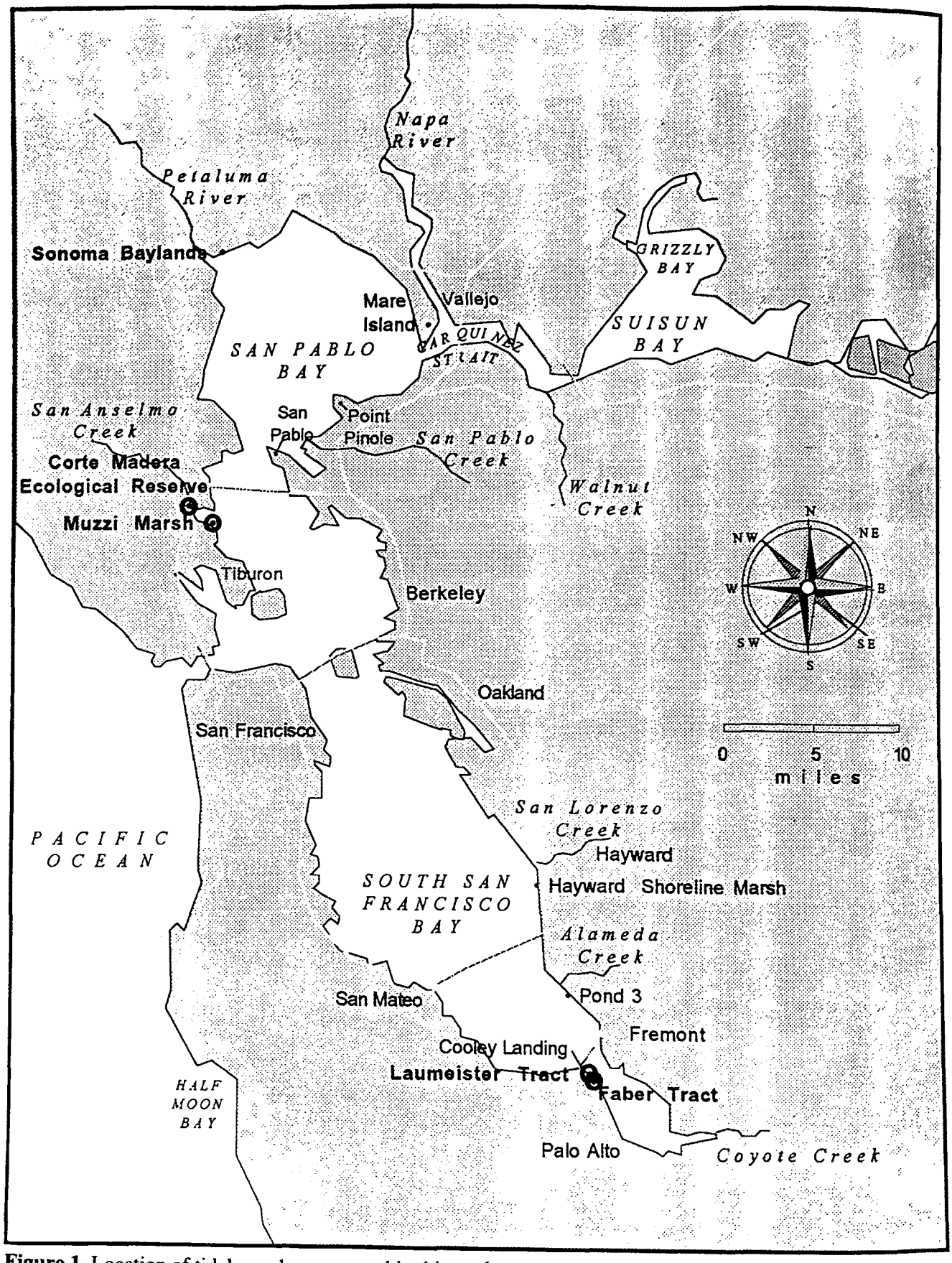


Figure 1. Location of tidal marshes surveyed in this study.

Faber Tract and Laumeister Tract were once part of an extensive tidal marsh along the west side of south San Francisco Bay. The Laumeister Tract remained undiked, while the historical marsh at Faber Tract was diked and drained in 1945 for cattle grazing. In 1968, 200,000 cubic yards of sediment was deposited at Faber Tract for the restoration of a tidal marsh and opened to tidal circulation in 1971 (Harvey 1970, Harvey et al. 1982). Our estimate of the average elevation of Faber Tract following discharge of the dredged material in 1969 ranges from 2.9 to 3.5 feet NGVD (Gahagan & Bryant Associates et al. 1994). The upper portion of Faber Tract ranged in elevation from 4.2 feet NGVD (~0.1 ft. below MHHW) to an average elevation of 3.5 feet NGVD near the middle of the marsh, while the lower portion of Faber Tract sloped down to an elevation of 2.6 feet NGVD.

Methods

Physical Measurements

Topographic profiles were surveyed along a transect from the high end to the low end of the marsh plain in each of the four tidal marsh systems. The preliminary location of each transect was selected using recent aerial photographs of each of the tidal marsh systems and finalized during the field survey. The topographic profiles provided data on the elevation and gradient of the existing marsh plain and the location and depth of slough channels not visible on aerial photographs. In addition, the aerial photographs were used to determine the following morphometric characteristics of the tidal marsh slough channel system: stream order, drainage density, and sinuosity. These measures provide a quantitative basis for determining the similarity of tidal slough channel systems between constructed tidal marshes and natural tidal marshes.

Stream Order

We used Horton's system of stream ordering to characterize the tidal slough channels on the marsh plain (see Straler 1964). We used a consistent scale of 1 inch equal to 500 feet on all of the morphometric measurements to address the problem of map or photo scale (Leopold and Miller 1956) so that the marshes could be compared.

Drainage Density

Drainage density (D) provides a measure of the abundance of slough channels. Drainage density is defined as the length of all channels in a marsh (ΣL) divided by the marsh area (A):

$$D = \frac{\Sigma L}{A}$$

In tidal marshes, a high drainage density is associated with frequency and sinuosity of slough channels, both of which are characteristic of good tidal circulation. The frequency of slough channels refers to the number present within a specific area. The sinuosity refers to the length of the channel along its thalweg (or deepest part) relative to the straight line distance between its head in the upper part of the marsh and its mouth at the Bay.

Sinuosity

Slough channel sinuosity is the ratio of slough channel length (L_c) to the straight line length (L_d) of the slough channel drainage system:

$$S = \frac{L_c}{L_d}$$

Sinuosity is a characteristic of slough channel development on a marsh plain. Poorly developed slough channels tend to be relatively straight while well developed channels tend to be relatively sinuous.

Biological Measurements

Vegetation structure was investigated along a single transect established in each marsh. This transect extended from the upland/tidal marsh boundary of each marsh to the low marsh/mudflat boundary (edge of the vegetation). The selection of transects at CMER, Faber Tract, and Laumeister Tract was based on the physical characteristics of the marsh and on review of recent aerial photographs. The transect surveyed at Muzzi Marsh was the same as that used as part of continuing studies at Muzzi Marsh (Faber 1990). We limited our sampling points to the topographic transect because we wanted to investigate the relationship between measured vegetation parameters and the physical data (e.g., elevation, extent of slough channels).

The location of plots along each transect was determined randomly. At each sampling plot, the following data were collected from a 1-meter-square quadrat: percent live cover by cover class (class 1 = 0-1 percent, class 2 = 2-5 percent, class 3 = 6-25 percent, class 4 = 26-50 percent, class 5 = 51-75 percent, class 6 = 76-100 percent), percent bare ground, and height of each species. In plots with a cover class for *Spartina foliosa* greater than 3, the length of each shoot of *Spartina* in a 0.1-meter-square quadrat randomly located in one of the four corners of the larger quadrat was measured to the nearest centimeter.

Data Analysis

Differences in percent cover of *Spartina foliosa* and *Salicornia virginica* between pairs of stations (CMER vs. Muzzi Marsh and Laumeister Tract vs. Faber Tract) were investigated using the Mann-Whitney test at a significance level of 0.05. Analysis of other marsh vegetation species was not attempted as these species were not common at all the stations. Differences in plant height of the tallest occurrence of *S. virginica* in each plot was investigated using the paired *t*-test. Analysis of variance was used to investigate differences in the average shoot length of *Spartina* between Muzzi Marsh, Laumeister Tract, and Faber Tract. *Spartina* was not present in the sample plots at CMER.

Results

Topographic Profiles

The marsh plain at CMER was relatively flat with an average elevation of 3.4 feet NGVD (slightly higher than MHHW). Slough channel bottoms along the transect range in elevation between 3.0 to 1.1 feet NGVD. The marsh plain at Muzzi Marsh can be divided into two portions, the upper, higher elevation marsh from the western levee to about 950 feet at the location of the training dike, and the lower elevation marsh from 950 feet to the Bay (Figure 2). The upper marsh had an average elevation of 3.3 feet NGVD (slightly higher than MHHW). The lower marsh had an average elevation of 2.3 feet NGVD between MLW and MTL. Slough channel bottoms along the transect in the upper part of the marsh ranged in elevation between -0.1 and 2.0 feet NGVD (Figure 3).

The marsh plain at Faber Tract sloped toward the north away from the original dredge material discharge locations with a gradient of about 0.0005 from a distance of 400 feet to 1,800 feet along the transect (Figure 4). The upper part of the marsh had an average elevation of about 4.0 feet NGVD and a higher gradient of about 0.001. There were no developed slough channels in the upper portion of the Faber Tract from a distance of 0 to 400 feet along the transect (Figure 5). In the lower portion of the marsh, the average marsh plain elevation was about 3.3 feet NGVD and slough channel bottoms along the transect range in elevation from 3.2 to 0.9 feet NGVD. The marsh plain at Laumeister Tract was relatively flat and had an average elevation of 4.2 feet NGVD (between MHW and MHHW)(Figure 4). The bayward portion of the marsh was slightly higher (0.1 to 0.2 feet) on the average than the landward portion of the marsh, which indicates greater sedimentation on the bayward portion of the marsh plain. Slough channel bottoms along the transect ranged in elevation between 3.5 to 0.0 feet NGVD (Figure 5).

Tidal Slough Channel Morphology

Slough channel morphology on the natural tidal marshes was found to differ from that on the constructed tidal marshes (Table 1). The natural tidal marshes and lower portions of the constructed marshes where the dredged material was placed at a relatively low elevation, dense networks of sinuous slough channels have developed (drainage densities range from about 30 to 40 mi/mi^2). In contrast, on portions of the constructed marshes where the initial elevation of the dredged materials was too high (upper portion of Muzzi Marsh and the upper portion of Faber Tract) slough channels were poorly developed after at least two decades (see Figure 2 [Muzzi Marsh] and 4 [Faber Tract]).

The difference in tidal slough development is reflected in the morphometric measures made at each tidal marsh (Table 1). At Muzzi Marsh, the low marsh had substantially greater drainage density than the higher marsh. Although Faber Tract had a higher slough channel density than Laumeister Tract, the sloughs at Laumeister Tract were more sinuous. At Faber Tract, there were few tidal slough channels in the higher southern portion of the marsh. In addition, the fourth order channel may be an artifact of construction of the marsh as it slopes to the north rather than bayward (Figure 4).

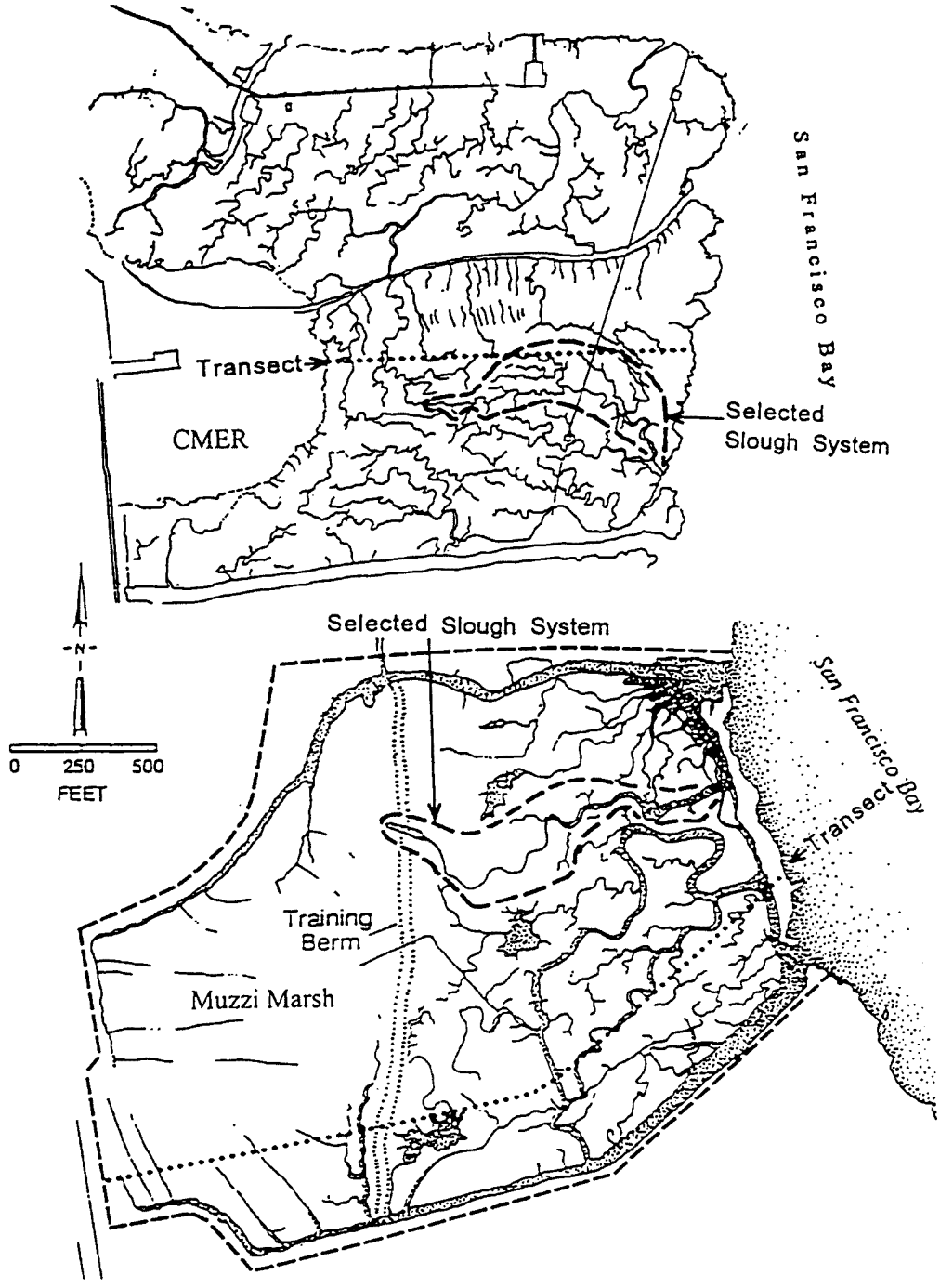
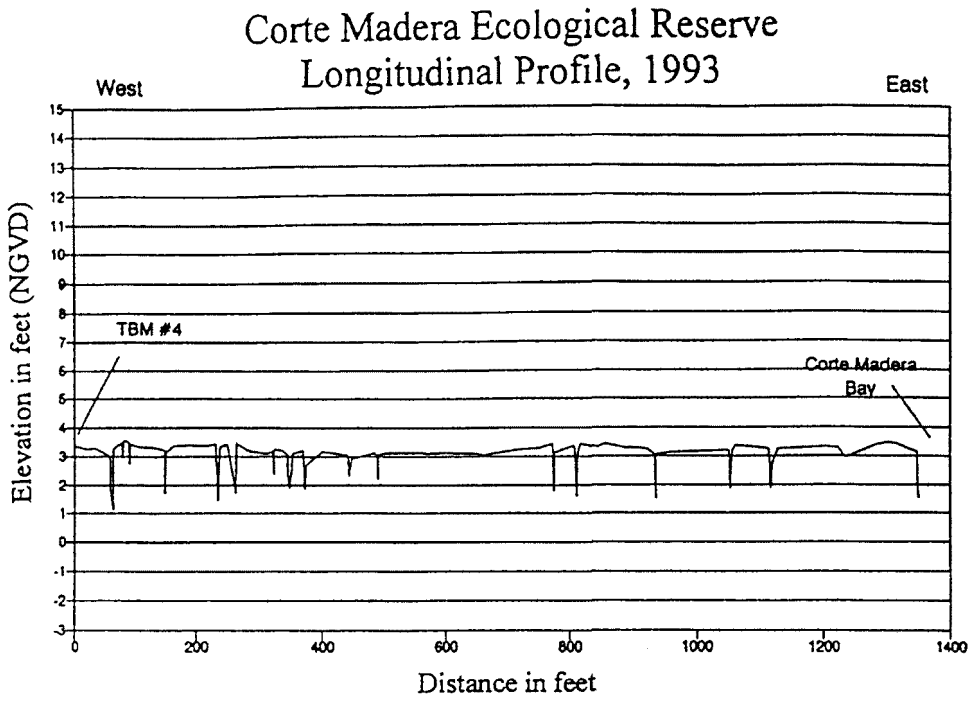


Figure 2. Tidal Slough Channels at Corte Madera Ecological Reserve (top) and Muzzi Marsh (bottom).



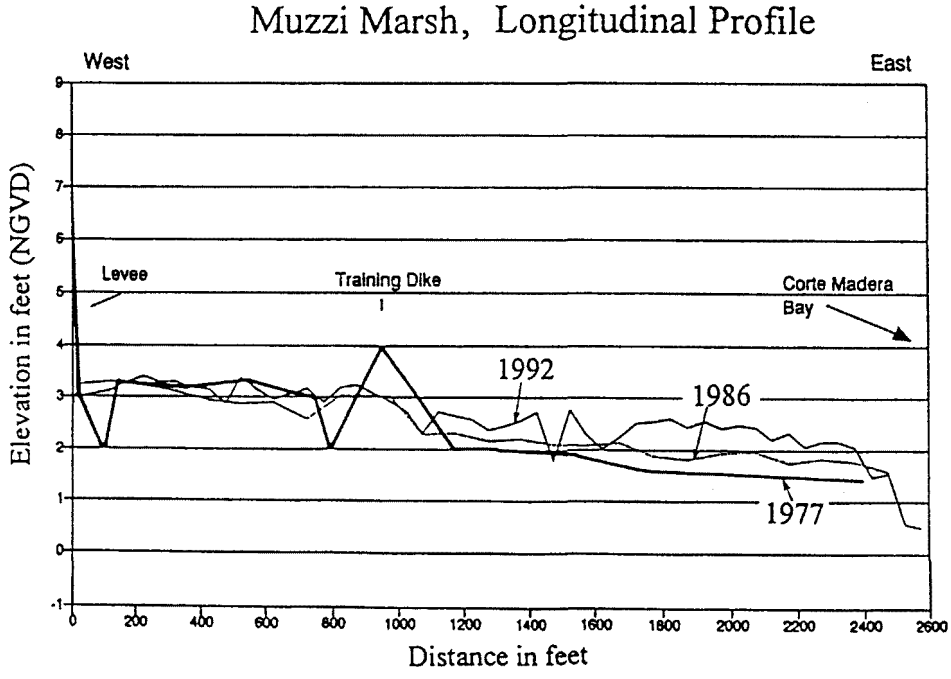


Figure 3. Topographic Profile at Corte Madera Ecological Reserve (top) and Muzzi Marsh (bottom).

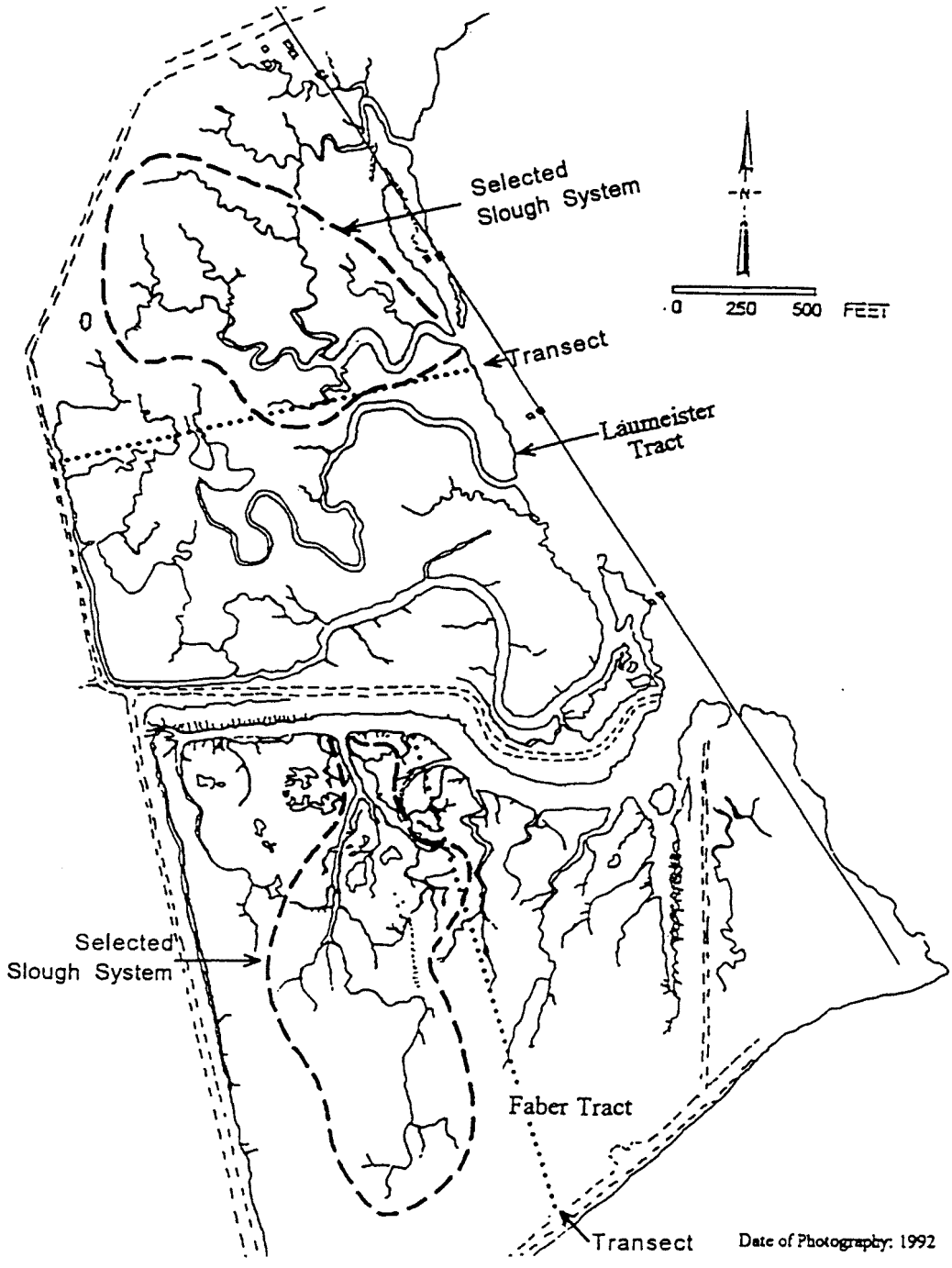


Figure 4. Tidal Slough Channels at Laumeister Tract and Faber Tract.

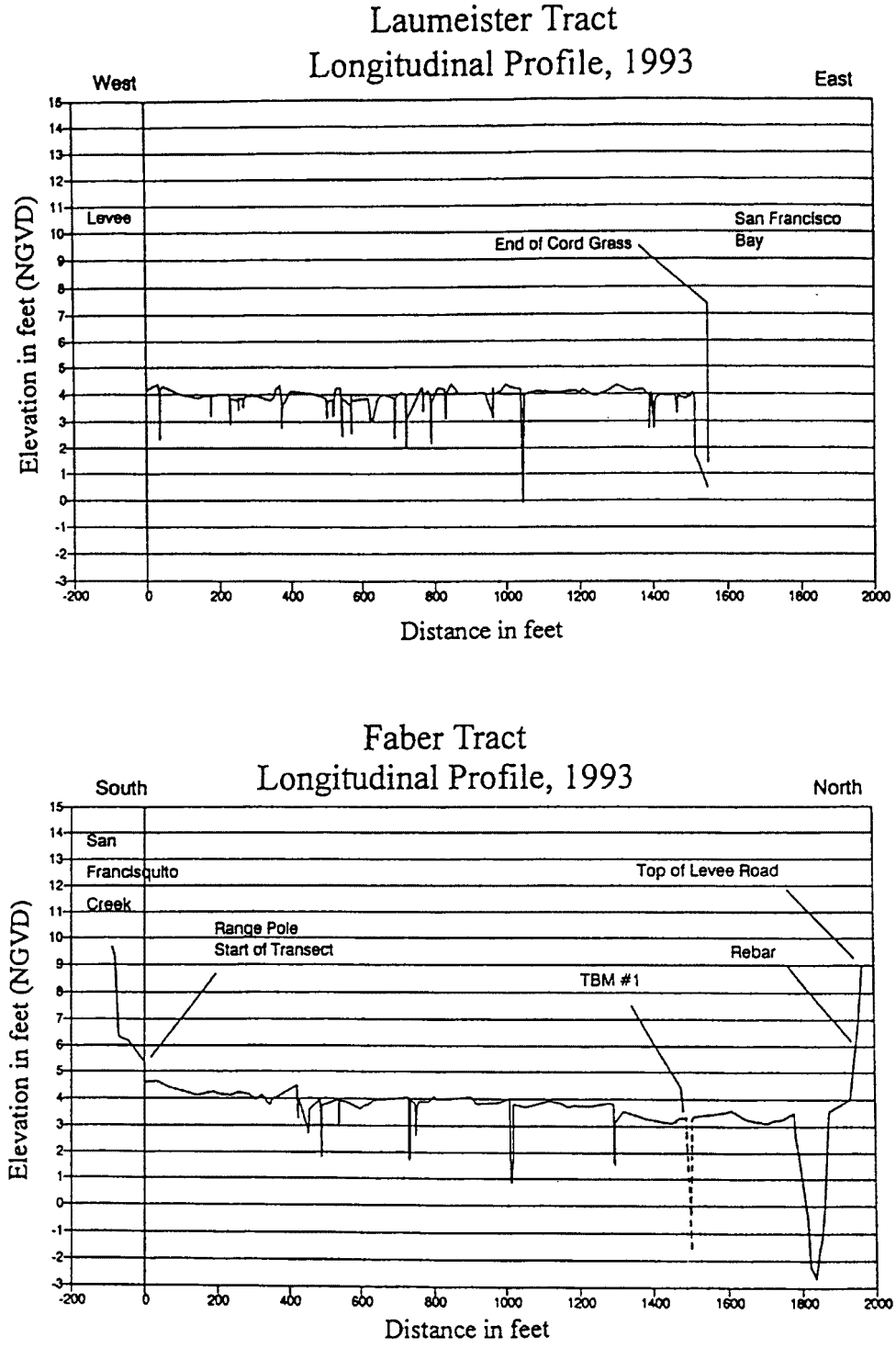


Figure 5. Topographic Profile at Laumeister Tract (top) and Faber Tract (bottom).

Table 1. Morphometric Characteristics of Selected Slough System.

CORTE MADERA ECOLOGICAL RESERVE

Slough Channel Order	Number of Sloughs in Order	Length of Sloughs in Order (mi.)
1	19	0.26
2	4	0.19
3	2	0.06
4	1	0.05

- Drainage Density of Selected Slough System = 56 mi./mi.^{2}
- Sinuosity = 1.11

MUZZI MARSH

Slough Channel Order	Number of Sloughs in Order	Length of Sloughs in Order (mi.)
1	10	0.22
2	3	0.05
3	1	0.12
4	0	

Lower Marsh

- Drainage Density of Selected Slough System = 39 mi./mi.^{2}
- Sinuosity = 1.51

Upper Marsh

- Drainage Density of Entire Marsh = 9 mi./mi.^{2}

LAUMEISTER TRACT

Slough Channel Order	Number of Sloughs in Order	Length of Sloughs in Order (mi.)
1	13	0.42
2	5	0.28
3	1	0.22
4	0	0

- Drainage Density of Selected Slough System = 31 mi./mi.^{2}
- Sinuosity = 1.21

FABER TRACT

Slough Channel Order	Number of Sloughs in Order	Length of Sloughs in Order (mi.)
1	33	0.56
2	9	0.34
3	2	0.28
4	1	0.04

- Drainage Density of Selected Slough System = 41 mi./mi.^{2}
- Sinuosity = 1.10

The relation of slough channel development to initial elevation after placement of dredged material at Muzzi Marsh and Faber Tract suggests that the initial elevation of the dredged material was the most important determinant of slough channel development in marshes constructed using dredged materials. The upper portion of Muzzi Marsh was filled to 4.4 feet NGVD, an elevation higher than MHHW. In contrast, the lower portion of Muzzi Marsh ranged between 1.0 to 2.0 feet NGVD (between MTL and MHW) similar to the elevation prior to discharge of dredged material. Only one small slough channel system has developed naturally in the upper portion of Muzzi Marsh. A dense network of sinuous slough channels exist on the mudflat and marsh plain in the lower portion of Muzzi Marsh, indicating that the initial elevation of the dredged materials has allowed for the successful evolution of the slough channel system and simultaneous development of the marsh plain.

Tidal Marsh Vegetation

Vegetation Cover

Salicornia virginica was the most abundant and frequently occurring species at each of the marshes surveyed. At CMER, *S. virginica* occurred in 38 of the 39 plots sampled while at Muzzi Marsh, *S. virginica* occurred in 34 of the 41 plots surveyed. The greatest number of species (7) was encountered in the sample plots at Laumeister Tract. In addition to *Salicornia virginica* and *Spartina foliosa*, other species included *Distichlis spicata, Jaumea carnosa, Suaeda* sp., *Atriplex patula*, and *Frankenia grandifolia*. *S. virginica* occurred in 35 of the 38 plots sampled and *S. foliosa* occurred in 33 of the 38 plots. Only *S. virginica* and *S. foliosa* were encountered at Faber Tract, with *S. virginica* occurring in all the plots.

Mean percent cover for *S. virginica* varied from approximately 38 percent at Laumeister Tract to approximately 67 percent at Faber Tract (Table 2). At the south Bay and north Bay sites, the mean percent cover of *S. virginica* at constructed marshes (Muzzi Marsh and Faber Tract) was significantly greater than at the natural marshes (CMER and Laumeister Tract).

Because of the influence of elevation on the occurrence of plants in tidal marshes, possible differences in percent cover of *S. virginica* and *S. foliosa* at similar elevation classes between the natural marsh and constructed marsh were investigated. Since most of the sample plots occurred between MTL and MHW, percent cover in this elevation class was compared between the four tidal marshes. The difference in percent cover of *S. virginica* between Muzzi Marsh and CMER in the north Bay was not significant in this elevation range, but the percent cover at Faber Tract was significantly higher than at Laumeister Tract. Above MHW, the percent cover of *S. virginica* was significantly higher at Muzzi Marsh than CMER. There were not enough plots above MHW at the other two sites to test for difference in percent cover of *S. virginica*.

Plant Height

The height of *S. virginica* was significantly greater at Laumeister Tract was than at Faber Tract and Muzzi Marsh. The difference between the other stations were not significant. Although the mean height of *S. virginica* in the elevation class MTL to MHW at Muzzi Marsh was greater than the mean height at CMER at the same elevation, the difference was not significant. However, at the south Bay stations, the height of *S. virginica* was significantly greater at Laumeister Tract than at Faber Tract between MTL and MHW.

Spartina Stem Length and Density

The mean stem length (Table 3) was significantly greater at Muzzi Marsh than at Faber Tract or Laumeister Tract, but *Spartina* density and total stem length per plot (Table 3) were not significantly different from one another. Total stem length and distribution pattern of individual stem length were similar to that observed by Zedler (1993) in San Diego County tidal marshes. Most of the stems occurred in the 61 to 90 cm range, as shown by the cumulative percent frequency curves for the three sites (Figure 6). Although the highest stem densities occurred at Faber Tract, a constructed marsh in south Bay, the difference in mean density between marshes was not significant.

Discussion

If we are to construct tidal marshes that are functionally equivalent to natural tidal marshes, implementing an approach that takes advantage of the self-designing capacity of nature (Mitsch and Wilson 1996) is the preferred option. Attempting to define the final planform for a constructed tidal marsh presents several challenges, each of which could extend the time it takes the developing constructed tidal marsh from approaching functional equivalency with natural tidal marshes and possibly even could prevent

Table 2. Percent Cover for *Spartina foliosa* (SPFO) and *Salicornia virginica* (SAVI).

MARSH	SPFO	SAVI
CMER		
No. of quadrats sampled	39	39
Mean	0	47.2
Range	0	0-87.5
Muzzi Marsh		
No. of quadrats sampled	41	41
Mean	20.55	60.37
Range	0-87.5	0-87.5
Laumeister Tract		
No. of quadrats sampled	38	38
Mean	22.17	37.83
Range	0-67.5	0-87.5
Faber Tract		
No. of quadrats sampled	42	42
Mean	9.29	67.14
Range	0-67.5	2.5-87.5

Table 3. Stem Length, Total Stem Length and Density for *Spartina foliosa*.

Parameter	Muzzi Marsh	Laumeister Tract	Faber Tract
Stem Length (cm)			
Number of stems	123	324	181
Mean	68.9	67.9	61.7
Range	8-118	14-98	5-102
Total Stem Length (cm)			
Number of quadrats	8	21	9
Mean	1059.2	1047.4	1240.6
Range	319-1626	271-1986	141-2936
Density ($/m^2$)			
Number of quadrats	8	21	9
Mean	153.8	154.3	201.1
Range	50-250	40-280	50-460

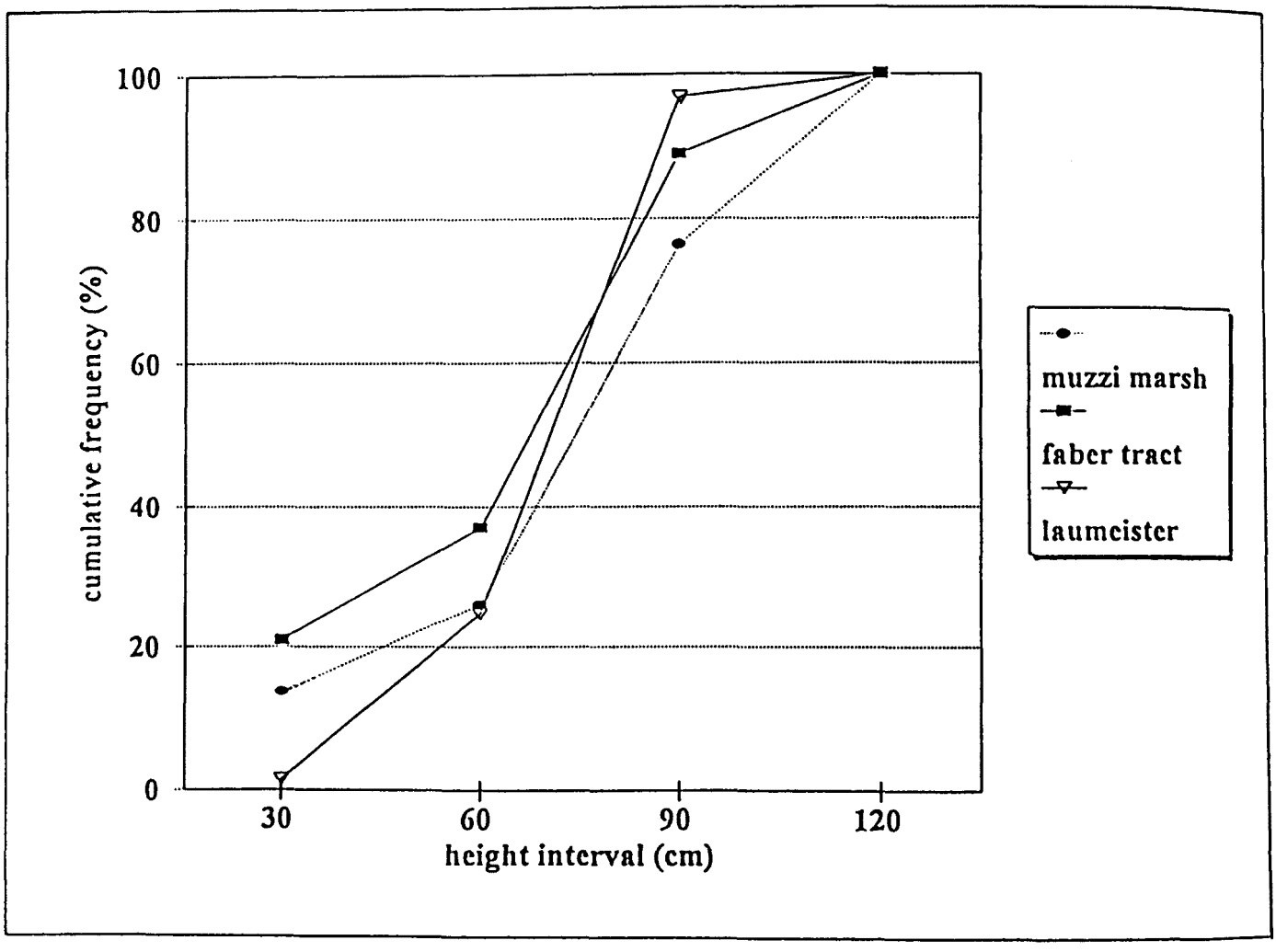


Figure 6. Cumulative Frequency for *Spartina foliosa* Stem Length at Muzzi Marsh, Faber Tract and Laumeister Tract.

the attainment of desired functions. The lack of development of a tidal slough channel system at the upper Muzzi Marsh, upper areas of Faber Tract and Salt Pond 3 appears to be the result of overfilling. Overfilling is more likely to occur if attempting to achieve the desired marsh planform during construction instead of allowing the system to evolve from lower elevations as natural sedimentation occurs.

The differences in the structure of the vegetation communities between the natural marshes and constructed marshes appear to be related to the development of the tidal slough channel system, which in turn is the product of the initial elevation of the constructed site when tidal activity was restored. Observations at Muzzi Marsh and Faber Tract indicates that in portions of the marsh where the initial surface elevations were too high, slough channels have not developed. At Faber Tract, slough channels were absent in the upper part of the marsh, occurred intermittently in the middle portion of the marsh, and were abundant in the lower marsh. Cover of *S. foliosa* was significantly lower in the upper marsh area compared to the area of the marsh where there were intermittent and abundant slough channels. *Spartina* growing in the area of the marsh where slough channels were abundant was significantly taller than plants occurring where there were intermittent or no slough channels. These results suggest that good tidal flushing is necessary to get maximal growth of *Spartina* and other tidal marsh plant species.

The cover of *Salicornia virginica* between MTL and MHW at Laumeister Tract was significantly greater than at comparable elevations at Faber Tract and the plants were significantly taller. The shorter plants at Faber Tract, especially at the higher elevations, could reduce the value of this area as habitat for species dependent on the upper tidal marsh areas, such as the federally endangered salt marsh harvest mouse (*Reithrodotomys raviventris*).

The importance of tidal flushing and the development of a full array of tidal channels, including the small first-order channels (tidal rivulets of some authors), to the biological functioning of the tidal marsh system is well documented. Simenstad and Thom (1996) surmise that development of the tidal slough channel system facilitates access to the intertidal marsh surface by fish and motile macroinvertebrates. Although they did not test whether fish usage of the marsh surface was a function of dendritic tidal slough channel development, they did suggest that development of a dendritic tidal slough channel system may be an indicator of marsh development (Simensted and Thom 1996).

Others have also addressed the importance of tidal flushing and the development of tidal rivulets on the marsh surface. McIvor and Odum (1988) suggest that the sinuosity of tidal creeks, channel depth and bank stability may affect the utilization of the tidal creeks and adjacent marsh surface by fish. Minello and Zimmerman (1983) and Rozas et al. (1988) discuss the importance of tidal rivulets and suggest that the tidal rivulets provide access to the marsh surface for juvenile fish and macroinvertebrates and refuge during low tide. Havens et al. (1995) emphasize the importance of tidal rivulets and recommend that tidal rivulets, as a component of marsh microtopography, be considered in the design of constructed wetlands. Tidal rivulets also increase the amount of edge in the marsh system and this "edge" habitat has been found to be important for fish and macroinvertebrates (e.g., Thomas et al. 1990, Rozas 1992, Baltz et al. 1993, Peterson and Turner 1994).

In north Bay, the presence of the slough channels (tidal rivulets of other authors), especially in the upper marsh plain, may have an affect on the presence of the federally endangered California clapper rail (*Rallus longirostris obsoletus*). Recent work on clapper rails in the north Bay suggests that the primary channels (rivulets) found in the upper marsh may be important to the existence of this species in a particular tidal marsh system (J. Collins, SFEI, personal communications). The clapper rail appears to preferentially nest in the upper marsh adjacent to the primary channels near the upper part of the tidal zone and use the primary channel as travel routes to the lower marsh where it forages. If this is the case, the lack of adequate development of a tidal slough channel system, especially the primary and secondary slough channels, could affect the ability of the tidal marsh to support clapper rails.

Construction of the "final" marsh plain elevation can also lead to a delay in attainment of functional equivalency. After approximately 20 years, the portion of Muzzi Marsh and Faber Tract that were above MHW when tidal flushing was restored, were still structurally different from nearby natural marshes. The delayed development of the tidal slough channel system at Muzzi Marsh and upper Faber Tract, resulting from overfilling the area with dredged materials, has resulted in greater cover by pickleweed but the plants are shorter, less robust. The lack of tidal flushing related to the poor development of a tidal

slough channel system in the constructed marshes is probably the major factor that has contributed to the observed differences in the structure of the marsh vegetation.

In the Gog-Le-Hi-Te estuarine wetland restoration project (Washington), the main tidal channels were dug during construction Simenstad and Thom (1996). Recent monitoring of the constructed tidal channels indicates that they are filling with sediments and that a dendritic tidal channel pattern is developing in the adjacent intertidal areas, including the marsh surface. Construction of the tidal channels may have slowed the attainment of a functional tidal slough channel system, thus delaying development of the desired functional attributes. As Simensted and Thom (1996) caution, the use of "ecotechnological fixes," such as the construction of tidal channels to facilitate tidal flushing, may not produce the same endpoints as natural, longer-term processes, such as allowing the final marsh surface to develop through natural sedimentary processes.

Results of this study and analysis of available information provides a basis for developing design criteria for future projects involving the use of dredged materials to create or restore tidal marsh habitat. Creating the proper physical conditions is key to developing the desired tidal marsh ecosystem. If the surface elevation of the created tidal marsh system is too high, development of slough channels will be slow and, in some cases, the slough channels may not develop. The resulting tidal marsh habitat would be restricted to high marsh dominated by *Salicornia virginica* with few other species being present. The resulting marsh system would be incomplete and have limited functional capacity relative to natural tidal marsh systems.

Conclusion

The major conclusion of our study is that the initial elevation of the dredged materials, when tidal activity is restored, is an extremely important determinant of slough channel density and morphology in constructed marshes. The presence of a well-developed tidal slough channel system is important for development of desired biological structure and related function at the constructed tidal marsh. This conclusion is based on the following findings of this study:

- Slough channels appear to form initially at elevations between MTL and MHW (~1.5 feet below MHHW) the reported approximate lower elevation range for *Spartina foliosa*.
- Slough channels either do not form naturally, or form very slowly, at elevations about 1.5 feet below MHHW.
- Slough channel morphology at natural tidal marshes differs from constructed marshes in the orientation of the slough channels, sinuosity and drainage density.
- There are significant differences in the structure of the tidal marsh vegetation (primarily *Spartina* and *S. virginica*) between the natural tidal marshes and constructed tidal marshes. These differences appear to be related to differences in the formation of a tidal slough channel network at the constructed tidal marshes.
- Plant height of *Spartina* and *S. virginica* are greater where the drainage density of slough channels is higher. At the two constructed tidal marshes, the plant heights are greater where there is a well-developed slough channel system compared to areas where marsh surface elevation has apparently prevented the formation of slough channels.

Acknowledgements

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Literature Cited

Atwater, B. F. 1979. History, landforms, and vegetation of the estuary's tidal marshes. In T.J. Comonos (ed.) San Francisco Bay, The Urbanized Estuary.

American Association for the Advancement of Science, Pacific Division, San Francisco, CA.

Baltz, D. M., C. Rakocinski, and J. W. Flegger. 1993. Microhabitat use by marsh edge fishes in a Louisiana estuary. Environmental Biology of Fishes 36: 109-126.

Cammen, L. M., E. D. Seneca, and B. J. Copeland. 1974. Animal colonization of salt marshes artificially established on dredge spoil. University of North Carolina, Chapel Hill, NC. Sea Grant Program Publication UNC-SG-74-15.

Craft, C. B., S. W. Broome, and E. D. Seneca. 1988. Soil nitrogen, phosphorus, and organic carbon in transplanted estuarine marshes. pp. 351-358 In D. D. Hook (ed.) The Ecology and Management of Wetlands. Timber Press.

England, A. S. and F. Nakaji. 1990. Reclaiming flooded river islands as wetlands and riparian habitat using dredged material. In Proceedings of Beneficial Uses of Dredged Material in the Western United States. U.S. Army Engineers Waterways Experiment Station, Vicksburg, MS.

Environmental Laboratory. 1978. Wetland habitat development with dredged material: engineering and plant propagation. Technical Report No. DS-78-16, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Faber, P. 1983. Marsh restoration with natural revegetation: A case study in San Francisco Bay. pp. 729-734 In O. T. Magoon (ed.) Coastal Zone '83. American Society of Civil Engineers, NY.

Faber, P. 1988. Current Status of the Muzzi Marsh: Report No. 5, Final Addendum. Prepared for the Golden Gate Bridge, Highway, and Transportation District.

Faber, P. M. 1990. Long-Term Observations of the Muzzi Marsh, Corte Madera, CA.

Gahagan & Bryant Associates, Bechtel Corp, ENTRIX Inc., and Philip Williams & Associates. 1994. A Review of the Physical and Biological Performance of Tidal Marshes Constructed with Dredged Materials in San Francisco Bay, CA. Prepared for Corps of Engineers, San Francisco District.

Harvey, H. T. 1970. Letter to Mr. Forest Warthman, BCDC Research Assistant, BCDC File.

Harvey, H. T., P. B. Williams, Madrone Associates, and San Francisco Bay Conservation and Development Commission. 1982. Guidelines for enhancement and restoration of diked historic baylands. A technical report prepared for San Francisco BCDC.

Havens, K. J., L. M. Varnell, and J. G. Bradshaw. 1995. An assessment of ecological conditions in a constructed tidal marsh and two natural reference tidal marshes in coastal Virginia. Ecological Engineering 4: 117-141.

Josselyn, M. 1983. The ecology of San Francisco Bay tidal marshes: A community profile. U.S. Fish and Wildlife Service, Biol. Serv. Proj. FWS/OBS-83/23.

Landin, M. C. 1990. Wetland restoration, protection, and establishment by beneficially using dredged material. Proceedings of American Society of Civil Engineers Water Forum '92, NY:114-118.

Landin, M. C. 1985. Long-term monitoring of habitat development sites, 1974-1984. Environmental Effects of Dredging. Information Exchange Bulletin D-85-3: 1-7.

Landin, M. C., J. Webb, and P. Knutson. 1989. Long-term monitoring of eleven Corps of Engineers habitat development field sites built of dredged material, 1974-1987. Dredging Operations Technical Support Program Technical Report D-89-1. U.S. Army Corps of Engineers Waterways Experimentation Station.

LaSalle, M. W., M. C. Landin, and J. G. Sims. 1991. Evaluation of the flora and fauna of a *Spartina alterniflora* marsh established on dredged material in Winyah Bay, SC. Wetlands 11:101-108.

Leopold, L. B. and J. P. Miller. 1956. Ephemeral streams–hydraulic factors and their relation to the drainage net. USGS Professional Paper 282-A.

McIvor, C. C. and W. E. Odum. 1988. Food, predation risk, and microhabitat selection in a marsh fish assemblage. Ecology 69:1341-1351.

Minello, T. J. and R. J. Zimmerman. 1983. Fish predation on juvenile brown shrimp, *Panaeus aztecus* (Ives): the effect of simulated *Spartina* structure on predation rates. J. Exp. Mar. Biol. Ecol. 72:211-231.

Mitsch, W. J. and R. F. Wilson. 1996. Improving the success of wetland creation and restoration with know-how, time and self-design. Ecological Applications 6:77-83.

National Research Council. 1992. Restoration of Aquatic Ecosystems. National Academy Press, Washington, D.C.

National Research Council. 1994. Restoring and Protecting Marine Habitat. National Academy Press, Washington, D.C.

Peterson, G. W. and R. E. Turner. 1994. The value of salt marsh edge vs. interior as a habitat for fish and decapod crustaceans in a Louisiana tidal marsh. Estuaries 17: 235-262.

Rozas, L. P. 1992. Comparison of nekton habitats associated with pipeline canals and natural canals in Louisiana marshes. Wetlands 12: 136-146.

Rozas, L. P., C. C. McIvor, and W. E. Odum. 1988. Intertidal rivulets and creek banks: corridors between tidal creeks and marshes. Mar. Ecol. Prog. Ser. 47: 303-307.

Seneca, E. D., S. W. Broome, W. W. Woodhouse, L. M. Cammen, and J. T. Lyon. 1976. Establishing *Spartina alterniflora* marsh in North Carolina. Environmental Conservation 3: 185-188.

Simenstad, C. A. and R. M. Thom. 1996. Functional equivalency trajectories of the restored Gog-Le-Hi-Te estuarine wetland. Ecological Applications 6: 38-56.

Straler, A. N. 1964. Quantitative geomorphology of drainage basins and channel networks. pp. 4-39 through 4-76 In V.T. Chow (ed.) Handbook of Applied Hydrology. McGraw Hill, NY.

Thomas, M. A., R. J. Zimmerman, and T. J. Minello. 1990. Abundance patterns of juvenile blue crabs (*Callinectes sapidus*) in nursery habitats of two Texas bays. Bulletin of Marine Science 46: 115-125.

Zedler, J. B. 1988. Salt marsh restoration: lessons from California. pp. 123-138 In J. Cairns (ed.) Rehabilitating Damaged Ecosystems. CRC Press, Boca Raton, FL.

Zedler, J. B. 1993. Canopy architecture of natural and planted cordgrass marshes: selecting habitat evaluation criteria. Ecological Applications 3: 123-138.

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Regional Restoration Planning for San Francisco Bay Salt Marshes: A Biogeographic Approach

Kevin MacKay[1]

Abstract: **Eighty-three wetland sites, located along the southwestern shoreline of San Francisco Bay, were studied to identify possible relationships between the diversity of resident wildlife species and the biogeographic characteristics of the sites. The distribution of five special status species: the California clapper rail, California black rail, Alameda song sparrow, salt marsh harvest mouse, and the salt marsh wandering shrew, were used as indicators of species diversity. National Wetland Inventory maps and aerial photographs were used to identify and quantify ten biogeographic characteristics for each site: shape, cover type, tidal regime, area, total perimeter, percentage of natural perimeter, percentage of unnatural perimeter, percentage of aquatic perimeter, degree of channelization, and isolation. Six of the ten characteristics, cover type ($r = 0.729$), tidal regime ($r = 0.86$), area ($r = 0.3$), total perimeter ($r = 0.383$), percentage of aquatic perimeter ($r = 0.664$), and degree of channelization ($r = 0.702$) were positively correlated with occurrences of the indicator species. The remaining four characteristics, shape ($r = -0.441$), percentage of natural perimeter ($r = .108$), percentage of unnatural perimeter ($r = -0.616$), and isolation ($r = -0.254$) were negatively correlated with indicator species occurrences. Overall, the relationship between the wetland sites and species diversity was best explained by a combination of characteristics ($r^2 = 0.631$) including cover type, tidal regime, total perimeter, and isolation. The final results of the study were used to develop a predictive model to guide the restoration of wetland habitat and the conservation of associated resident wildlife species in the San Francisco Bay Estuary.**

Introduction

Human development of the San Francisco Bay Estuary has significantly reduced the area of historic wetlands and led to the endangered or threatened status of certain wetland-dependent species. Before 1850, 1400 km^2 of freshwater marsh surrounded the confluence of the Sacramento and San Joaquin rivers, and another 800 km^2 of saltwater marsh fringed the Bays' shores. As population increased, tidal marshes were diked to create farmland, salt evaporation ponds, and later residential and industrial land. Reclamation of freshwater marshes in the Delta was essentially complete by the 1920's, but the filling and conversion of the Bay's saltwater marshes continued until the early the 1970's. Only 125 km^2 or five percent of the original 2200 km^2 of tidal marsh remains today (Nichols, et al. 1986).

While mitigation to compensate for these historic losses has taken place under federal and state permit policies for over 25 years, it has been implemented on a project-by-project basis, often ignoring the regional perspective. The legacy of this historic development and site-specific mitigation strategies is the fragmentation of once contiguous bands of tidal marsh habitat into patches scattered across the regional landscape. These patches or remnants support different vegetation types and tidal regimes, and vary greatly in size, shape, cover type, and proximity to other sites. Within the Estuary the conservation of regional biodiversity depends entirely upon the retention and maintenance of these remnants. Regulatory and planning agencies are therefore faced with the dual issues of determining whether the remnants have any practical conservation values, and if they do, of how they should be managed to restore or enhance these values (Saunders et al. 1991).

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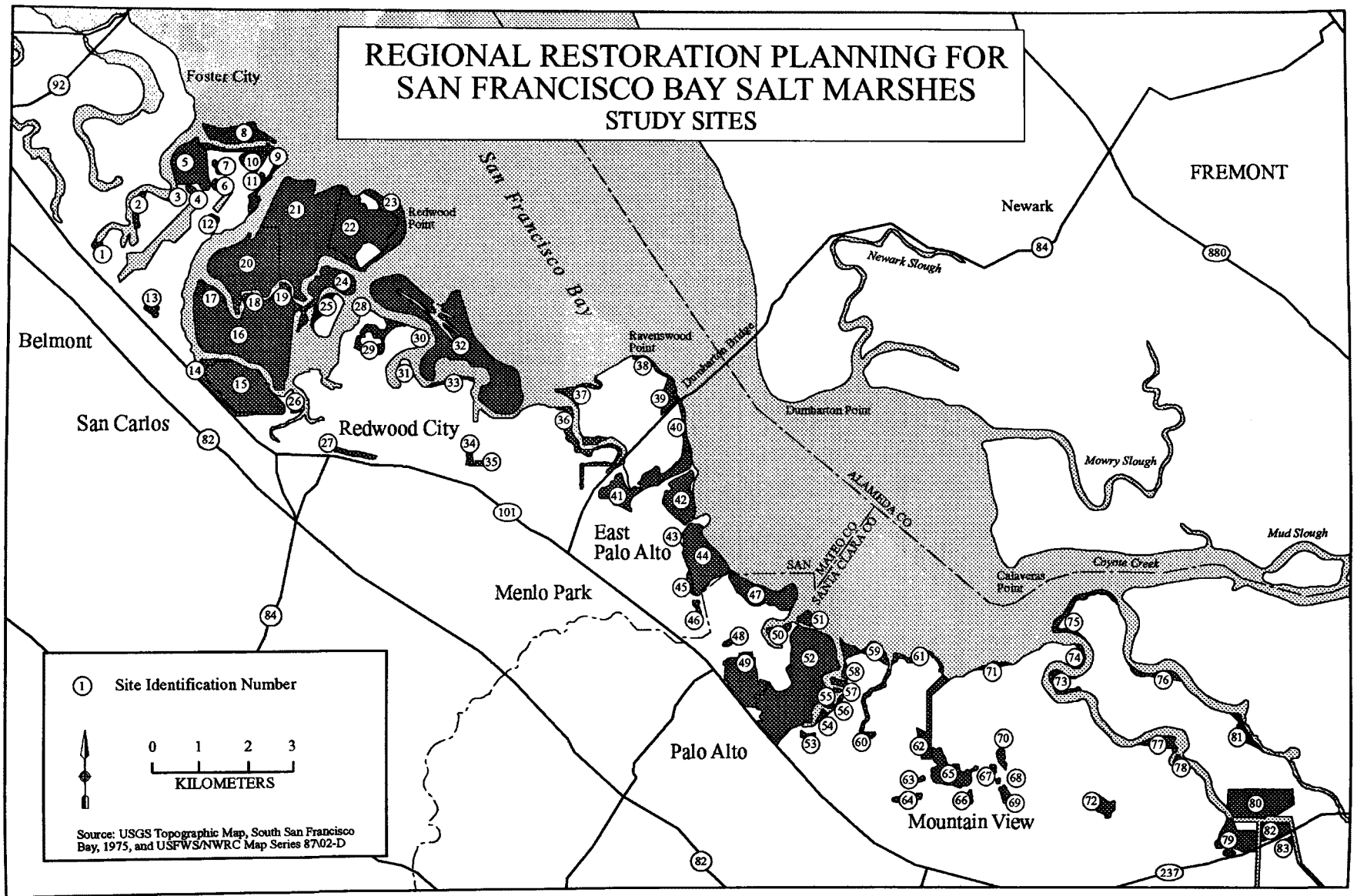

Figure 1. Map of the southwestern shoreline of San Francisco Bay showing the distribution of the 83 study sites.

The concept of remnants of natural vegetation as habitat islands, and the changes that result from the isolation of these islands have been the subject of considerable debate in the scientific literature (Saunders et al. 1991). This debate has centered primarily on the theory of island biogeography and its applicability to conservation planning (Preston 1962; MacArthur and Wilson 1967; Soule and Wilcox 1980; Wilcove 1987). In particular, the importance of biogeographic characteristics such as size, shape and proximity, as well as the spatial arrangement of fragments within the overall landscape matrix (Temple and Wilcox 1991). Although many studies have addressed the relationship between these factors and species diversity in forests, grasslands, chaparral and coastal sage scrub habitats, limited knowledge exists relative to their applicability to wetlands.

The primary objective of this study is to examine the influence of the biogeographic characteristics of individual salt marsh sites on the diversity of resident wildlife species. These characteristics include cover type, tidal regime, degree of channelization, site area, site perimeter, site shape, and isolation. The second objective is to use the data produced from this analysis to develop a predictive model that can be used to guide the regional restoration planning process for the San Francisco Bay Estuary.

Materials and Methods

Study Area

The general study area is composed of a 25-kilometer long band located along the southwestern shoreline of San Francisco Bay, from Belmont Slough, San Mateo County in the north, to Alviso Slough, Santa Clara County in the south (Figure 1). This area was selected because it encompassed a mosaic of wetland sites and land use types representative of the San Francisco Bay Estuary. In addition, the area is large enough to provide a regional context for assessing species diversity, and small enough that there are no pronounced differences in the wildlife and vegetation associated with the individual sites. Eighty-three smaller study sites, classified as tidal estuarine wetlands, or diked estuarine/seasonal wetlands, were selected within the general study area to assess the potential relationships between the biogeographic characteristics of the individual sites and the diversity of resident wildlife species.

Indicator Species

Since a survey of all resident wildlife species that occur in the study sites was outside the scope of this study, five species were selected as indicators of species diversity: the California clapper rail, California black rail, Alameda song sparrow, salt marsh wandering shrew, and salt marsh harvest mouse. These species were chosen because of their special legal status, as well as the general availability of information relevant to their ecological requirements. Species distribution data was obtained from the U.S. Fish and Wildlife Service (USFWS 1993, 1994), the California Department of Fish and Game's Natural Diversity Data Base, and other wildlife studies (Walton 1975, Posternak 1986, WESCO 1986, WRA 1994).

Sampling

Ten biogeographic variables were identified and quantified for each of the 83 study sites (Table 1). These variables are geographic, biotic, and hydrologic characteristics that potentially influence the biological diversity, or otherwise diminish the long-term viability of a wetland site. Jandel Scan/Image software was used in conjunction with the appropriate National Wetland Inventory maps to calculate the geographic characteristics of each site. Dominant vegetation types and hydrologic regimes were determined from recent aerial photographs (scale 1:12,000), and assigned relative point values based upon the ecological requirements of wetland-associated species.

Data Analysis

Statistical analyses were performed with Jandel Sigma Stat computer software. Spearman rank order correlations were used to determine possible relationships between the number of indicator species and the biogeographic variables of the 83 study sites, and multiple regression analysis to develop a predictive model for species diversity.

Results

Biogeographic Variables

The study sites varied greatly in size, shape, and perimeter (Table 2). The area of the largest and smallest sites were separated by more than two orders of magnitude. Only eight percent (7 of 83) of the sites exceeded 1 km^2 with the majority falling between the

Table 1. Biogeographic variables calculated for the 83 study sites.

Biogeographic Variable	*Description*
Shape Factor	Measure of site shape relative to a circle of equal area
Cover Type	Dominant vegetation type
Tidal Regime	Degree of tidal inundation
Area	Site area in square kilometers
Perimeter	Site perimeter in kilometers
Natural Perimeter	% of site perimeter surrounded by tidal salt marsh vegetation
Unnatural Perimeter	% of site perimeter site perimeter bordering urban development, salt ponds, roads, riprap, or dikes/levees
Aquatic Perimeter	% of site perimeter slough channels or the Bay
Channelization	% of site infiltrated by tidal channels
Isolation	Distance in kilometers to the nearest "source" site that contains one or more of the indicator species

Table 2. Descriptive statistics and results of the Spearman rank order correlation analysis.

Biogeographic Variable	*Mean/Mode*	*SE*	*r Value*	*p Value*
Shape Factor	0.42	0.22	-0.441	<0.0001
Cover Type	3.0	0.09	0.729	<0.0001
Tidal Regime	3.0	0.1	0.86	<0.0001
Area (km^2)	0.34	0.67	0.3	0.0059
Perimeter (km)	2.96	0.28	0.383	<0.0001
% Natural Perimeter	4.06	1.33	-0.108	0.3308
% Unnatural Perimeter	62.49	3.55	-0.616	<0.0001
% Aquatic Perimeter	32.87	3.54	0.664	<0.0001
Channelization (%)	17.3	2.04	0.702	<0.0001
Isolation (km)	0.35	0.05	-0.254	0.0206

mean value of 0.34 km^2 and the low value of 0.01 km^2. Site perimeters ranged from 0.38 to 13.33 km, with a mean value of 2.96 km. Of the three perimeter types measured, aquatic, natural and unnatural, the later occupied a greater percentage of overall perimeter (mean 62.5), with the other two types registering means of 32.87 and 4.06 percent respectively. Only 18 percent (15 of 83) of the sites had some amount of natural perimeter, and thus were contiguous with adjacent sites. Site shape was predominately linear with the mean shape factor equaling 0.42 (oval/linear). Only 4 percent of the sites (3 of 83) registered a shape factor above 0.75, roughly the equivalent of a round/irregular shape. The degree of isolation of the study sites ranged from 0.00 to 1.90 km, with the mean distance equaling 0.346 km.

Cover type and tidal regime were predominately coastal salt marsh with full tidal inundation. Almost two-thirds (60 percent, 50 of 83) of the sites were coastal salt marsh, 27 percent (23 of 83) upland, and the remaining 13 percent (10 of 83) a mixture of the two habitat types. Approximately 72 percent (60 of 83) of the sites are at least partially inundated by the daily tidal cycles. The degree of channelization within these sites varied from 5 to 50 percent, with a mean of 27.9 percent. The remaining 23 sites (28 percent) are completely diked off and only fill during periods of heavy precipitation.

Data Analysis

Spearman rank order correlation was used for a preliminary analysis to determine the relationships

between the individual biogeographic variables and species diversity. Six of the ten variables were positively correlated with increasing species diversity (Table 2). Four variables, shape, natural perimeter, unnatural perimeter and isolation were negatively correlated with species diversity. In all but one case, natural perimeter, these correlations were statistically significant ($p<0.05$). The r values for the individual correlation equations ranged from -0.616 to 0.860.

Multiple Regression Analysis. The relatively low r values derived from the Spearman rank order correlation analysis indicate that high species diversity is not entirely explained by any single regression model. As such, a best subsets regression analysis was used to select the combination of independent variables that best contributed to increasing species diversity. Adjusted r^2 values (r^2 value adjusted for the number of variables) ranged from 0.514 to 0.643, with the highest value associated with the combination of independent variables that included cover type, tidal regime, total perimeter, percentage of natural perimeter, percentage of unnatural perimeter, percentage of aquatic perimeter, and isolation.

However, as this analysis indicated redundancy or correlation between some of the biogeographic characteristics, variables with Variance Inflation Factors (VIF) higher than 4.0 were removed from the best-subset analysis. This final step removed the percentage of aquatic perimeter, percentage of unnatural perimeter, and percentage of natural perimeter variables, resulting in a model with an r^2 value of 0.631. As such, it appears that high species diversity in the study sites is correlated with a combination of independent variables including isolation, perimeter, tidal regime, and cover type.

Based upon this correlation, the four remaining variables were used to formulate a model to predict species diversity in salt marsh sites. This model uses multiple linear regression analysis to develop an equation that predicts potential species diversity based upon a site's degree of isolation, total perimeter, tidal regime, and cover type. The equation derived from the data obtained during this study is: Potential Species Diversity = -0.734 - (0.258 x isolation) + (0.153 x total perimeter) + (0.896 x tidal regime) - (0.214 x cover type).

This equation was used to formulate Potential Species Diversity Indices (PSDI) for the 83 sites examined in this study. PSDI scores ranged from -0.70 to 3.33, with a mean value of 0.975. The total PSDI scores were used to place the 83 study sites into four categories representing very high, high, moderate, and low potential species diversity (Figure 2). Five sites, or 6 percent of the total, fell within one standard deviation (SD) of the high value and were assigned a very high potential value. Twenty-three sites, or 28 percent of the total, fell between the first and second SD and were assigned high potential value. Twenty sites, or 24 percent, fell between the second and third SD and were assigned to the moderate value category. The remaining 35 sites (42 percent) fell below the third SD and were relegated to the low potential value category.

Discussion

Influence of Biogeographic Variables

The results from this study suggest that the ten biogeographic variables have varying levels of influence over the species diversity of the 83 study sites. Factors such as area and shape, prominent in previous studies of terrestrial systems, appear to have only slight positive correlation with species diversity in wetland sites. While others, such as perimeter, that had negative effects in previous studies have a significant positive correlation with greater species diversity in this study.

Site area had a slight positive correlation with high species diversity ($r = 0.30$). This figure was somewhat lower than the correlation between area and species diversity in studies of chaparral requiring bird species in California ($r = 0.55$), and obligate wetland birds in Maine ($r = 0.61$) (Soule et al. 1988, Gibbs et al. 1991), but was still higher than the correlation ($r = 0.22$) with facultative wetland birds in New York (Grover and Baldassare 1995). This difference can be explained by two factors. First, the loss of historic coastal marsh habitat in the South Bay has forced the indicator species to colonize even the smallest site. Secondly, previous studies (Power 1972, Johnson 1975, Western and Ssemakula 1981) concluded that larger sites support a greater diversity of habitat and topographic relief, and that these features are more significant than area in determining species diversity.

Thus, because of the relatively homogenous nature of salt marsh vegetation and the lack of site relief, larger study sites would not necessarily support a higher diversity of species.

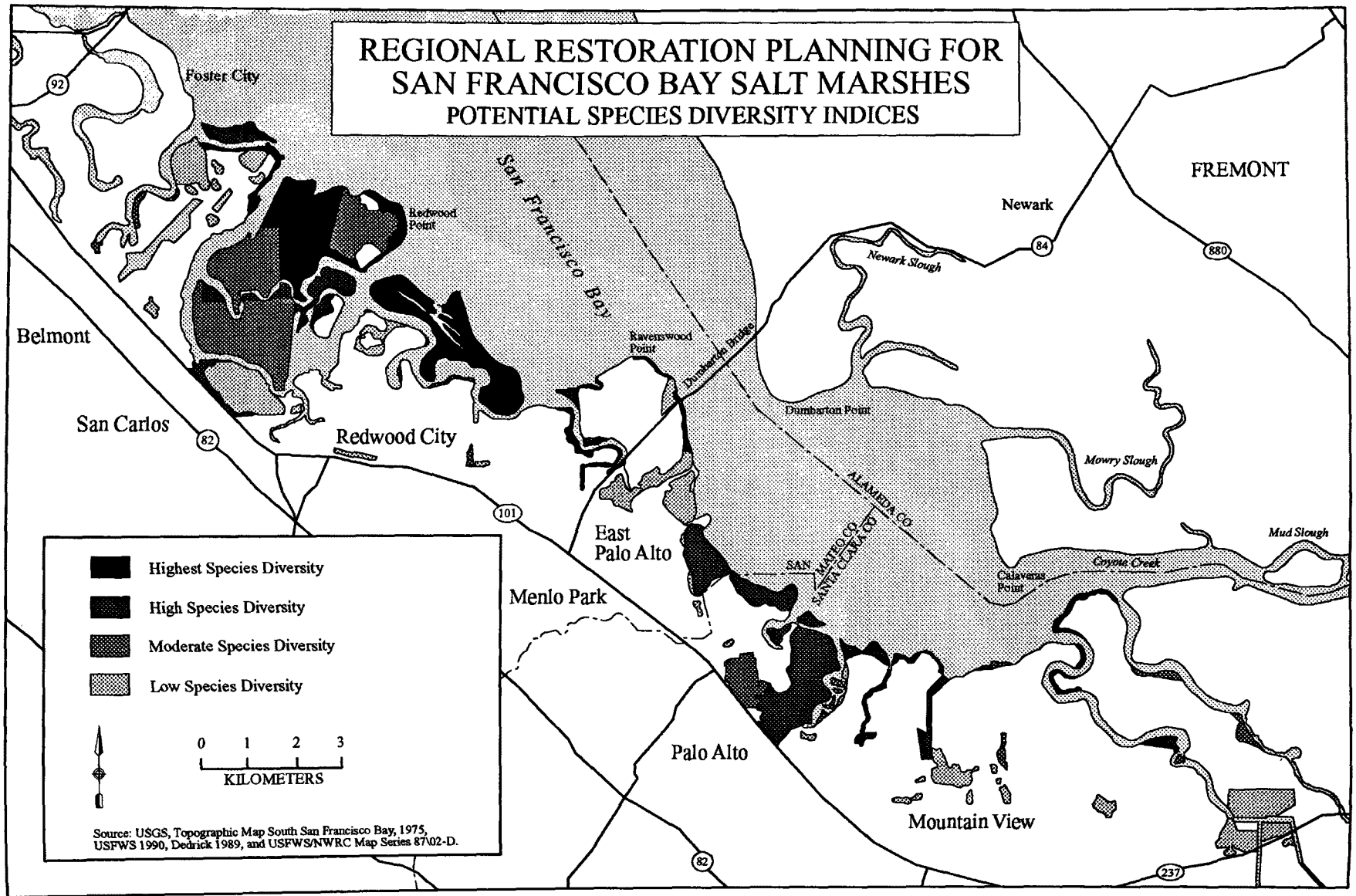

Figure 2. Map of the southwestern shoreline of San Francisco Bay showing the potential species diversity indices for the study sites.

The correlation between cover type (r = 0.729), tidal regime (r = 0.860), and channelization (r = 0.702) were consistent with the known habitat preferences of the indicator species. California clapper rails, California black rails, Alameda song sparrows, and salt marsh wandering shrews preferred channelized sites with coastal salt marsh vegetation, full tidal inundation, and a moderate to high level of channelization. Whereas, occurrences of salt marsh harvest mice extended to diked marshes with mixtures of upland and coastal salt marsh vegetative types. Geissel et al. (1988) found that salt marsh harvest mice act as a refugial species as they are forced to move into areas of lesser cover and higher salinity by superior competitors such as western harvest mice, house mice and California meadow voles. Salt marsh harvest mice are able to live in these areas because of their ability to tolerate high levels of salt in their diet (Shellhammer et al. 1988). For example, Zetterquist (1977) found that salt marsh harvest mice were the last species to be excluded from highly saline diked marshes.

The negative correlation between the shape factor (r = -0.44) and high species diversity suggests that the optimal site is linear in shape. This contradicts the guidelines provided by much of the scientific literature (Diamond 1976, Wilson and Willis 1975) that declares that a round shape is far superior to any other shape. This difference can be explained by the fact that these guidelines were reportably derived from island biogeographic theory rather than from empirical studies of species diversity. In fact, no research has demonstrated that round reserves hold more species than long thin ones (Shafer 1990). The results of this study suggest that linear sites may be more visible to immigrants, thus promoting the movement of species into the site from adjacent sites, slough channels, or bays.

This conclusion is borne out by the relationship between the perimeter variables and species diversity. Total perimeter (r = 0.383) and percent of site perimeter surrounded by water (r = 0.664) were both positively correlated with high species diversity. Whereas, the percent of site perimeter surrounded by natural vegetation (r = -0.108) and unnatural vegetation (r = -0.616) had negative correlations with high species diversity. This latter relationship is much more significant and is probably a result of detrimental "edge" effects. This finding is consistent with previous studies (Simberloff 1982, Noss 1983) that indicate that reserves with longer perimeters appear to be more susceptible to detrimental edge effects such as invasion by exotic plants, species predation and nest parasitism. A long unnatural perimeter made up of agricultural or urban land would increase the ability of opportunistic mesopredators such as Norway rats, red foxes, or feral cats to move into the study sites and feed on the eggs and young of the indicator species.

However, the positive correlations between total perimeter and percent surrounded by aquatic perimeter and high species diversity is quite dissimilar from the results of these earlier studies. This is probably because of a number of factors specific to wetland sites. First, physiographic factors such as high salinity and regular tidal inundation, that encourage colonization of native salt marsh vegetation, discourage the establishment of invasive plant species. Secondly, while exotic predators are able to move into sites from adjacent agricultural or urban land, a high percentage of aquatic perimeter would act as a barrier to these species. In addition, a long aquatic perimeter would promote the entry of the indicator species into the site by swimming or rafting. California clapper rails, salt marsh harvest mice, and salt marsh wandering shrews are all known to use these methods of dispersal.

Degree of isolation had a slight negative correlation (r = -0.254) with the diversity of indicator species. This relationship differs from that discovered by Grover and Baldassarre (1995) in New York wetlands, who found that wetland isolation was not a factor affecting species richness of wetland obligate birds. However, this correlation is consistent with the results of Soule et al. (1988), which reflected a slight benefit of patch proximity for small mammals such as rodents, rabbits and hares. This is further substantiated by the studies of Walton (1975), Shellhammer et al. (1982), and Bias (1994) which revealed the limited dispersal ability of the indicator species. For example, the mean dispersal distance of young Alameda song sparrows is estimated to be 185 meters, and salt marsh harvest mice have not been known to move distances greater than 30 meters on a regular basis (Bias 1994). As such, the average distance between sites of 350 meters (0.35 km), may be too great to allow easy dispersal between sites. This is especially true when the sites are separated by non-native habitat which may act as a complete barrier to dispersal.

Predictive Models

Conservation planners and biologists have often relied upon multiple regression analysis to obtain equations that can be used for predictive purposes. For example, Johnson and Simberloff (1974) used multiple regression to determine that area, latitude and distance were significant predictors of plant species number in the British Isles. More recently, Soule et al. (1988) developed a predictive model to anticipate the fate of chaparral-requiring birds in fragmented canyon habitats. The model developed from the multiple regression results of this study can be used to predict the relative species diversity of individual salt marsh habitat sites based upon a series of habitat characteristics. The data derived from this model can be extremely valuable as regulatory agencies are often forced to rely upon inadequate or preliminary species census data; a problem that has been identified as one of the major pitfalls of conservation planning (Baughman and Murphy 1990).

The predictive model can also be used to predict potential increases in species diversity that would result from restoration or enhancement, and to link these activities to a landscape-based planning procedure. Sites that might be suitable for restoration can be re-assessed for species diversity based upon changes to characteristics such as size, shape and accessibility. For example, the restoration of full tidal inundation to study site 52 would result in a 57 percent increase in the PSDI for that site, elevating it one level to the very high species diversity category. In addition, PSDI scores could be used to identify sites with lower relative values (Figure 2) that might be suitable for development. The loss of these sites, in turn, would be mitigated through a regional approach that would enlarge or connect existing sites currently identified as supporting high species diversity. These connections would facilitate species movement between the populated sites, encouraging recolonization of sites following local extinctions, and allowing genetic enrichment of existing populations of wetland-associated species.

Although predictive models are somewhat coarse, they do provide a statistical tool to assess the impacts of habitat fragmentation and to evaluate potential conservation and restoration strategies developed to ameliorate these effects. Moreover, the advent of statistical software packages make these models relatively quick and easy to use, and the necessary data can often be obtained from existing aerial photographs or National Wetland Inventory maps. However, some caution must be emphasized when using these models as they only provide a relative measure of species diversity, not a comprehensive evaluation of wetland performance and function.

Conclusions

In recent years, state and federal regulatory agencies have rejected isolated, site-by-site mitigation strategies in favor of regional management plans that strive to protect critical habitat functions and preserve the overall biodiversity of wetland ecosystems. However, the development and implementation of these plans has been a difficult process due to a lack of agreement on consistent criteria for assessing relative wetland values, and identifying sites suitable for restoration and enhancement (Rumrill and Cornu 1995). As a result, many of the recent plans have been based upon proven strategies that emphasize functional assessments of internal wetland characteristics as indicators of conservation value. This failure to recognize the importance of landscape level processes upon individual sites can result in the failure of the regional planning process, and the eventual extinction of threatened and endangered wetland-associated species. Future planning strategies must place a greater emphasis on site location and on habitat characteristics such as size, shape, and proximity in order to ensure that restored sites are successfully integrated into the regional wetland mosaic. While science is still decades away from discovering and understanding all of the major components that govern landscape level process in wetland ecosystems, the results of this study provide a basis for a series of guidelines for restoring wildlife and habitat values to the San Francisco Bay Estuary:

1. Both large and small sites should be preserved.

Although the results of this study demonstrate that size of a site is not strongly correlated with high species diversity, large sites should be preserved, as they generally support a greater number of individuals of a particular species. These larger populations enable the species to survive random environmental events (environmental stochasticity), random variations in birth and death rates (demographic stochasticity), and random changes in genetic composition (genetic stochasticity). Small sites should also be preserved as they seem to play a greater role in population dynamics than indicated by their size. These sites may function to increase the number of sources of potential colonists for sites that have recently

undergone local extinction. Also, small wetlands may act as stepping stones that decrease inter-site distance, and facilitate the dispersal of individuals within the regional wetland mosaic. This would increase the probability of site colonization and lead to a greater proportion of sites having extant populations at any one time. This, in turn, would increase the likelihood of metapopulation persistence over time (Gibbs 1993).

2. Long linear sites are better than round ones.

Long linear sites were positively correlated with high species diversity, possibly because they are more visible to immigrants, thus promoting the colonization of the site by species from adjacent areas. This is particularly important in sites that have long aquatic perimeters, as this characteristic would enable species such as California clapper rails, salt marsh harvest mice, and salt marsh wandering to enter the site by swimming or rafting. In addition, sites with long aquatic perimeters would support a greater percentage of low marsh vegetation which would increase the amount of potential habitat available for California clapper rails.

3. Sites should be surrounded by buffer zones.

The results of the study found that the percent of site perimeter surrounded by unnatural vegetation types, including agricultural and urban lands, was negatively correlated with high species diversity. This is probably a direct result of detrimental "edge" effects related to predation by opportunistic mesopredators. While not specifically addressed in this study the provision of buffer zones around sites would ameliorate this problem by filtering out exotic species such Norway rats, red foxes, and feral cats. In addition, buffer zones enlarge the effective size of the site and provide critical refuge areas during periods of extreme high tide.

4. Sites should be located close together.

The results of this study show a slight negative correlation between the isolation of a site and species diversity. As the distance between remnant patches increases, the probability of successful dispersal decreases, resulting in a decreased species diversity in the more isolated sites. In addition, endangered populations are more likely to be "rescued," by dispersing individuals from other sites, if they are in close proximity to each other (Brown and Kodric-Brown 1977). This movement between sites can help prevent local extirpation caused by demographic stochasticity, and the loss of genetic variability (Wilcove 1987).

5. Sites should be connected by corridors.

Interpopulation dispersal, as mentioned above, is important for regional species persistence. In cases where the intersite distance is greater than the dispersal abilities of resident species, habitat corridors should be developed between sites. In addition to providing habitat for plant and animal species, these corridors can serve as conduits for intersite movement.

6. Sites should support a full range of salt marsh vegetation and be subject to full tidal inundation.

Both cover type and tidal regime were correlated with increased species diversity. All of the indicator species occurrences, except the salt harvest mouse, were recorded in sites that supported coastal salt marsh vegetation and were subject to full tidal inundation. In addition, while diked marshes with mixed vegetation do support populations of salt marsh harvest mice, the species is at a competitive disadvantage in these areas and is often relegated to the poorest habitat patches.

7. Sites should be channelized.

The correlation between channelization and species diversity was consistent with the known habitat preferences of the indicator species. California clapper rails, California Black Rails, Alameda song sparrows, and salt marsh wandering shrews preferred sites with a moderate to high level of channelization. This is probably because their primary food sources consist of aquatic invertebrate species such as ribbed mussels and yellow shore crabs. As salt marsh harvest mice feed primarily on seeds and terrestrial insects, this characteristic seems to be less critical to their survival. However, they have been known to use channels as dispersal corridors between sites.

8. Proposed projects should be planned and reviewed from a regional perspective.

The review of projects on an individual basis ignores the regional distribution of resident species. For example, the relative isolation of plants and animals in San Francisco Bay salt marsh sites makes it highly unlikely that these sites will be recolonized following local extinctions. Therefore, the replacement of a developed marsh with a newly created or enhanced one in a different location would be of little value to a localized species that is unable to reach the new site. The recolonization of sites could be encouraged, rather than inhibited, if compensatory replacement

mitigation policies were changed to emphasize regional rather than site specific goals. For example, wetland development could be mitigated through an off-site banking approach that enlarges or connects existing sites that currently support high species diversity. These connections would facilitate species movement between the populated sites and increase the overall viability of the regional populations. The creation or restoration of upland buffer zones could also be used to mitigate project development.

Literature Cited

Baughman, J. F., and D. D. Murphy. 1993. Beware of snapshots at the bottleneck—temporal considerations in conservation planning. Endangered Species Update, 7, 8, and 9.

Bias, M. A. 1994. Ecology of the salt marsh harvest mouse in San Pablo Bay. Doctoral Dissertation, University of California Berkeley, Berkeley, CA.

Brown, J. H., and A. Kodric-Brown. 1977. Turnover rates in insular biogeography: effect of immigration on extinction. Ecology 58:445-449.

Diamond, J. M., and R. M. May. 1976. Island biogeography and the design of nature reserves. In R.M. May (ed.), Theoretical Ecology: Principles and Applications, pp. 163-186. W.B. Saunders, Philadelphia, PA.

Geissel, W., H. Shellhammer, and H. T. Harvey. 1988. Ecology of the salt marsh harvest mouse in a diked salt marsh. Journal of Mammalogy 69:696-703.

Gibbs, J. P., J. R. Longcore, D. G. McAuley, and J. K. Ringelman. 1991. Use of wetland habitats by selected nongame waterbirds in Maine. U.S. Fish and Wildlife Service, Fish and Wildlife Research 9.

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

Grover, A. M., and G. A. Baldassare. 1995. Bird species richness within beaver ponds in south-central New York. Wetlands 15:108-118.

Johnson, M. P., and D. S. Simberloff. 1974. Environmental determinants of island species numbers in the British Isles. J. Biogeography 1:149-154.

Johnson, N. K. 1975. Controls of number of bird species on montane islands in the Great Basin. Evolution 29:545-567.

MacArthur, R. H., and E. O. Wilson. 1967. The theory of island biogeography. Princeton University Press, Princeton, NJ.

Nichols, F. H., J. E. Cloern, S. N. Luoma, and D. H. Peterson. 1986. The modification of an estuary. Science 231:567-573.

Noss, R. F. 1983. A regional landscape approach to maintain biodiversity. Bioscience 33:700-706.

Posternak, L. D. 1986. A general study of birds using three wetland areas of shoreline at Mountain View. City of Mountain View, Mountain View, CA.

Power, D. M. 1972. Number of bird species on the California islands. Evolution 26:451-463.

Preston, F. W. 1962. The canonical distribution of commonness and rarity. Ecology 6:88-97.

Rumrill, S. S., and C. E. Cornu. 1995. South Slough coastal watershed restoration. Restoration and Management Notes 13:53-57.

Saunders, D. A., R. J. Hobbs, and C. R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5:18-28.

Shafer, C. L. 1990. Nature reserves: island theory and conservation practice. Smithsonian Institution Press, Washington, DC.

Shellhammer, H., R. Jackson, W. Davilla, A. M. Gilroy, H. T. Harvey, and L. Simons. 1982. Habitat preferences of salt marsh harvest mice. Wasmann Journal of Biology 40:102-114.

Shellhammer, H., R. Duke, H. T. Harvey, V. Jennings, V. Johnson, and M. Newcomer. 1988. Salt marsh harvest mice in the diked salt marshes of southern San Francisco Bay. Wasmann Journal of Biology 46:89-103.

Simberloff, D. S., and L. G. Abele. 1982. Refuge design and island biogeographic theory: effects of fragmentation. American Naturalist 120: 41-50.

Soule, M. E., D. T. Bolger, A. C. Roberts, R. Sauvajot, J. Wright, M. Sorice, and S. Hill. 1988. Reconstructed dynamics of chaparral-requiring birds in urban habitat islands. Conservation Biology 2:75-92.

Soule, M. E., and B. A. Wilcox. 1980. Conservation biology: an ecological-evolutionary perspective. Sunderland Associates, Sunderland, MA.

Temple, Stanley A., and Bruce A. Wilcox. 1991. Introduction: predicting effects of habitat patchiness

and fragmentation. pp. 261-262 In D. A. Saunders, R. J. Hobbs, and P. R. Ehrlich (Eds.), Nature Conservation 2: The Role of Corridors. Surrey, Beatty & Sons, NSW, Australia.

Terborgh, J. 1976. Island biogeography and conservation: strategy and limitations. Science 193:1029-1030.

U.S. Fish and Wildlife Service (USFWS). 1993. Unpublished Sensitive Species distribution data.

U.S. Fish and Wildlife Service (USFWS). 1994. Unpublished Sensitive Species distribution data.

Vickery, P. D., M. L. Hunter, and S. M. Melvin. 1994. Effects of habitat area on the distribution of grassland birds in Maine. Conservation Biology 8:1087-1097.

Walton, B. J. 1975. The status of the salt marsh song sparrows of the San Francisco Bay system, 1974-1975. M.S. Thesis, San Jose State University, San Jose, CA.

Western Ecological Services Company (WESCO). 1986. A Review of the Population Status of the Salt Marsh Wandering Shrew. U.S. Fish and Wildlife Service, Sacramento, CA.

Western, D. and J. Semakula. 1981. The future of the savannah ecosystems: ecological islands of faunal enclaves? Aft. J. Ecology 19:7-19.

Wetlands Research Associates (WRA). 1994. Draft Environmental Assessment: Cargil Salt Maintenance Activities, Permit Application No. 4-93, SCH# 94023030. San Francisco Bay Conservation and Development Commission, San Francisco, CA.

Wilcove, D. S. 1987. From fragmentation to extinction. Natural Areas Journal 7:23-29.

Wilson, E. O., and E. O. Willis. 1975. Applied biogeography. pp. 522-534 In M. L. Cody and J. M. Diamond (eds.), Ecology and Evolution of Communities. Belknap Press of Harvard University, Cambridge, MA.

Zetterquist, D. K. 1977. The salt marsh harvest mouse in marginal habitats. Wasmann Journal of Biology 35:68-76.

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Habitat Restoration in an Urban Setting: Integrating Desert Riparian Habitat, Groundwater Recharge, and Community Participation

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Abstract: **Desert riparian habitat has been scarce and of great importance to wildlife in the southwest United States. Because the amount and quality of this habitat type has been greatly reduced from historic levels, restoring desert riparian habitat is a high priority in Arizona. This presentation describes a plan that is being implemented by the Town of Gilbert, Arizona, to establish riparian habitat and provide other wildlife and public use benefits at the town's 75-acre groundwater recharge pond complex. The facility is located in an urban area and uses treated water from the town's wastewater reclamation plant to help recharge the local groundwater. The project is funded primarily through two Arizona Department of Game and Fish Heritage Grants and has already won two state environmental and planning awards. The project has been largely implemented by volunteers from the local community. Design features of the project include establishment of riparian vegetation along pond margins, marsh vegetation in a permanent pond, and upland vegetation on pond slopes; wildlife nesting structures; interpretive exhibits; and blinds for viewing wildlife. Jones & Stokes Associates, with assistance from Wild Seed of Tempe, Arizona, assisted the town with preparing the grant applications, developing the plan and details for implementation, collecting and installing cuttings and seeds, and implementing the project using community volunteers. The project successfully integrates enhancement of wildlife and public use values with the operation of a groundwater recharge facility and could serve as a model for other similar urban wildlife habitat restoration projects.**

Overview

"We need nature as much in the city as in the countryside....It is not a choice of either the city or the countryside: both are essential, but today it is nature, beleaguered in the country, too scarce in the city which has become precious. (McHarg 1969)"

Riparian habitat in Arizona is a scarce and precious resource that is extremely important to wildlife, as well as to people. Particularly in urban areas, riparian habitat is scarce. What does exist often is of low value for wildlife. The Town of Gilbert, Arizona, has realized a way to enhance or create riparian habitat within its rapidly developing urban boundaries while simultaneously improving the community and the environmental education opportunities for people.

The town developed its wastewater recharge ponds to provide the following benefits for wildlife, people, and the community:

- Open water and riparian vegetation that provides resting and foraging habitat for wildlife,
- Opportunities for people to closely view diverse wildlife in an urban setting, and
- Groundwater recharge and water planning to ensure available water resources for a growing community.

Using a readily available resource, reclaimed wastewater, the town integrated these benefits into construction and operation of groundwater recharge ponds.

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Groundwater Conservation Efforts and Benefits

The town's growth and location in the northern Sonora Desert had previously challenged decision makers to develop plans for efficient use of groundwater and renewable water resources. The answer to this challenge was to make better use of reclaimed water, and the town's leaders developed a policy of 100 percent reuse of water. In addition, under state legislation (1980 Groundwater Management Act [ARS 45-564.A.2]), the town measures and reports the volume of water recharged and recovered and receives credits for reclaimed water in the area where the effects of recharge can be detected and measured. To ensure the groundwater credits, the town has constructed 12 recharge ponds on 75 acres and a deep well that recovers water for potable use. Water reclamation has helped the town achieve the state's groundwater management goals for individual municipalities and provided a safe resource for current and future water demand (Frost et al. 1993).

An added benefit of water use planning has been the enhancement of urban habitat at the recharge ponds. After the original six ponds were constructed on 38 acres in 1989, wildlife began using the open water habitat for resting and foraging. A benefit to the town was the resulting wildlife viewing opportunities at the recharge ponds. The Sun Circle Trail along the north edge of the pond complex provided a viewing site used by local Audubon Society members. The viewing area soon attracted many other birdwatchers. The absence of vegetation around the ponds, however, limited their habitat value and the diversity of species that could use the pond area, which was expanded to six more ponds on 37 acres in 1993.

Plans for Site Improvements

In 1993, the town took an important step to improve the resource values of the pond area by seeking a Heritage Fund grant from the Arizona Game and Fish Department. Jones & Stokes Associates (JSA) assisted the town in identifying opportunities to enhance the habitat value of the ponds and provide interpretive facilities for the community. JSA prepared the grant application, which proposed that the town and JSA work as a team to design and construct numerous wildlife habitat and public use improvements. JSA would design the urban wildlife enhancement plan and interpretive features and assist the town in supervising volunteers during construction. The town would coordinate volunteers for facilities construction and supervise overall plan implementation. After award of the initial grant in 1993, JSA, with assistance from Wild Seed, Inc., a local native seed and revegetation consultant, began refining the design concepts for wildlife habitat enhancement and opportunities for public use and environmental education at the new ponds.

Design Concepts

The town was awarded two grants for habitat improvements. The first, awarded in 1993 for the west half of the pond complex, included establishing riparian, upland, and marsh vegetation, and designing and installing bird nest and bat roost boxes, creating viewing blinds, designing interpretive exhibits, and constructing a viewing ramada. The second, awarded in 1994, provided funding for expanding the habitat and public use improvements to the six original ponds on the east half of the pond complex by planting a greater diversity of riparian and upland species, planting an interpretive garden around the viewing ramada, and developing two additional interpretive exhibits.

Maximizing Resource Opportunities

As part of the design concept planning, existing resources were assessed and utilized. The town had three valuable resources: reclaimed water, existing wildlife, and a central location.

The primary resource was the availability of reclaimed water, which provided an excellent opportunity to establish riparian vegetation around the recharge ponds. Although the water source is managed, it is a valuable resource in the Desert Southwest. Riparian vegetation occupies less than 1 percent of the Arizona landscape (Brown 1982); however, it supports the densest and most diverse wildlife communities in the state. Its diverse plant species, vegetative growth forms, and microclimate conditions (e.g., increased humidity and lower temperature) provide a variety of important foods and habitat conditions for wildlife (Warner 1979). Riparian vegetation provides nesting and foraging habitat for resident wildlife and important refuge for migratory species. Approximately 75 percent of Arizona's wildlife, including fish, amphibians, insects, birds, and mammals, depends on a riparian area for some phase of its life cycle (Tellman 1992).

The project site is on the Mohall-Contine soil association, which consists of well-drained, deep soils on old alluvian fans. The area lies between the floodplains of the Salt River and the Queen Creek Wash. Because the permeability of the soil is slow, the ponds are periodically disked to improve permeability. Historic vegetation was probably creosotebush, saltbush, mesquite, cactus, and annual grasses. The historic land use was predominantly irrigated agriculture using an expansive system of canals originally developed by the Hohokam people native to the region. Crops planted more recently include alfalfa, cotton, vegetables, and citrus.

Initially the ponds were relatively barren of vegetation except for a few weedy grass species. Nonetheless, the ponds provide water, a critical habitat component that attracts numerous wildlife species. Maricopa Audubon Society members have observed 140 bird species at the ponds and on agricultural lands surrounding the project site. The ponds provide resting opportunities and limited foraging opportunities for various species; because of a lack of vegetation, however, the site does not have the cover and shelter habitat that wildlife need to occupy the site on a regular basis.

The recharge pond complex is completely surrounded by a chain-link fence, and public access within the fenced area is prohibited. The public does have visual/pedestrian access to the ponds along the north boundary via Maricopa County's Sun Circle Trail, a more than 200-mile, regional multipurpose trail. The pond complex is centrally located and accessible to town residents and Gilbert schools.

Design Elements and Construction Methods

In planning the new ponds, the Town of Gilbert recognized an opportunity to incorporate features that would improve their value for both wildlife and people. The goals of the project were to enhance wildlife habitat and provide environmental education and wildlife viewing opportunities for town residents. These goals could be achieved by establishing diverse vegetation types on the project site to ensure the following results:

- Encourage and maximize wildlife use,
- Encourage wildlife occupancy of the site by providing structural habitat, and
- Accommodate public use and enjoyment by providing additional viewing and interpretive facilities.

As a result of the Heritage grant, more extensive design concepts were developed to enhance the habitat value of the ponds and manage them for wildlife, as well as for recharging groundwater. Features of the new ponds (west half) that address these goals include curvilinear embankments to improve aesthetics and create more edge habitat, a permanent pond, and an elevated viewing area for the public.

West Pond Habitat Enhancements

Three vegetation types were established at the west ponds, including cottonwood and willow thicket habitat, emergent marsh habitat, and desert herbaceous and scrub habitat. Establishing this vegetation took advantage of the existing resources on the site (water, soil, wildlife) and met the goals of the project (increased habitat, improved education opportunities, and community involvement). Establishing this vegetation also met secondary considerations about the availability of material, cost, and appropriateness for wildlife species already using the site or expected to use the site.

The cottonwood and willow thicket habitat established at the west ponds included Fremont cottonwood, Goodding's willow, seep willow, and coyote willow. Cuttings of these species had been collected from the area by volunteers. The cuttings were placed in wet cold storage for several weeks until the next volunteer activity planting.

The cuttings were placed at the toe of the pond embankment slopes and arranged in clusters to maximize shade, density, canopy structure, habitat value, edge effect, and diversity. The planting clusters provided 25 percent coverage of the total area of the inside slopes of the ponds; this percent coverage was intended to provide essential habitat components and open corridors for wildlife viewing. The clusters were located to minimize interference with maintenance activities requiring vehicular access, and the gaps between planting clusters were designed to meet the needs of ducks such as pintails, northern shovelers, green-winged teals, and various diving ducks (ruddy ducks) that use a long, low trajectory for landing and takeoffs. Plantings were increased along the north boundary to minimize disturbance of wildlife by partially screening from view the people on the Sun Circle Trail.

The emergent marsh habitat was planted in a small, permanent pond in one comer of a larger pond. Marsh species, including sedge, spike rush, rush, tule, and arrow weed, were collected and planted as tubers by volunteers. The emergent marsh habitat provides nesting and foraging habitat for waterfowl, grebes, rails, and wetland songbirds.

Desert upland herbaceous and scrub habitat was established along the top of the pond embankments where the only available water source is rainfall. Volunteers broadcast seed, with direction from Wild Seed, Inc., along the top third of the embankments and gently raked the seed into the top 1/4-inch of the soil. Wood chip mulch was donated by a local tree service, and a light coverage was distributed along the seeded banks by volunteers. The species composition included a diverse mixture of native grasses, forbs, and shrubs to provide wildlife cover and food and to control erosion. Unfortunately the pond embankments were compacted to 95 percent during pond construction and only the most tenacious desert seeds germinated, predominantly saltbush and a few desert sunflower.

East Pond Habitat Enhancements

The same planting concept was used for the east ponds during the second Heritage grant with the addition of two desert riparian plant associations: Interior Southwest Riparian Deciduous Forest and Woodland and Sonora Riparian and Oasis Forest Mesquite Series. These associations are identified as sensitive by the Arizona Department of Game and Fish and were included to create more diverse wildlife habitat and vegetative species composition and expand the educational opportunities of the site. Both vegetation groups were planted midslope around the ponds from 1-gallon containers and are provided with drip irrigation until they become established. Volunteers planted the vegetation in holes previously drilled by the town's staff. The Interior Southwest Riparian Deciduous Forest and Woodland is dominated by Arizona sycamore, but also includes desert hackberry, wolfberry, mesquite, Mexican elderberry, graythorn, desert willow, and hackberry. The Sonora Riparian and Oasis Forest is mesquite-dominant and includes desert hackberry, wolfberry, and graythorn. (Brown 1982).

Wildlife and Public Use Structures

Food, cover, and shelter are important habitat requirements for wildlife. The diversity of riparian and upland vegetation will help meet these requirements by providing fruit and seed for forage, protection from predators, protection from adverse weather conditions, and habitat for breeding. The establishment of vegetation is one of the habitat enhancement goals for the project; however, it will be years before the vegetation is mature enough to provide tree cavities and high branches for adequate cover and shelter for certain species of wildlife. In the interim, human-made structures were designed to provide additional wildlife cover and shelter habitat. Nest and roost boxes for American kestrels, wood ducks, black-bellied whistling ducks, tree swallows, black phoebes, and bats were designed and installed to supplement the riparian vegetation. The structures were constructed by the Maricopa Audubon Society volunteers and placed on poles around the ponds by the town staff. It is possible that even the boxes may not be utilized until surrounding vegetation provides suitable environmental conditions to attract wildlife.

A number of public use improvements were also constructed by volunteers. Boy Scouts constructed benches at the viewing blinds along the Sun Circle Trail, and a local architect donated designs for the ramada. The town also constructed an elevated observation area and access ramp up to the ramada on the southwest corner of the project site. The viewing ramada is the main public use area; other facilities include public parking, benches, and a screened viewing area. Seven interpretive exhibits created by JSA explain the wildlife, riparian habitat, and groundwater recharge system at the ponds.

Town Commitment and Citizen Participation

The Town of Gilbert was responsible for publicizing the planting days and soliciting volunteers and organizations to participate. For the first planting effort, more than 50 volunteers donated their time on a Saturday to plant, dig, or spread seed. They came from the Audubon Society, the Rotary Club, the Boy Scouts, the local high school ecology club, and the general public. The consultants organized and instructed the volunteers about the day's activities. For the subsequent activities, the Audubon Society constructed roost and nest boxes and Eagle Scouts

constructed benches, dug planting holes, and planted seeds for the interpretive gardens.

The future monitoring of the ponds will be conducted by a local junior college biology department in cooperation with the high school and elementary schools. The monitoring activities will be used to teach students of all levels about soil, water, vegetation, and wildlife.

Conclusions

As the vegetation around the ponds matures, its value to wildlife will increase and expand the variety of wildlife species attracted to the ponds. It will also offer residents and students visiting the ponds on a regular basis a unique opportunity to observe changes in plant growth and habitat value through time.

The ponds provide valuable wildlife habitat in a desert environment, which will continue to increase in value as the vegetation matures and as long as the ponds are maintained. As of fall 1995, 11 species of waterfowl were routinely identified at the ponds, including black-bellied whistling duck, great blue heron, northern shoveler, ruddy duck, and cinnamon teal, and a marsh wren has moved into the emergent marsh habitat. The ponds continue to provide the Town of Gilbert with economic value in the form of water credits for groundwater recharge. They also provide the town with a unique civic and community resource, providing passive recreation opportunities for the community; educational opportunities for local students and adults; and, perhaps most importantly, a civic project people of all ages can be actively involved in through volunteer plantings, monitoring, and maintenance.

The project received public recognition on two occasions during 1994. It received the Governor's Pride in Arizona Award in the Environmental Leadership category and the Best Project Award from the Arizona Planning Association, recognizing the ponds as a site-specific project using original ideas to address local, regional, or national concerns.

The project successfully integrates enhancement of wildlife and public use values with the operation of a groundwater recharge facility in an urban setting and could serve as a model for other, similar urban wildlife habitat projects. Uniting economic values with wildlife habitat and community values ensures human and natural benefits.

Literature Cited

Brown, D. E. (ed.). 1982. Biotic communities of the American southwest - United States and Mexico. (Desert Plants, Volume 4, No. 1-4.) The University of Arizona. Prepared for the Boyce Thompson Southwestern Arboretum, Superior, AZ.

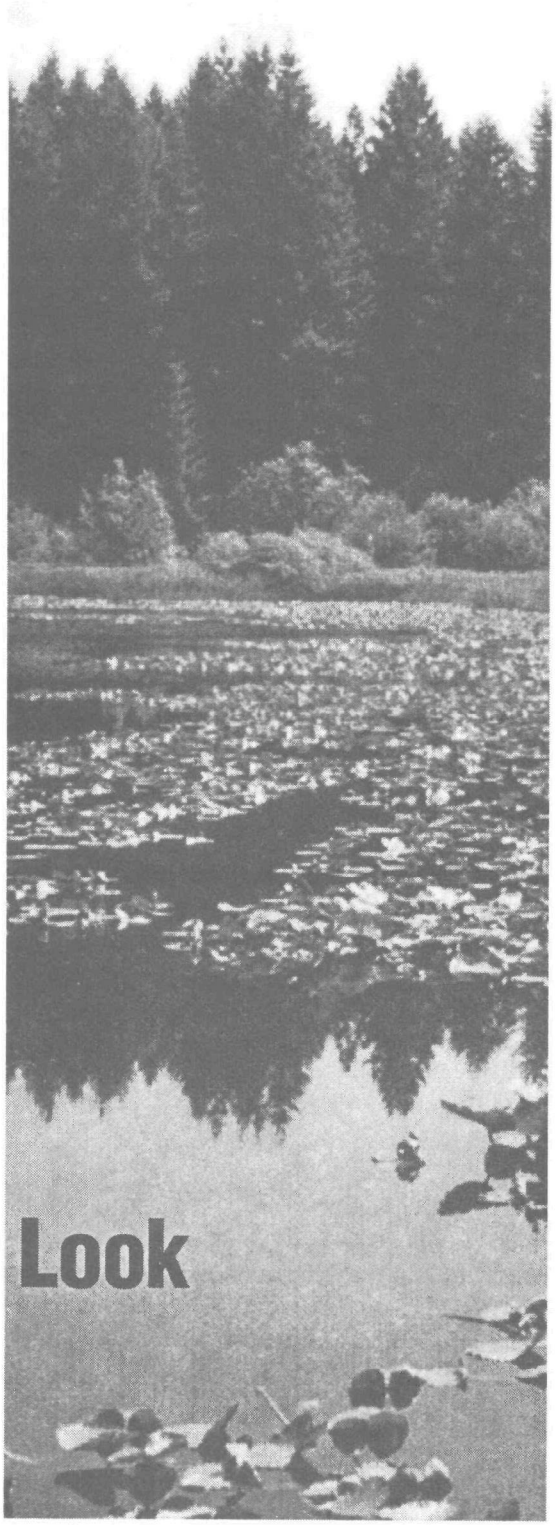
Frost, L. K., N. Hallman, R. F. Buss, R. D. Johnson, K. D. Schmidt, and D. Cotter. 1993. The Town of Gilbert's operational experience with recharge and recovery of reclaimed water. Proceedings of the 29th Annual Conference on Innovations in Groundwater Management and Symposium on Effluent Management, August 29-September 2, 1993, American Water Resources Association. Tucson, AZ.

McHarg, I. L. 1969. Design with nature. Doubleday & Company, Inc. NY.

Tellman, B. 1992. Arizona's effluent dominated riparian areas: issues and opportunities. Water Resources Research Center. University of Arizona. Tuscon, AZ.

Warner, R. E. 1979. California riparian study program: background information and proposed study design. Department of Fish and Game. Sacramento, CA.

sea971820017.doc



Taking A Closer Look

"(Wetland) biology is not simple and you cannot make it simple. Every time you try to make it simple you're probably doing something very wrong and seriously misleading yourself."

Elizabeth Copper
Ornithologist
San Diego–1990

Aquatic Habitat Restoration in the United States: A Review of Design and Costs

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Abstract: **The objective of this study was to compile and summarize non-Corps of Engineers aquatic habitat restoration projects throughout the United States for the purpose of assisting the Corps plan and develop accurate cost estimates for future projects. A subset of 91 projects were evaluated from an initial list of well over 200 projects identified from a literature search. Project descriptions and costing were fully developed for 39 of the 91 projects. The projects ranged from coastal ecosystems to palustrine systems, and were classified into 16 different types (e.g., wetland construction, bottomland hardwood restoration). We found that most were largely done on an individual basis with little impetus to follow set methods or guidelines for establishing goals, performance criteria, monitoring, or documentation. Documentation of all aspects of restoration projects is, in particular given low priority. Costing, was not documented at all in some cases or was not accessible even after considerable effort. We recommend that all aspects of a restoration project should be completely and consistently documented. Planning of future projects requires this type of information to help minimize costs and maximize the probability of success.**

Introduction

The U.S. Corps of Engineers is developing guidance documents to assist Corps planners in the development and implementation of environmental restoration projects. In the coming decades, the Corps will be involved with or leading major environmental restoration efforts. The objective of our study was to compile and compare management measures, engineering features, monitoring techniques, and detailed costs for a representative sample of non-Corps of Engineers environmental restoration projects. Analysis and conclusions reported here are from a project report entitled: *National Review of Non-Corps Environmental Restoration Projects* for the Institute for Water Resources (Shreffler et al. in press). The present paper deals primarily with the issue of costing and cost reporting.

In order to identify appropriate environmental restoration projects for our review, we conducted a literature search of recent conference proceedings and contacted colleagues within the restoration community. Over 200 non-Corps projects were identified. Of these a total of 91 were pursued. We conducted a set of interviews over the phone with experts or individuals with specific knowledge about particular restoration projects, followed by a written questionnaire.

Information from the questionnaire and supplemental project documentation were used to develop 2- to 3-page summaries for each project. Of the 91 projects pursued, 39 were complete enough to be included in final analysis.

Results and Discussion

Project Descriptions

The projects identified in this study were categorized into 16 types, which reflect the primary features of these projects. The classification generally follows Mitsch and Gosselink (1995) for system types and the National Research Council (NRC, 1992) for definitions of restoration, enhancement, and creation. The 16 types of projects included:

- Bottomland hardwood forest restoration

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- Enhancement of fish and wildlife habitat
- Estuarine wetland restoration
- Estuarine wetland restoration and wildlife enhancement
- Mitigation banks
- Stream enhancement
- Stream restoration
- Water quality remediation
- Wetland creation
- Wetland restoration and enhancement

The primary goals of the various projects included reestablishing historical vegetation, restoring or enhancing habitat for wildlife and fish species, stabilizing shorelines, mosquito control, treatment of wastewater, and restoration of hydrology. Many projects were conducted as mitigation designed to offset impacts from another project.

The engineering features cover a wide range of activities. For example, the Natural Resources Conservation Services (NRCS) Wetland Reserve Program has targeted agricultural land that can be relatively easily returned to wetlands. These projects typically involved little physical work but include activities such as removal of impediments to natural hydrological processes. Other projects such as the North Fraser Harbor Habitat Compensation Bank involved a wide variety of actions such as excavation, rip rap shoreline stabilization, and placement of sod.

Monitoring techniques ranged widely from simple observations and photographs taken annually to very complex and integrated sampling and modeling studies. We found no project that exclusively relied on established procedures such as the HEP or the WET, although some projects employed these techniques during part of the assessment or planning phase. This result indicated that although standardized techniques exist, they have not been widely applied.

All projects with monitoring information covering more than one year indicated some level of success relative to goals and criteria for the projects. Because monitoring varied so widely among projects, the success measures varied widely as well. Hydrology, plant growth and cover, and bird use were most often cited as being clear indicators of the performance of the system.

Cost Analysis

A number of previous attempts have been made to analyze restoration costs. In these studies cost estimates are assessed on a total per acre basis and reported in constant cost dollars. King and Bohlen (1994) reported estimates of average wetland restoration costs (excluding land costs) derived from primary data on 90 wetland restoration projects. Costs gathered for their study ranged from $5 per acre to $1.5 million per acre. Guinon (I989) presents results of a cost survey of 25 wetland restoration projects throughout California and illustrates a wide variation in costs ($1,626 to $240,000 per acre). NOAA (1992) summarized some costs of typical wetland creation projects. The costs of restoration projects presented in this study range from $485 to over $70,000 per hectare.

U.S. Department of Interior (DOI 1991) presented the results of a comprehensive review of the available literature on wetland creation and restoration, and wetland mitigation plans. The range of overall restoration costs, (updated to 1989 dollars), is approximately $2,000 to $50,000 per acre although one study showed costs of $220,000 per acre. The DOI report suggests that a portion of this variation may be explained by the fact that there is no "typical" or standard restoration project.

Our study attempted to overcome some of the past limitations of restoration cost estimation efforts. We attempted to derive estimates of the cost of equipment, labor, materials, and supplies for each component of a project from a representative sample of 39 wetland and habitat restoration projects across the U.S.

Individual project contacts were asked to respond to the following questions (within the context of the overall survey):

1. What was the price of the engineering design?
2. What was the cost of monitoring the program?
3. What physical structures were built? How many of each? What was the cost? Year costed?
4. What were the labor costs associated with each major aspect of the project?
5. Are there maintenance requirements for physical conditions or structures? What is the cost of these?
6. Who paid for this program?

One of the biggest problems that we encountered was finding the right person with whom to discuss

project costs. In many cases the institutional knowledge about a project no longer existed or files including cost information had been misplaced. In many cases project costs were reported only as the initial project budget with no detailed accounting of actual expenditures including, for example, change orders, delays, salary and wage increases and etc. (Table 1). In other cases "reported" cost figures often did not include costs for land acquisition, permitting, planning, design, maintenance and monitoring. We attempted, through follow-up phone calls, to elicit information on the true costs of selected projects, those costs mentioned above as well as the opportunity cost of individual's time and donated or contributed labor and equipment.

While many project contacts eagerly reported "costs" it appears that they were unaware of where these figures came from, how they were assessed, or the implications of inaccurate and unreported hidden restoration costs such as project management, overhead and volunteer labor. As illustrated in Table 2, most costs were reported as "installed" or lump sum (Table 2). As a result separate labor, equipment and material costs for a project component were not available. In other cases, the lack of unit descriptions made estimation of costs per unit impossible. As a result, our final total costs estimates are probably, on balance, underestimates of the true costs of the various projects analyzed.

Because the elements associated with the restoration projects analyzed in this study vary across projects, and costs are allocated in different ways across our entire sample, it is impossible to make any statistically significant comparisons of the costs of specific components across projects. However, several components appeared more than once in our sample for which total and per unit costs were reported as follows:

- Gravel removal activity costs range from $3.27 to $3,239 per ton.
- Rip rap installation costs range from $5.00 to $19.00 per ton.
- Culvert installation costs range from $150 (for 48" diameter culvert) to $1,103.85 per ft.
- Channel cleaning costs range from $4.00 to $8.00 per cu.m.
- Erosion control costs range from $1.40 to $4.00 per sq. ft.
- Dike removal costs range from $1.92 to $2.67 per ft.
- Dike/dam/levees construction costs range from $5.00 to $20.00 per linear ft.

From our work we suggest that there are two primary variables determining restoration costs: (1) the specific project components required to restore the ecosystem, from conceptual design to monitoring; and (2) how the restoration costs are allocated and reported.

Costs of implementing restoration projects are unique to each project and are significantly influenced by: site access; grading; site preparation requirements; difficulty of plant community establishment; schedule delays; and, complexity of

Table 1. Cost by Component for the Barataria-Terrebonne National Estuary

Project Component	Quantity	Unit	Equipment, Materials, Supplies, etc.	Labor	Total Costs ($1994)	Total Costs ($1992)
Convert oil canals to marsh	3	ea	$76,059	$44,670	$126,236	$120,729
Fertilizing		LS	$3,370	$3,564	$7,250	$6934
Monitoring		LS	$4,017	$2,074	$6,369	$6,091
Engineering design		LS	$42,449		$44,385	$42,449
Total			$125,895	$50,308	$184,240	$176,204

Table 2. Cost by Component for Palo Alto Harbor marsh construction.

Project Component	Quantity	Unit	Equipment, Materials, Supplies, etc.	Labor	Total Costs ($1994)	Total Costs ($1992)
Mobilization		LS	$11,400	N/A	$11,920	$11,400
Demolition (includes launch ramp)		LS	$15,000		$15,684	$15,000
Clearing and grubbing		LS	$10,000		$10,456	$10,000
Earthwork	49,422	cy	$197,688		$206,706	$197,688
Stock pile embankment	49,422	cy	$45,962		$48,059	$45,962
2-inch decomposed granite on 4-foot aggregate base	14,540	sq ft	$12,504		$13,074	$12,504
2-inch AC on 6-foot aggregate base	1,320	sq ft	$6,600		$6,901	$6,600
Asphalt concrete berm	130	lf	$390		$408	$390
Retaining wall and handrail	140	lf	$32,739		$34,232	$32,739
Retaining wall	124	lf	$19,518		$20,408	$19,518
Bollards	98	ea	$5,880		$6,148	$5,880
Cable	1,520	lf	$3,040		$3,179	$3,040
Detail striping	50	lf	$50		$52	$50
12-inch solid line	50	lf	$50		$52	$50
Pavement marking	4	ea	$50		$52	$50
Directional sign	1	ea	$2,000		$2,091	$2,000
Park bench (double)	3	ea	$2,723		$2,847	$2,723
Park bench (single)	7	ea	$3,086		$3,227	$3,086
Bike rack bollard	6	ea	$480		$502	$480
Trash receptacle	7	ea	$6,523		$6,820	$6,523
Irrigation system		LS	$5,500		$5,751	$5,500
Topsoil, plain	102	cy	$3,500		$3,660	$3,500
Topsoil, mix	370	cy	$12,765		$13,347	$12,765
Weed control rings	903	ea	$1,806		$1,888	$1,806
Buffer plants	903	ea	$3,928		$4,107	$3,928
Marsh plants (cord grass)	1,590	ea	$14,310		$14,963	$14,310
Marsh plants (gumweed)	392	ea	$1,568		$1,640	$1,568
Disking	17,327	sq yd	$2,946		$3,080	$2,946
Hydroseeding	0.23	acre	$0		$0	$0
Siltfence	1,990	lf	$5,970		$6,242	$5,970
Maintenance		LS	$1,500		$1,568	$1,500
Design		LS	$165,000		$172,527	$165,000
Total			$594,476		$621,591	$594,476

habitat management programs. However, other factors affecting final restoration costs include: economies of scale; type of restoration; restoration design; restoration site quality; adjacent site quality; appropriate technology; simultaneous construction multiple use; and, project management.

When reporting, analyzing, and comparing restoration costs, it is essential to consider the project scope and identify all project elements included or excluded from reported figures. The scope of this study and inadequate nature of contact responses did not allow us, however, to adequately inquire about all significant components of each project on a per unit basis, assess costs in context, or specify all those project elements not reported by the project contact.

Conclusions

We found that most non-Corps restoration projects are largely done on an individual basis with little impetus to follow set methods or guidelines for establishing goals, performance criteria, monitoring, or documentation. Our review suggests that documentation of all aspects of restoration projects is, in particular given low priority. Costing, as indicated earlier was not documented at all is some cases or was not accessible even after considerable effort. This hinders the ability to develop the technologies for restoration, which in turn limits the predictability of actions undertaken during restoration projects.

As part of any restoration project we recommend that all aspects of a restoration project should be completely and consistently documented especially if environmental restoration becomes a national focus for federal agencies. Accurate and consistent record keeping is useful for documenting the effects of decisions and to show progress towards goals. In addition, planning of future projects require this type of information to help minimize costs and maximize the probability for success.

Acknowledgments

This study was conducted as part of the Evaluation of Environmental Investments Research Program sponsored by the Institute for Water Resources (IWR) of the U.S. Army Corps of Engineers. We gratefully acknowledge the support of Joy Muncy the study manager at IWR. The cost and project information for Barataria-Terrabonne and Palo Alto projects were kindly provided by Mamie Winter and Jim Harrington, respectively.

Literature Cited

Guinon. 1989. Project elements determining comprehensive restoration costs and repercussions of hidden and inaccurate costs. Paper presented to Society for Ecological Restoration and Management, 1989 Annual Meeting, Oakland, CA.

King, D. M. and Bohlen. 1994. Estimating costs of wetland restoration. National Wetlands Newsletter 16:3-8.

Mitsch, W. J. And J. G. Gosselink. 1995. Wetlands. Second Edition. Van Nostrand Reinhold, NY.

National Oceanographic and Atmospheric Administration (NOAA). 1992. Restoration guidance document for natural resource injury as a result of oil spills. National Oceanographic and Atmospheric Administration, Washington, D.C.

National Research Council (NRC). 1992. Restoration of Aquatic Ecosystems. National Academy Press, Washington, D.C.

Shreffler, D. K., R. M. Thom, M. J. Scott, K. F. Wellman, M.A. Walters, and M Curran. In press. National review of non-Corps environmental restoration projects. Institute for Water Resources, U.S. Army Corps of Engineers. IWR Report 95-R- 12. Alexandria, VA.

U.S. Department of the Interior (DOI). 1991. Estimating the environmental costs of OCS oil and gas development and marine oil spills: a general purpose model. Report prepared by A. T. Kearney, Inc. for the U.S. Department of Interior, Minerals Management Service, Washington, D.C.

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Wetland Mitigation Design: From Theory to Reality

Marc Boule[1]

Abstract: **Watershed planning, and other non-project habitat restoration techniques, offer approaches for providing suitable wetland mitigation for proposed project impacts. Experience suggests, however, that these basic ideas do not (and often cannot) always consider market forces and other issues facing the project proponent. Examination of the selection and design of the wetland mitigation for the Emerald Downs Racetrack offers an opportunity to compare the ideal approach with the reality of political, financial and scheduling constraints. These constraints may limit the opportunities for developing preferred or recommended mitigation approaches.**

Introduction

In the past, wetland mitigation has been described as more art than science. While there has been some truth to that, today it seems that what has become equally important is the art of communication. This is especially true with large or controversial projects (they seem to go hand-in-hand), where numerous agencies and individual citizens or citizen's groups are involved. The controversy, or at least confusion, is often complicated by the conflicting mandates between, and sometimes within, the various resource agencies. Consider, for example, the Washington Department of Fish and Wildlife, where some personnel are responsible for protecting heron rookeries while others find herons to be an egregious predator on juvenile salmon and steelhead.

This paper describes a controversial project that went through several years of planning and environmental review. The purpose here is not to discuss the environmental process; rather, the goal is to emphasize the changes in project and mitigation design in response to agency review, and to compare and contrast what was proposed, opposed, and ultimately approved, and is presently under construction.

Project Description

Pre-Project History

In 1992 the Longacres Thoroughbred Racetrack, just south of Seattle, Washington, was purchased by The Boeing Company with the intent of converting the site into a corporate campus. As a result of that purchase, the horseracing industry found it necessary to develop a new track facility. After almost three years of searching, a site was identified in the City of Auburn in the Green River Valley, about 20 miles south of Seattle (Figure 1).

Initially, the proposed project site encompassed 198 acres and required the fill of 53 acres of wetland. About 36 acres of this was wet agricultural land that had been abandoned for many years. Approximately 156 acres of the site was either actively grazed pasture (including 17 acres of wet pasture) or previously filled pasture, at the time it was identified as the potential racetrack site (Figure 2). Once the project proponents became aware of wetland regulatory issues, they began to search for a mitigation site. Because the project was located in the watershed of Mill Creek, a tributary of the Green River, the search was concentrated in the Mill Creek Valley.

Over the past 100 years, and especially the past 15 to 20 years, development in the Green River Valley has resulted in the loss of wetlands, floodplain, wildlife habitat, and other resources. As a result of these impacts and continued development pressure in the valley, federal, state, and local natural resource agencies have combined efforts to develop a Special Area Management Plan (SAMP) for the Lower Mill Creek Valley. Although the SAMP is not yet completed, some preferences for certain types of development or habitat protection in the valley have been identified. These include identification of areas where development might be considered more appropriate and where wetland preservation or restoration might be more appropriate. The basic goals of the SAMP process were used to identify a potential mitigation

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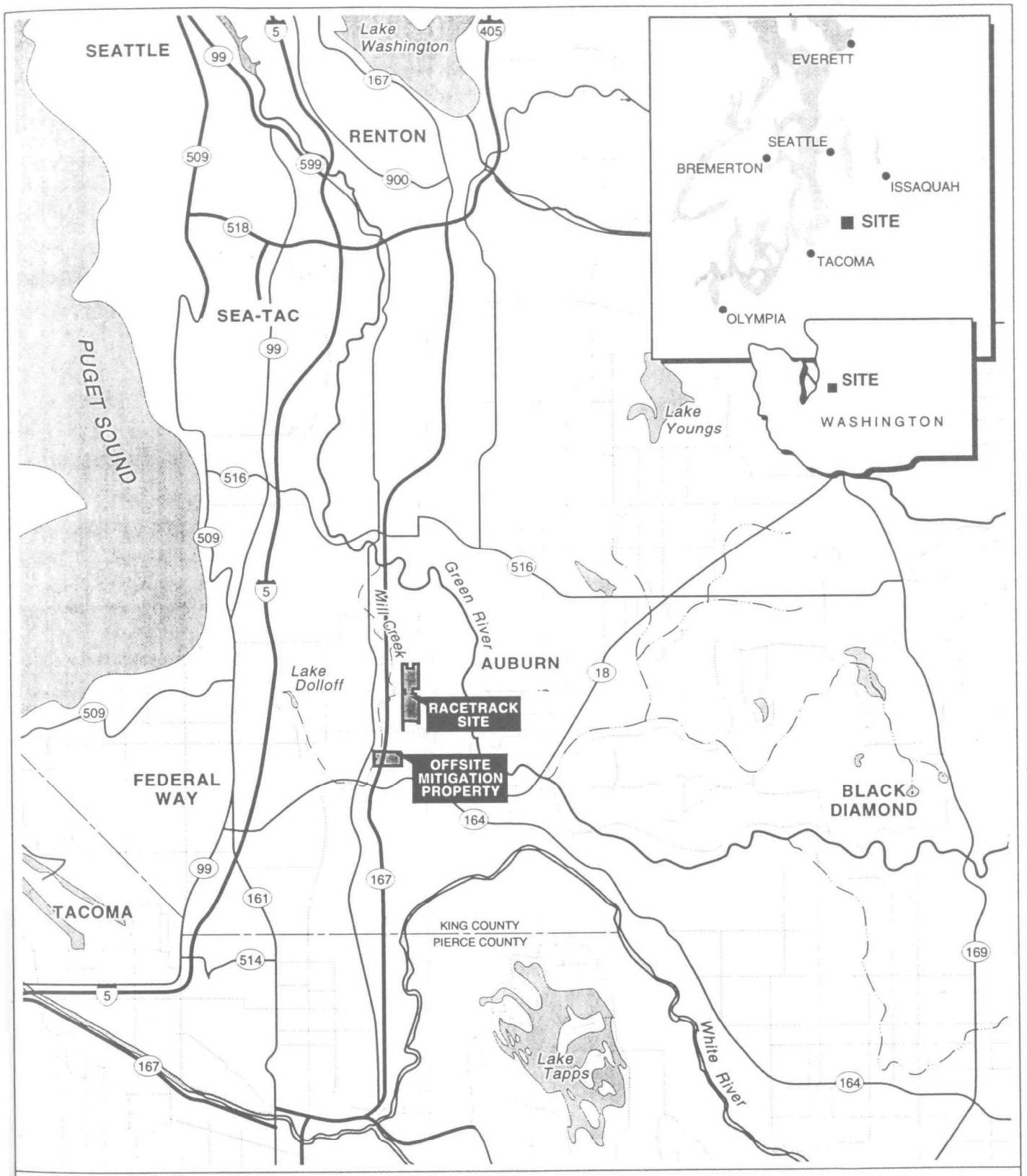

Figure 1. Site Location Map.

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Scale in Feet

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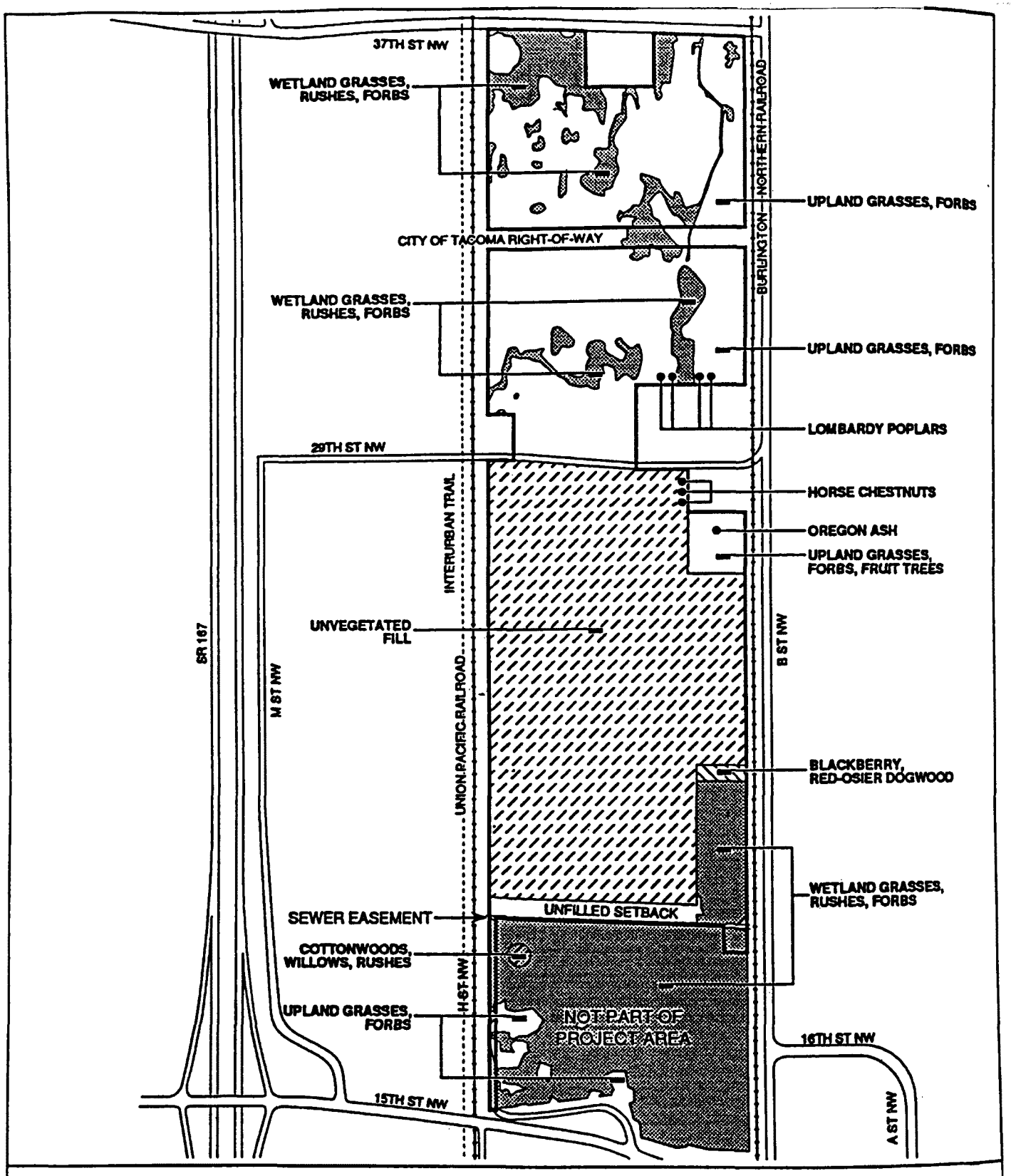

Figure 2. Existing Vegetation Racetrack Site.

site for the proposed racetrack facility and to develop a wetland mitigation plan. One of the SAMP goals is restoration of wetland habitats within the Mill Creek Basin.

Not much is known about the character of the Green River and Mill Creek Valleys prior to Anglo-American settlement. Soils suggest much of the area was wetland, with a variety of hydric soil types. Historic records, such as newspaper stories, refer to impenetrable swamps in the area. Many of the early roads were made of logs "floated" on top of wet soils. In the early 20th century, state forestry maps show much of the area converted to agriculture, but unconverted areas are designated as wet forest cover.

The Mitigation Planning Process

The First Proposal

The first mitigation proposal by the project proponent was to use a series of abandoned sewage treatment plant lagoons immediately adjacent to the project site(Figure 3). The lagoons are used by waterfowl at present, but the riparian habitat adjacent to the open water is limited because of the steep slopes on the berms enclosing the lagoons and because most of the upland is dominated by non-native species such as blackberries, Scot's broom, and weedy grasses and forbs. In addition, sludge in the lagoons has been a problem for which the City of Auburn is still trying to find a solution. The proposal, developed by Raedeke and Associates (1992), was to excavate and appropriately remediate the sludge material and then create a diversity of waterfowl and riparian habitat on the site.

At the U.S. Army Corps of Engineers pre-Application presentation to the resource agencies for the proposed racetrack, several objections were raised regarding the proposed mitigation. First, much of the area was identified as wetland and waterfowl habitat, so there would be little creation of new wetland to offset losses of wetland area associated with the project. Furthermore, the proposed mitigation site is immediately adjacent to Auburn Municipal Airport, so enhancement of waterfowl habitat is not recommended nor highly regarded by the Federal Aviation Administration. Finally, the SAMP process pointed toward restoration of wetlands in the vicinity of Mill Creek, and the abandoned sewage lagoons are even farther from Mill Creek than the project site. In general, this proposal was not well received.

The Second Proposal

With the concerns expressed at the pre-Application meeting in mind, the project team met with the SAMP committee to identify potential mitigation areas. As a result of those discussions, areas of Lower Mill Creek where restoration might be implemented were identified and the project proponents began the search for a site to purchase. In the lower valley, Mill Creek has been severely ditched along most of its length and is generally characterized as a vertical-walled channel up to 5 or more feet in depth with little or no overstory riparian canopy. Water quality, especially high turbidity, high temperatures, and low dissolved oxygen (DO), are the major factors limiting its value as salmonid habitat.

Ultimately the proponents were able to identify and acquire a 56.5-acre parcel about 1 mile south and upstream of the project site. Although it offered many opportunities, it also was bisected by SR 167, the major freeway through the area. The 26.5-acre portion of the site west of the freeway has no roadway access, straddles 1/4 mile of Mill Creek, and consisted of about 19 acres of wetland dominated by reed canarygrass and other non-native species, 4.6 acres of native emergent and scrub-shrub wetland, and 2.9 acres of upland, primarily abandoned pasture. The approximately 30-acre site on the east side of the freeway, consisted of 19 acres of grazed wet pasture, 9.2 acres of upland pasture, and about 1.5 acres of upland and filled wetland that supported a house and barns adjacent to the pasture.

The project team evaluated this new site in light of its existing wetland functional values and developed a restoration plan to initiate plant communities similar to those thought to have existed in the area prior to European settlement. The plan involved reconfiguring the creek channel away from a ditch cross-section and more toward a natural stream channel cross section, and vigorous mechanical control of reed canarygrass in combination with heavy planting of native trees and shrubs (Figure 4). The intent was to create, over time, an overstory community that resembled the pre-settlement community, which would shade out the reed canarygrass and would improve water quality in the creek through shading.

Some concerns were expressed by resource agencies about the specifics of the plan, particularly whether shading could effectively eliminate reed canarygrass and whether reconfiguration of the creek channel would result in significant water quality impacts during construction. The major concern

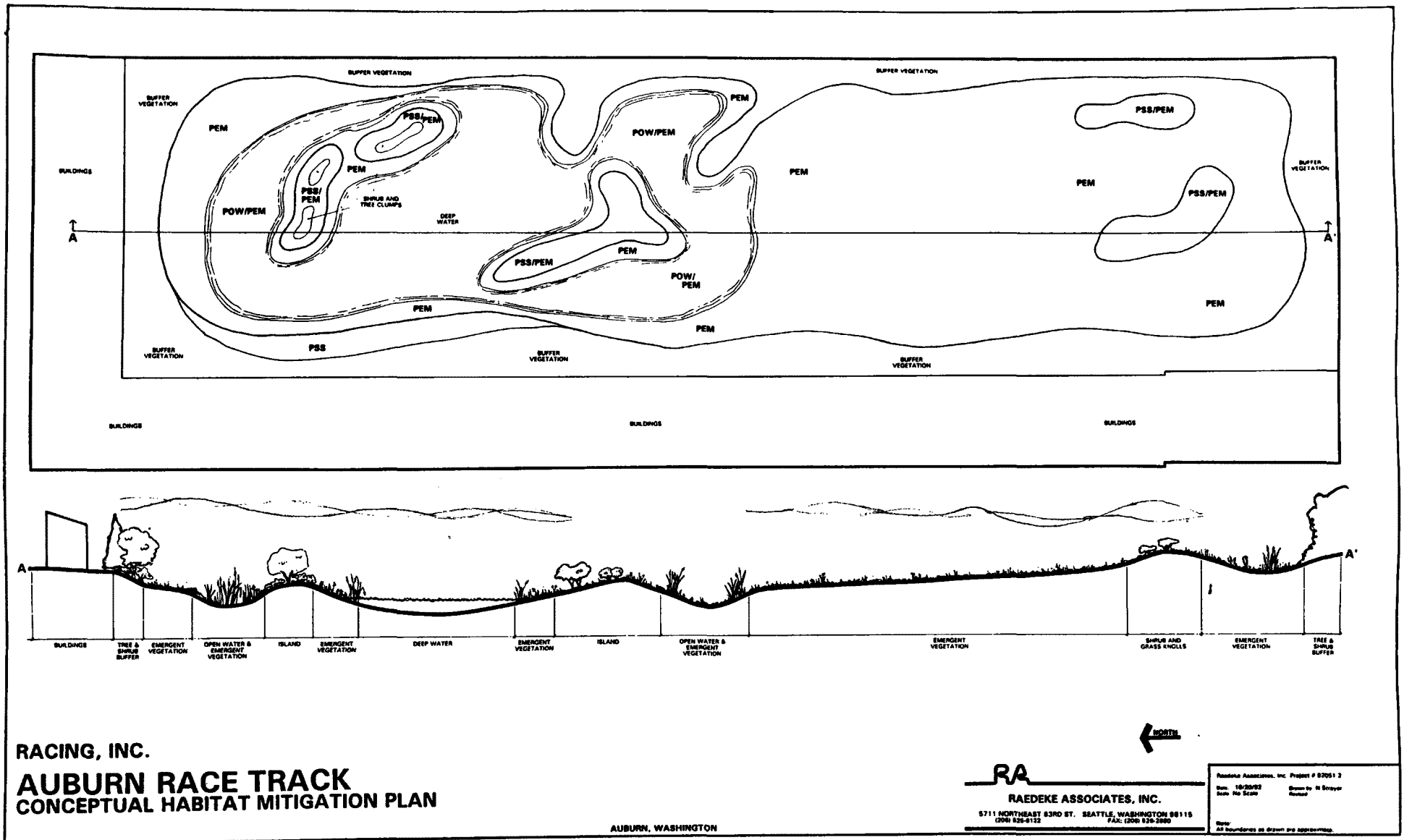

Figure 3. First Mitigation Proposal (1992).

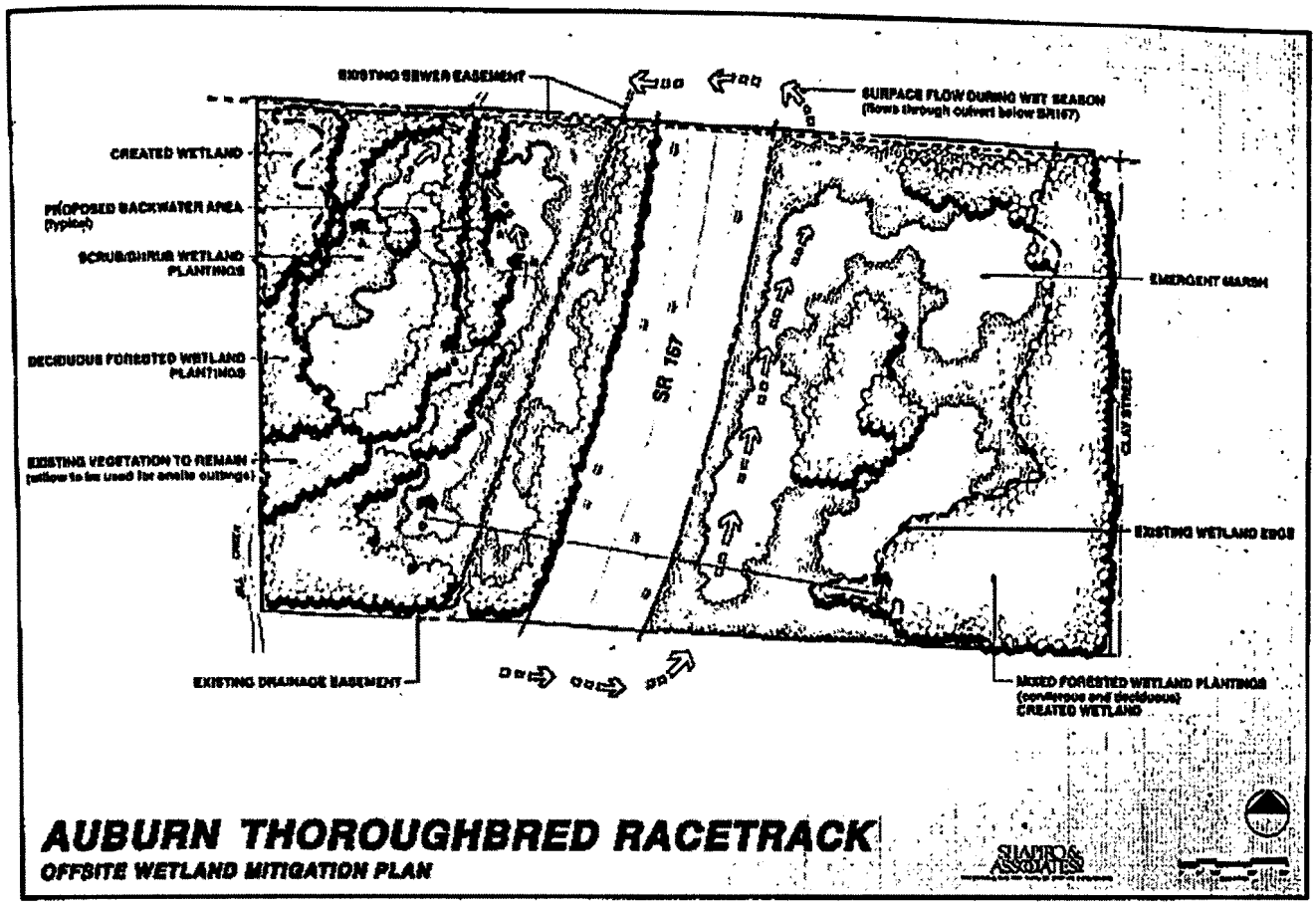

Figure 4. Second Mitigation Proposal.

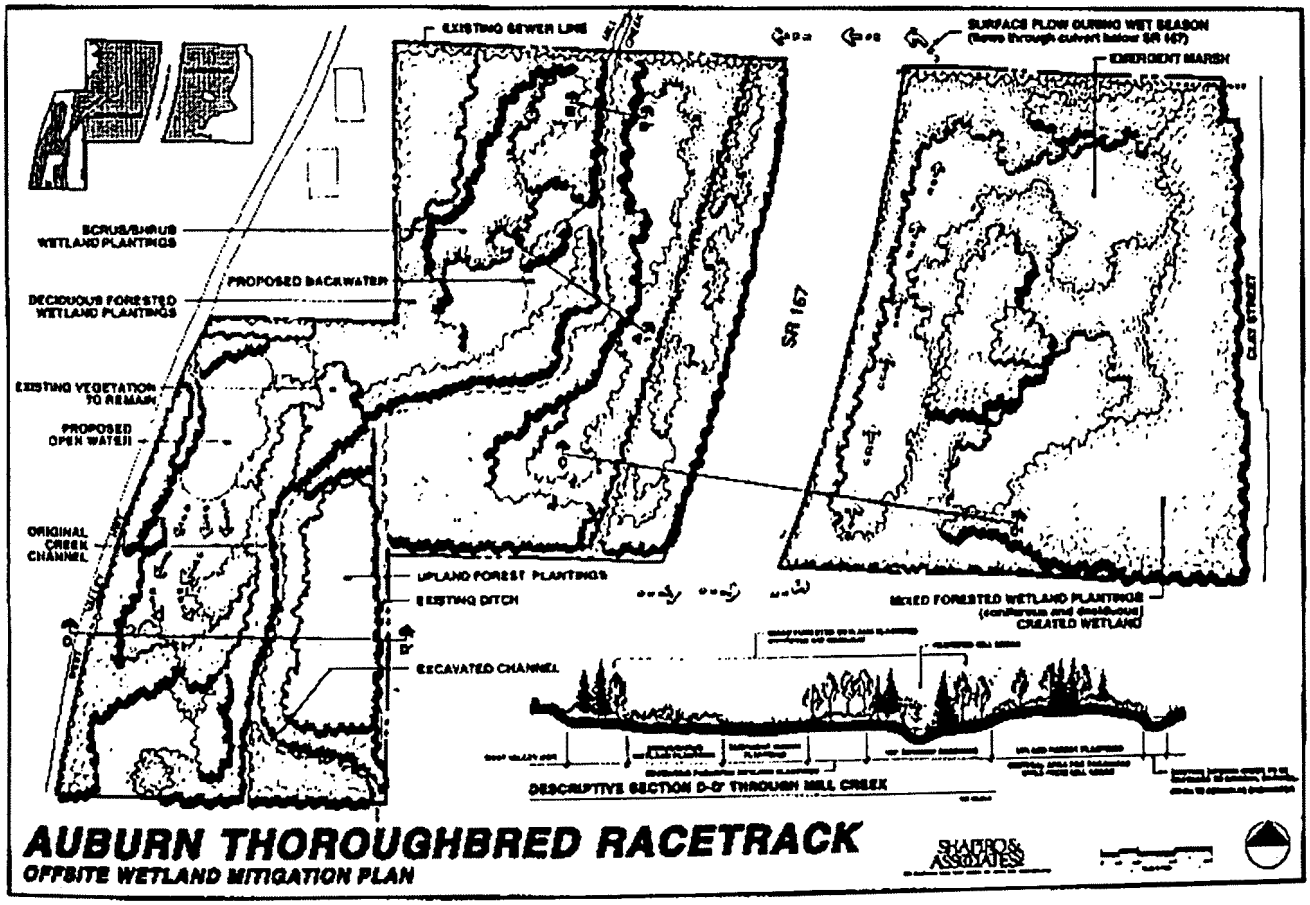

Figure 5. Third Mitigation Proposal.

expressed, however, was that the approximately 60-acre mitigation area, with only 10 acres of new wetland creation, would not adequately mitigate the 58 acres of proposed wetland impact.

The Third Proposal

The project proponents then acquired an option on an additional 20 acres of land immediately upstream of the initial mitigation site, about half of which was upland and included another 1/4 mile of Mill Creek. A third restoration proposal was now prepared that involved conversion of another 10 acres of upland and fill to wetland, and restoration of an additional 10 acres of wetland through control of pasture grasses and planting of native trees and shrubs (Figure 5). Additionally, this third proposal offered the opportunity to reintroduce Mill Creek into a portion of its original channel. At some time in the past, the creek had been diverted into a ditch along the back lot of this property, but remnants of the original channel remained. Creation of a short segment of new channel would allow the creek to be reintroduced into its original channel, while allowing the ditch to function as a backwater channel, for fish refuge during periods of high flow.

Resource agency concerns about this proposal included a perception that wetlands on this site were not significantly degraded (despite the fact they had been converted to agriculture and grazing land for almost a century and abandoned only in the last 10 to 20 years), and therefore could not be substantially improved. But more importantly, a concern was expressed that reintroduction of the creek into its old channel would affect water quality in the creek because of potential erosion from introducing the creek into an unestablished channel. Furthermore, the relocation would result in the loss of emergent wetland communities that had become established in the existing abandoned creek channel.

The Fourth Proposal

At this point the project proponent decided it would be expeditious to further reduce project impacts to expedite the regulatory process. Several elements of the project were reduced or eliminated, and the overall extent of the project was reduced. Most importantly, the abandoned agricultural land at the south end of the site, with its 39 acres of wetland, (Figure 2) was eliminated from the project area. As a result, wetland impacts were reduced to a fill of 17 acres of grazed wet pasture. With this reduction in the extent of wetland impact, the project proponent found it uneconomical to include in the mitigation plan the 20-acre parcel added in the third restoration proposal described above.

At the same time, two long meetings were held with representatives from the project proponent, the U.S. Army Corps of Engineers (lead permitting agency), and all other federal, state, and local agencies, and the tribes with jurisdiction or concern over the project. The purpose of these meetings was to develop a consensus of opinion on what would be appropriate mitigation for the proposed project. The result of these meetings was a series of guidelines–but not quite performance standards–describing what the mitigation needed to accomplish. The overriding focus of these guidelines was improved waterfowl habitat, because that was identified by the resource agencies as the greatest habitat value on the proposed project site. Fish habitat in Mill Creek was suggested as the second most important element in any mitigation plan. (Interestingly, while fish habitat is recognized as a limiting condition in Mill Creek, impacts of the proposed project on fishery resources was expected to be limited to stormwater runoff impacts, that, based on computer modeling and other assessment techniques, were anticipated to be minimal.)

Based on the input from these meetings, a fourth restoration proposal was prepared for the 56.5-acre site previously identified for mitigation. The emphasis of this fourth plan was on waterfowl breeding habitat, because the project site had been identified as waterfowl overwintering habitat. Interestingly, because green-winged teal had once been observed in the abandoned agricultural area originally proposed as the south part of the project site, this species became a key indicator for state and federal resource agency personnel, and its habitat requirements were the major criteria for developing the mitigation plan. The result was a mosaic of open water ponds scattered over the site and separated by zones of scrub-shrub wetland community (Figure 6). A forested wetland community surrounded the outside. Little was proposed for Mill Creek itself, except to provide a canopy of forested and scrub-shrub species along its banks.

Although the goal had been to reflect agency priorities, response to the fourth restoration proposal was mixed, and several revisions were suggested by agency personnel. Most importantly, it was determined that there was too great an emphasis on

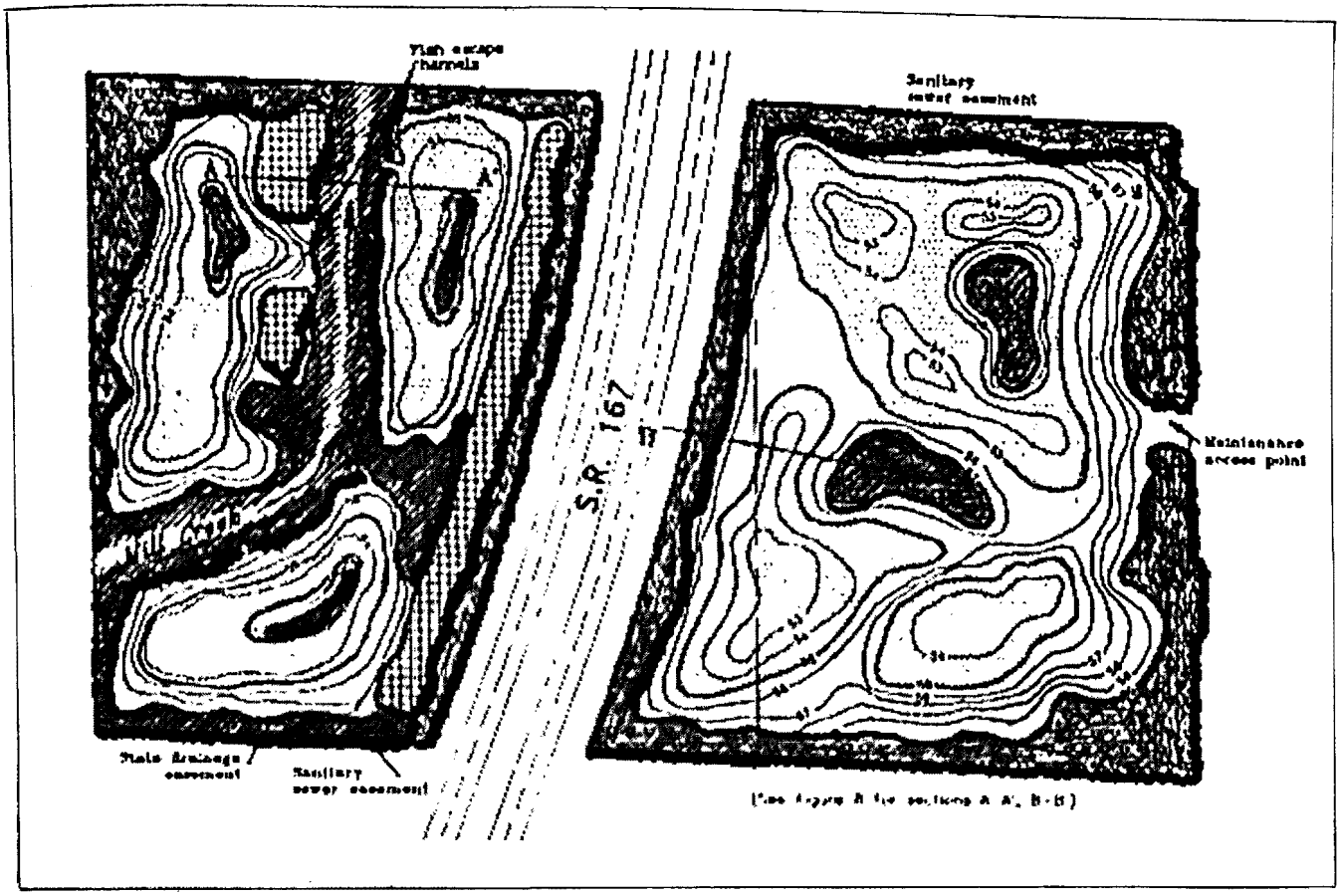

Figure 6. Fourth Mitigation Proposal.

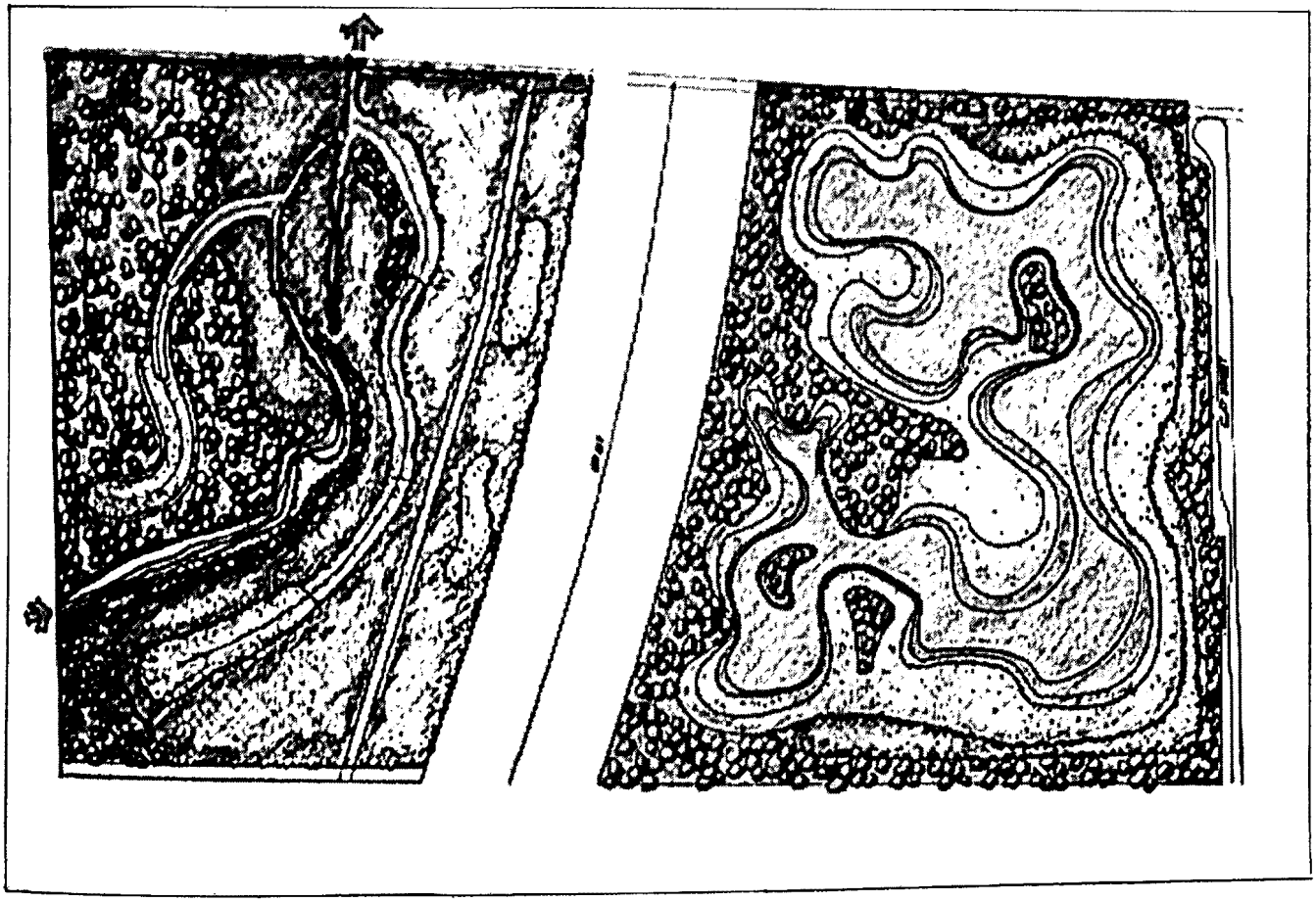

Figure 7. Final Mitigation Proposal.

waterfowl habitat, especially on the west side of the parcel. In addition, a suggestion was made to develop a new meander to the creek channel and to provide backwater channels as salmonid refuge habitat. Finally, a desire was expressed to create some shallow sedge meadow communities.

The Fifth Proposal

Back to the drawing board again, the project team developed a fifth proposal that reflected the most recent revisions requested by the resource agencies (Figure 7). A new creek channel was included, as were backwater channels. Open water on the west side was limited to two small pond areas adjacent to the freeway that do not drain directly to Mill Creek. This was the proposal finally approved, with some minor modifications, as part of the Section 404 permit for the proposed project (USACOE 1995).

Summary

Controversial projects requiring especially large areas of wetland fill can require extensive coordination with resource agencies to determine appropriate mitigation. In this example, development of an acceptable wetland mitigation plan required extensive negotiation and numerous revisions to respond to the diverse, and sometimes changing, concerns of the agencies. The following is a brief summary of the mitigation design and negotiation process.

Proposal: 52 Acres of Wetland Fill

Step 1 Mitigation: Enhance waterfowl habitat adjacent to the project site.

Concerns: Too close to airport, not enough wetland creation, too much emphasis on waterfowl and not on wetland restoration.

Step 2 Mitigation: Restore 40 acres of wet pasture and 10 acres of upland to forested swamp, and enhancement of 1/4 mile of Mill Creek.

Concerns: Not enough wetland creation, potential construction impacts of reconfiguring stream channel.

Step 3 Mitigation: An additional 10 acres of mostly abandoned wet pasture for enhancement, an additional 1/4 mile of Mill Creek to restore, including relocating creek into original channel.

Concerns: Still not enough wetland creation or gain of functional value, need more waterfowl habitat, potential impacts of restoring creek to original channel.

Proposal: Reduce Project Impact to 17 Acres of Wetland Fill

Step 4 Mitigation: Reduce area of mitigation to 50 acres, emphasize waterfowl habitat.

Concerns: Too much open water, potential long-term impacts on water quality in creek, need to create new stream channel and sedge habitat.

Step 5 Mitigation: Reduce area of open water, create new creek channel, backwater channels, and sedge habitat.

Concerns: Still a 7-acre net loss in total wetland habitat within the Green River Valley.

Lessons Learned

Communication

Even with numerous meetings and telephone conversations, opportunities for miscommunication or missed communications abound. To reduce this, every meeting should be documented. Had we summarized our meetings with the SAMP committee, noting our understanding of what was implied by the concept of "restoration," we possibly would have avoided the effort and delays involved in creating a plan that did not meet other agencies' concerns. Had the resource agencies been clearer early in the process, about the importance waterfowl, the emphasis of the mitigation plan could have been redirected. There cannot be too much communication with regulatory agency personnel when developing large and controversial projects.

Consistency

The mandates, and thus the goals, of the myriad of resource agencies involved in the regulatory process may vary significantly. There is often a marked lack of consistency in these mandates and goals. Optimization for waterfowl habitat is not always the best for fisheries resources, and vice versa. Agency representatives must establish their parameters and priorities–preferably early in the process. Otherwise, fulfilling one agency's requirements only leads to direct conflict with another.

Collaboration

Agency staff should be included in the design process. It is a delicate, politically sensitive process, for agency staff often do not want to be seen as "encouraging" a project. It also leads to design by committee, but this might be preferable to, and more cost- and time-effective than, *redesign by committee*.

Consensus

Without consensus on the part of resource agencies, a proposed project is destined to flounder, losing time and costing money. The project proponent and/or their consultants are not always in the position to ensure consensus, but that should always be their emphasis.

The Emerald Downs wetland mitigation experience was certainly not a model of efficiency, and the project suffered more from political, non-wetland concerns that have not been discussed here. Nonetheless, this experience offers some important lessons in how to reduce the controversy and streamline the process.

Literature Cited

Raedeke Associates. 1992. Conceptual Wildlife Habitat Mitigation Plan for the Garrett/Lone/ Auburn Downs Race Track Property, Auburn, WA. Prepared for Garrett Enumclaw Company.

U.S. Army Corps of Engineers. 1995. Auburn Thoroughbred Horse Racing Facility, Final Federal Environmental Impact Statement (NEPA), USACOE, Seattle District, Seattle, WA. 2 vols and appendices.

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Ecological Functions of a Saltmarsh/Mudflat Complex Created Using Clean Dredged Material, Jetty Island, Washington

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Robert H. Gilmour[1]

Abstract: **The Port of Everett and the U.S. Army Corps of Engineers created a sand berm on an existing island in the winter of 1989 and 1990. Four objectives were established: (1) to balance erosion losses on the exposed west side of the island, (2) to create additional dune grass habitat, (3) to create a protected embayment that would be colonized by marine invertebrates, and (4) to demonstrate a beneficial use of clean dredged material. Development of the embayment was expected to allow increased productivity and invertebrate biomass, thereby improving habitat for juvenile salmonids, other fish, shorebirds, and waterfowl. Progress toward meeting these objectives has been monitored over 5 years. A 5.3-ha (13-ac) berm was constructed with clean dredged material, forming a protected soft mudflat that covers 7.7 ha (19 ac). Experimental planting of six saltmarsh species conducted in the spring of 1990 demonstrated that upper portions of the embayment (above +9 ft MLLW) would support such species. A second planting of *Jaumea*, *Salicornia*, and *Distichlis* in this area in 1991 has flourished and achieved over 100 percent cover in some areas. Natural colonization of salt-tolerant species also has occurred above +10 ft MLLW. Field sampling has demonstrated that juvenile salmon and other fish use the embayment during high tides and that the lagoon supports a rich population of epibenthic zooplankton prey. The protected mudflat is widely used by migrating dunlin, dowitchers, plovers, other sandpipers, and bald eagles. The project fully met its objectives by the 1995 date set for final project evaluation.**

Introduction

The Jetty Island berm project was planned, designed, and coordinated by the Port of Everett (Port) and the U.S. Army Corps of Engineers (Corps), Seattle District, to demonstrate beneficial use of clean dredged material for habitat development. By creating a sand berm on an existing island, the Port and the Corps expected four objectives to be met within 5 years: (1) to balance erosion losses on the exposed west side of the island, (2) to create additional American dunegrass (*Elymus mollis*) habitat, (3) to create a protected embayment that would be naturally colonized by marine invertebrates, and (4) to demonstrate a beneficial and economical disposal option for clean dredged material. The protected intertidal mudflat within the embayment was expected to contribute to increased productivity and biomass of fish prey and, therefore, improved habitat for juvenile salmonids outmigrating to northern Puget Sound.

Construction of the federal navigation channel in Everett Harbor from 1894 to 1903 resulted in large volumes of sediment requiring disposal. Creation of Jetty Island began in 1903 with construction of a rock jetty behind which these dredged materials could be placed. Corps maintenance dredging of the channel and settling basins (Figure 1) continued to build the island until 1969. Today Jetty Island is approximately 3 km long and covers approximately 40 ha (100 ac) above mean higher high water (MHHW). Upland vegetation is dominated by Scot's broom (*Cytisus scoparius*), blackberry (*Rubus* spp.), and many other shrubs and herbaceous species. Extensive mudflats and Possession Sound lie to the west of the island; the Snohomish River estuary, the federal navigation channel, and the City of Everett lie to the east (Figure 1).

In the mid-1980s the Corps identified an opportunity to enhance and increase existing Jetty Island habitat with material from maintenance dredging of the channel and lower settling basin. In 1987 an Agency Technical Advisory Committee (ATAC) was formed to develop specific project details. An environmental assessment of the area was completed prior to placement of dredged material (U.S. Army Corps of Engineers 1989a). Abundance and

[1] Pentec Environmental, Inc., Edmonds, Washington.

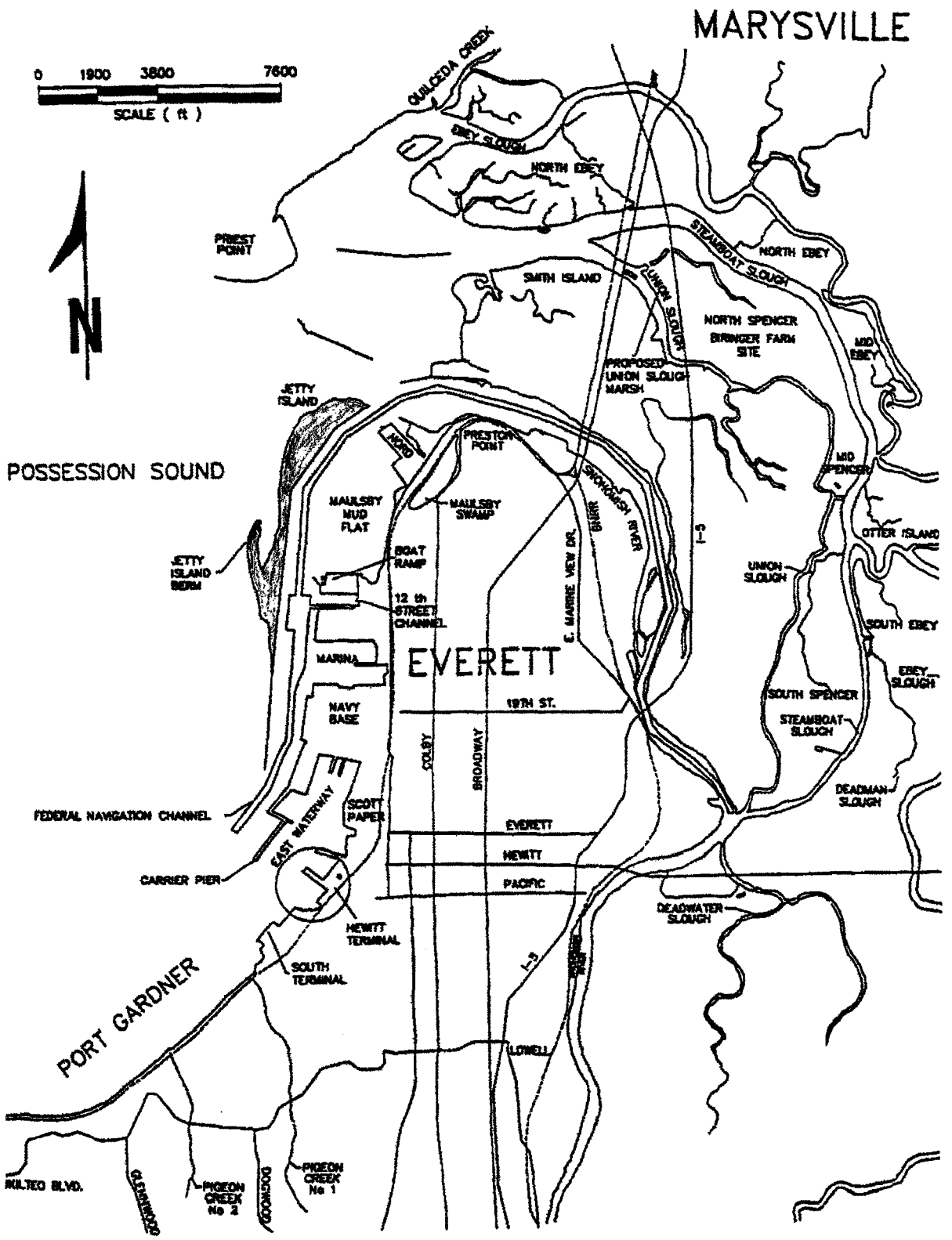

Figure 1. Location and Vicinity Map

distribution of benthic infauna and epibenthic Crustacea as well as use of the project area by resource species (salmon, Dungeness crab, birds) were investigated. Environmental design and monitoring programs were finalized in the Implementation Plan (U.S. Army Corps of Engineers 1989b), which stated project objectives, responsibilities of participants, performance standards, and methods for evaluating and monitoring success.

Specific monitoring was required over 5 years. A quantitative assessment method was developed to evaluate pre- and post-project juvenile salmon habitat losses and gains. For the project to be considered a success, surveys were required to demonstrate that (1) juvenile salmonids use the project area, and (2) juvenile salmonid habitat is increased by at least 10 percent over pre-project levels. Other studies evaluated the physical stability of the berm and the success of experimental plantings of saltmarsh vegetation within the lagoon formed by the berm. The criterion for success of saltmarsh habitat development was that the vegetative community cover at least 40 percent of the planted area inside the berm and not differ statistically from that of the saltmarsh reference site located on the eastern side of the island (Figure 2).

Hydraulic maintenance dredging was completed on January 6, 1990. A 550-m (1,800-ft) long berm, occupying a little over 5.3 ha (13 ac), was created by placement of 323,000 cy of clean sands and silts on the west side of Jetty Island. Placement of the berm created a 7.7-ha (19-ac) mudflat within a protected embayment on what had been the sandy west shore of the island.

Methods

Physical Monitoring

The Corps developed physical performance criteria for the project and monitors physical features (i.e., coastal geomorphology, topography). Approximately 65,000 cy of the berm was predicted to erode over 8 to 10 years. As long as the berm erosion rate does not exceed this predicted rate by more than 25 percent, the frequency of nourishment required to preserve it should be about every 10 years (U.S. Army Corps of Engineers 1989b).

Biological Monitoring

Epibenthic zooplankton was sampled in April and May of 1992 and 1995 using a 0.018-m^2, battery powered epibenthic sampler. Three replicate samples were analyzed from each of 18 stations representing three depth strata (+9, +6, and +3 ft mean lower low water [MLLW]) inside and outside the berm and at the north and south reference areas (Figure 2). Samples were sieved on a 0.25-mm^2 screen, and zooplankters recognized as key prey taxa for juvenile pink and chum salmon were enumerated and wet weighed.

To evaluate the success of the project in terms of juvenile salmonid habitat, an equation was developed and adopted by the ATAC to describe the quality of the habitats in terms of biomass of key prey species and other relevant factors. This equation (U.S. Army Corps of Engineers 1989a; Pentec Environmental, Inc. 1994) is as follows (see Table 2 for definitions of factors):

Habitat Units (HU) = ac X PWF X (Cover factor + Slope factor) X RIF

Fish use of the area was evaluated using a 40-m (130-ft) beach seine in the spring of 1992 and 1994 under high tide conditions at locations inside and outside the berm and at the north and south reference areas.

Because the artificial establishment of saltmarsh vegetation in urban estuaries of Puget Sound was not well understood, experimental planting was conducted in 1990, immediately after placement of the dredged material, to determine at which elevations within the embayment saltmarsh vegetation would survive. Experimental plugs of *Deschampsia caespitosa, Carex lyngbyei, Triglochin maritima,* and *Scirpus maritimus* were obtained from Wave Beach Grass Nursery in Florence, Oregon. Plugs of native saltmarsh vegetation, predominantly *Distichlis spicata*, were separately dug from a reference marsh on the east side of the island (Figure 2). Dunegrass also was planted over approximately 1.5 ha (3.6 ac) along the western side of the berm.

By September 1990, it was apparent that vegetation would survive only within a limited area of the embayment; thus, plantings in the spring of 1991 focused on areas between +9 and +12 ft MLLW. *Distichlis*-dominated plugs collected from the reference marsh were planted along the inside of the berm primarily between +11 and +12 ft MLLW to augment *Distichlis* surviving from the 1990

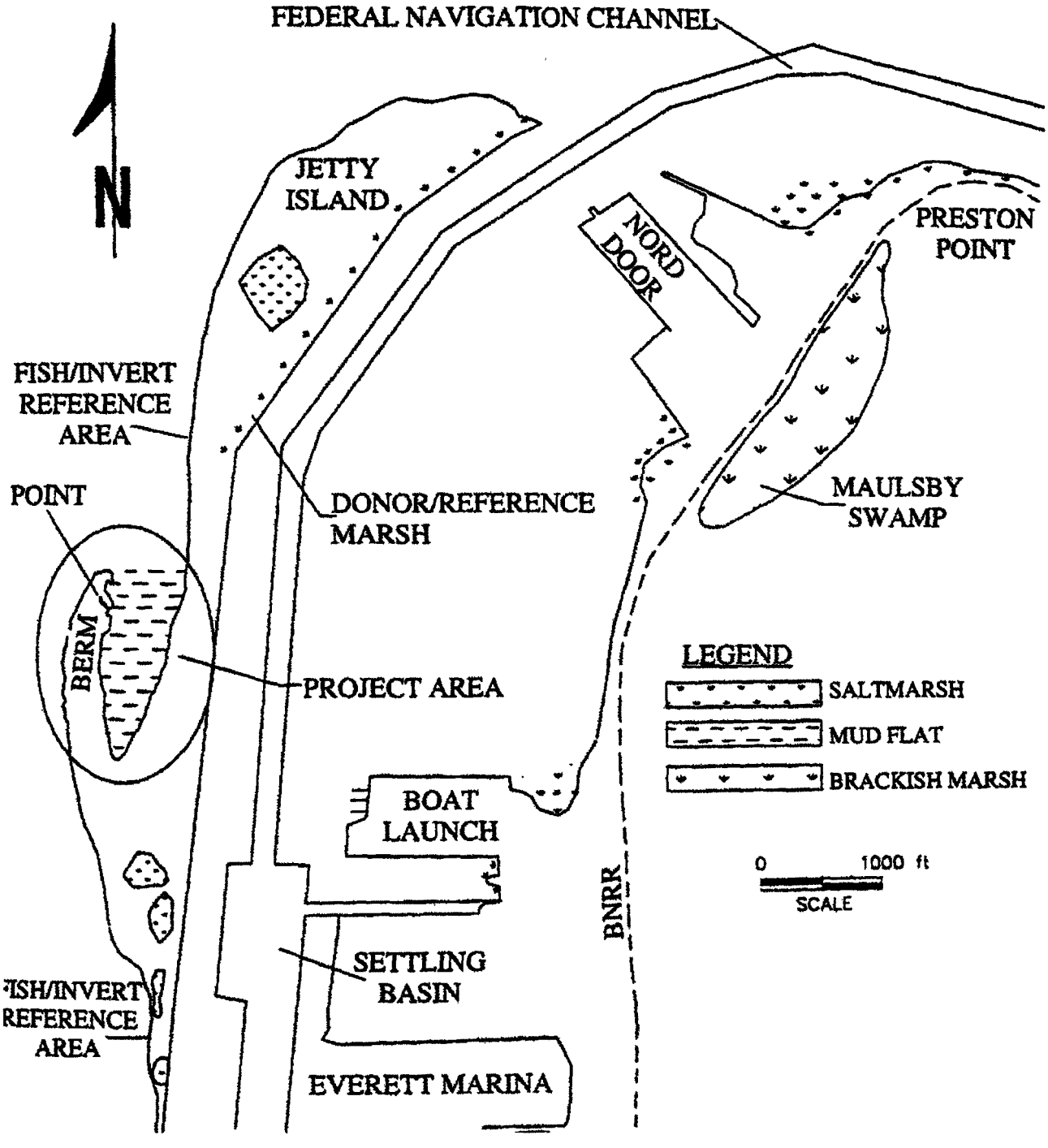

Figure 2. Project Area

plantings. *Salicornia* and *Jaumea* plugs from Wave Beach Grass Nursery were planted primarily between +9.5 and +11 ft MLLW along the inside base of the berm (southern tip of the protected lagoon) approximately 106 m to the north along the inside of the berm and approximately 40 m north along the island shoreline (east side of the lagoon).

Field monitoring was conducted in early September 1993 and in late August 1995 (Years 3 and 5) to measure the success of the saltmarsh plantings as well as natural colonization. In the project planting area, three relatively distinct vegetative zones were noted:

1. A lower zone between +9.5 and +11 ft MLLW that was dominated by *Jaumea carnosa* with a significant component of *Salicornia virginica* (the "lower zone" or "*Jaumea/Salicornia* zone").

2. An upper zone beginning at about +11 and +12 ft MLLW and continuing up into the supralittoral in some areas that was dominated by *Distichlis spicata*, *E. mollis*, and *Spergularia* (the "upper zone" or "*Distichlis/Elymus* zone").

3. A supralittoral zone generally above +12 ft MLLW that was dominated by *E. mollis* from natural colonization.

The *Jaumea/Salicornia* and *Distichlis/Elymus* zones were sampled as distinct strata; the supralittoral zone was not sampled. North of the planted area, the pattern of three distinct zones was not evident, and a single zone dominated by *Spergularia* was sampled. The reference marsh was sampled to gauge success of the marsh developing within the protected embayment. A small sub-embayment inside the first small spit that has formed at the end of the berm was also sampled to document natural colonization that is occurring there. Percent cover was determined by species in 25, 0.25-m^2 quadrats randomly placed over the upper and lower zones within the inner berm and at the reference site. Similar data also were recorded for the reference area and the "Point" area at the tip of the berm (Figure 2). Individual species cover values were summed to derive total cover in the quadrat.

Randomized analysis of variance (ANOVA) and t-tests (Edgington 1987) were used to test for significant elevation or site differences in species cover and total plant cover.

Results

Physical

A portion of the western side of the berm first eroded during winter storms in 1990/1991 and has continued to erode (Nelson 1996). Natural beaching of logs, supplemented by placement of old boomsticks by Port staff, has added some stability to the berm. Three small sand spits have formed at the end of the berm (Figures 2 and 3), creating small protected flats at about +9 to +10 ft MLLW. By early 1991 some areas along the eastern side of the embayment became eroded or covered with sand that had been transported by storms. Elevations changed from 0.5 to 2.0 ft within the embayment. Woody debris from the dredging formed thick mats (to 20 cm) over some areas inside the embayment. These mats have gradually dissipated to the point that their coverage is insignificant to epibiota.

The Corps has completed yearly surveys to assess the stability of the berm and developed a post-project bathymetric map in 1992 and 1995. Recent analysis (Nelson 1996) has indicated that the west shore of the berm has lost up to approximately 150 ft to erosion and that the berm has lost approximately 37,000 cy of sediments. Renourishment is now being considered on a 7- or 8-year cycle to maintain the integrity of the berm with minimal risk of overtopping or breaching.

Biological

Epibenthic Zooplankton Productivity

The biomass of epibenthic fauna from both the Year 3 and 5 samples was dominated by Crustacea other than key prey taxa at all locations except for the +9 ft stations inside the berm during April. Biomass of key prey taxa by month and elevation in 1995 is presented in Table 1.

In April the highest biomass of key prey taxa was found at +9 ft MLLW in the inner berm of the project area. Biomass was significantly higher ($p < 0.05$) at these stations than at the +9 ft stations at the reference areas. Also, in April no key prey taxa were taken from +9 ft MLLW in the exposed project area, nor at +7 ft MLLW inside the berm.

In May the highest biomass of key prey taxa was found at +7 ft MLLW inside the berm but the difference was not significant. No key prey taxa were taken from +9 ft MLLW in the exposed project

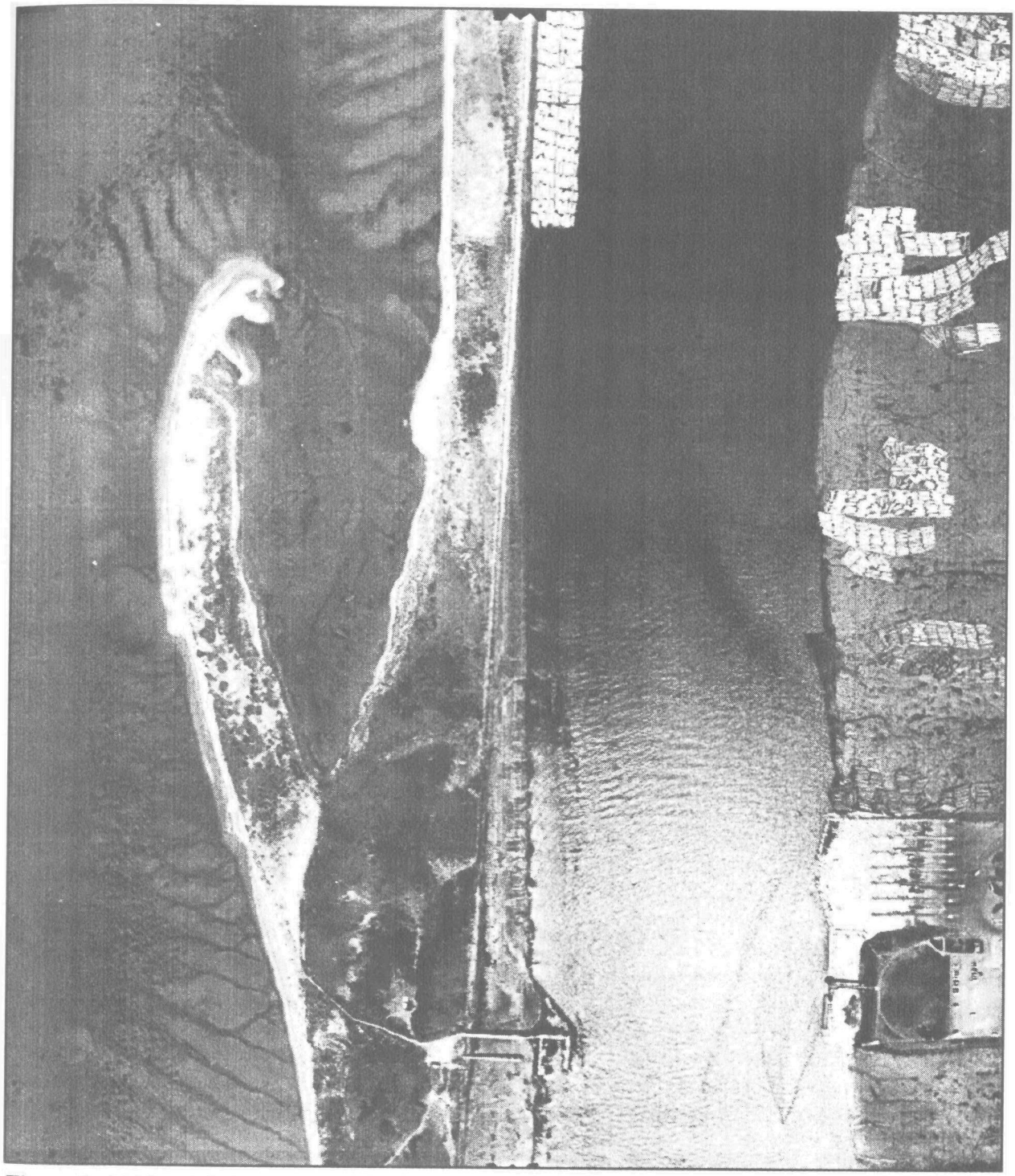

Figure 3. 1995 aerial photograph of the Jetty Island berm.

Table 1. Mean key prey biomass and the prey weighing factor (PFW) from epibenthic sampling in Port Gardner Bay, 1995.

		Biomass (grams/square meter)			Prey Weighing Factor		
	Elevation	Reference	Project area		Reference	Project area	
Month	(ft MLLW)	area	Inside	Exposed	area (= before)	Inside	Exposed
April	+9	0.0006	0.1161[a]	0.0000	1.00	90.70	0.00
	+6 or +7	0.0003	0.0000	0.0006	1.00	314.75	2.00
	+3	0.0050	—	0.0014	1.00	—	0.09
May	+9	0.0011	0.0350[a]	0.0000			
	+6 or +7	0.0000	0.0875	0.0000			
	+3	0.0139	—	0.0003			

[a]Includes both inside berm + 9 stations. The + 7-ft transect is included in the + 6-ft (MLLW) category.

area. Key prey taxa were also not taken from +6 ft MLLW in the exposed project area or reference areas.

Salmonid Habitat Evaluation

Project area prey weighting factors (PWFs) from the Year 5 data were greater than 1.0 at the inside berm +9-ft and +7-ft MLLW elevations, and at the exposed +6 ft MLLW elevation; but less than 1.0 at the exposed +9-ft and +3-ft MLLW elevation (Table 1). The high PWFs in the inside berm project area occur because of the greatly enhanced abundance of key prey taxa within the protected embayment. In Year 2 it was expected that epibenthic zooplankton productivity at +3-ft MLLW on the outside of the berm would increase as the berm continued to stabilize. This appears to be the case; the PWF at +3 ft MLLW, on the outside of the berm, in Year 2 was 0.01, and in Year 5 the PWF at this same elevation was 0.09.

The following analysis recalculates the habitat units using Year 5 PWFs, 1995 percent cover data from project planting area (inner berm), and habitat areas (acreages) derived from the Corps' May 1993 reconnaissance flight. The 1993 acreages within each tidal elevation and substrate type are included in Table 2 along with acreages that would be extant (based on pre-project bathymetry) had the berm not been built. The project area is defined as the area between two perpendicular lines running westward from the shoreline of Jetty Island (+12 ft MLLW) out to the 1993 +2 ft MLLW elevation. A portion of this area was below +2 ft before project construction. Changes of less than 1 ft elevation in this area between 1990 and 1993 probably resulted from gradual erosion of material off the berm and were not considered significant in this analysis; acreage below +2 ft in 1989 was included in the +2 to +5 ft elevation category on Table 2.

The habitat loss of approximately 13.2 ac between +2 and +12 ft (the area of berm above +12 ft; Table 2) eliminated production of prey items for juvenile salmon; to avoid a net loss of salmon feeding opportunities, one success criterion for the project was that the habitat units calculated by the above formula be at least 10 percent above the pre-project level at Year 5 (1995). The high epibenthic zooplankton productivity in the mudflats within the protected embayment generated a high PWF that in turn led to a calculation of a high number of habitat units and a high percentage (4,853 percent) of increase over pre-project conditions.

As in 1992, there is a high degree of variability in the Year 5 epibenthic zooplankton data which leaves some uncertainty regarding this analysis. However, these data are consistent with those from the Year 2 epibenthic sampling which indicated a 4,021 percent increase in habitat. Thus, it can be concluded that the project has greatly exceeded the salmonid habitat criterion.

Fish Use of the Project Area

Beach seining in 1992 and 1994 confirmed that juvenile salmon were present within the Jetty Island lagoon and at two reference areas during their outmigration from the Snohomish River system. Mean catch in the project area in 1994 (1.2 salmon/set) was somewhat lower than that from the West Jetty Island reference areas (8.3 salmon/set) and substantially lower than that from the north end of the island (71.0 salmon/set). Mean catch/set of juvenile pink salmon at the exposed sites on the berm was greater than catches in the reference areas, which are also exposed. However, catches of juvenile chum salmon were substantially higher at the reference areas than in the project area, exposed or sheltered. Regardless, the presence of juvenile salmonids at the six project sites sampled indicates use of the project area as required by the criterion for success.

In addition to chum and pink salmon, several other fish species were captured in 1994. Numbers of juvenile surf smelt (*Hypomesus pretiosus*) were much higher in the project area (49.7/set; averaging 47 mm) than in the reference areas. Catch of staghorn sculpin (*Leptocottus armatus*) was also much higher in the project area.

Vegetation

Mean percent cover and total percent cover for species sampled in the upper zone at the inner berm and reference site in 1995 are plotted on Figures 4 and 5. Similar 1995 data for the lower vegetative zone at the inner berm and reference site are plotted in Figures 5 and 6 (the lower zone is not represented at the Point). Mean percent cover for each species (*Jaumea carnosa, Salicornia virginica, Spergularia marina, Distichlis spicata, Atriplex patula,* and *Elymus mollis*) and mean total percent cover for upper and lower zones in the inner berm were compared with upper and lower zones at the reference site (Table 3). In the upper zone, there

Table 2. Pre- and post project habitat area (acres) and quality (habitat units).*

	Mud								Sand												Summary					
	Without Berm		With Berm (1995)						Without Berm						With Berm (1995)						Total Acres			Habitat Units		
Elevation (MLLW)	Area	HU	Area	Slope	Cover	PWF	RIF	HU	Area	Slope	Cover	PWF	RIF	HU	Area	Slope	Cover	PWF	RIF	HU	Before	After	Change	Before	After	Change
+ 12 to + 10 ft	0.0	0.0	3.1	0.5	0.0	90.7	1.00	139.8	3.1	0.5	0.0	1.0	1.0	1.5	2.7	0.5	0.0	0.0	1.0	0.0	3.1	5.7	2.6	1.5	140	138
+ 10 to +8 ft	0.0	0.0	6.5	0.5	0.0	90.7	1.01	304.7	4.9	0.5	0.0	1.0	1.0	2.4	2.4	0.5	0.0	0.0	1.0	0.0	4.9	8.9	4.0	2.4	305	302
+ 8 to +5 ft	0.0	0.0	9.6	0.5	0.0	315.0	1.00	1507.3	24.1	0.5	0.0	1.0	1.0	12.1	21.0	0.5	0.0	2.0	1.0	21.0	24.1	30.6	6.5	12.1	1528	1516
+5 to +2 ft	0.0	0.0	0.0	0.5	0.0	0.0	1.00	0.0	47.7	0.5	0.0	1.0	1.0	23.8	21.3	0.5	0.0	0.2	1.0	2.0	47.7	21.3	-26.3	23.8	2	-22
Total +12 to +2 ft	0.0	0.0	19.2					1951.7	79.7					39.9	47.4					23.0	79.7	66.5		39.9	1975	
Area gain (loss)																							-13.2 (acres)			
Habitat units gain (loss)																										1935
Percent change																								-16.6%		4853%

*The project area is defined vertically by the 1989 +12-ft contour and the 1993 + 2-ft contour. A portion of the project area that is currently above +2-ft MLLW was below that elevation before the berm was constructed in 1989. The difference of 13.2 acres represents area filled to above +12-ft to form the berm.

Key

HU (Habitat Units) = Acres x PWP x (Cover factor + Slope factor) x RIF.

PWF = Prey Weighting Factor, local mean key prey species biomass/reference area key prey species biomass at same elevation. Data from April and May 1992 sampling (see Table 6 in Pentec 1993).

Cover and slope factors are defined in Pentec 1993.

RIF = Relative Regional Importance Factor, a recognition that some habitats are more valuable because of past losses. Assigned a value of 1.3 for saltmarsh areas.

Assumptions

RIF of 1.3 and cover factor of 0.2 assigned to 0.04 acres between +10 and 12 ft with average vegetative cover of 25 percent.

RIF of 1.3 and cover factor of 0.4 assigned to 0.16 acres between +8 and 10 ft with average vegetative cover of 59 percent.

Table 3. Mean percent cover for individual plant species August, 1995.

		Reference		Berm				
				Inner			Point	
	Taxon	Mean	SD	Mean	SD	p	Mean	SD
Upper Distichlis/Elymus	Jaumea carnosa	21.45	18.91	25.19	31.59	0.6699	0.12	0.44
	Salicornia virginica	8.18	10.42	9.69	15.19	0.7380	18.20	22.18
	Distichlis spicata***	12.38	8.77	32.75	23.57	0.0003	1.46	3.80
	Atriplex patula	5.45	6.90	3.47	4.23	0.3607	0.56	1.96
	Elymus mollis	17.73	18.24	23.50	31.59	0.414	---	---
	Total cover**	68.20	25.27	97.41	35.86	0.0078	31.84	21.81
							Unplanted Area	
							Mean	SD
Lower Jaumea/Salicornia	Jaumea carnosa*	69.35	18.06	83.75	16.75	0.0170	11.13	17.03
	Salicornia virginica	24.15	15.61	13.38	24.69	0.1170	26.25	22.70
	Spergularia marina*	0.10	0.31	0.63	0.96	0.0269	5.25	8.51
	Distichlis spicata**	11.60	6.09	5.31	3.14	0.0011	6.38	9.40
	Total cover	109.40	11.39	87.25	33.23	0.4101	49.88	29.42

*$p \leq 0.05$
**$p \leq 0.01$
***$p \leq 0.001$

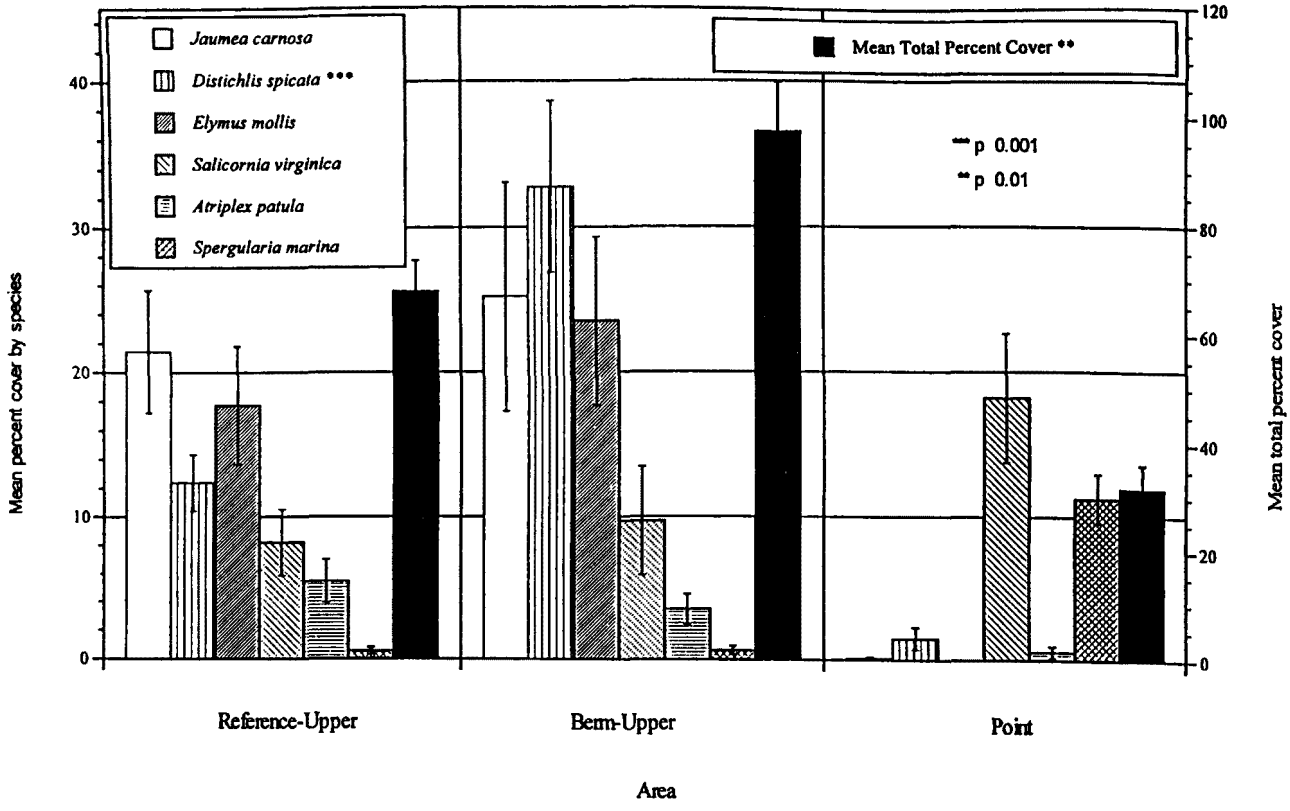

Figure 4. Key species present in the *Distichlis/Elymus* zone, 1995.

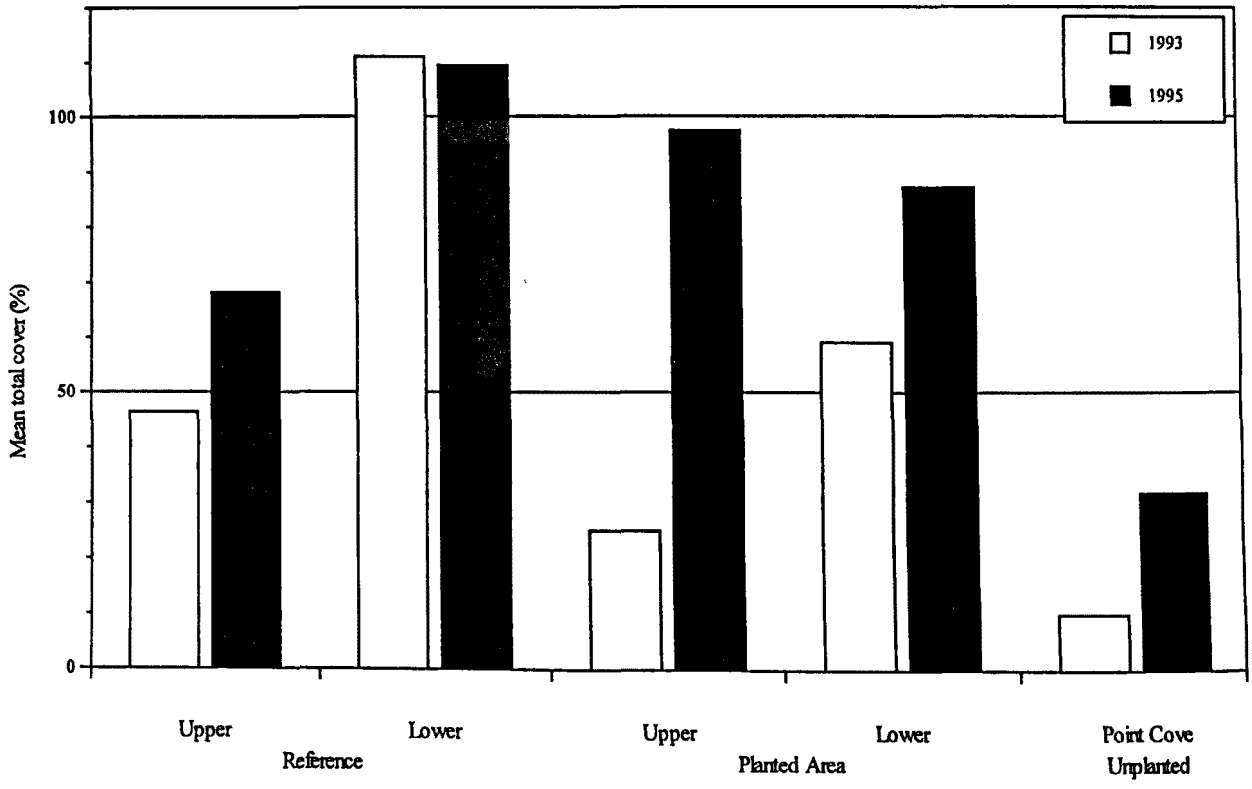

Figure 5. Mean total vegetative cover (%) in late summer 1993 and 1995.

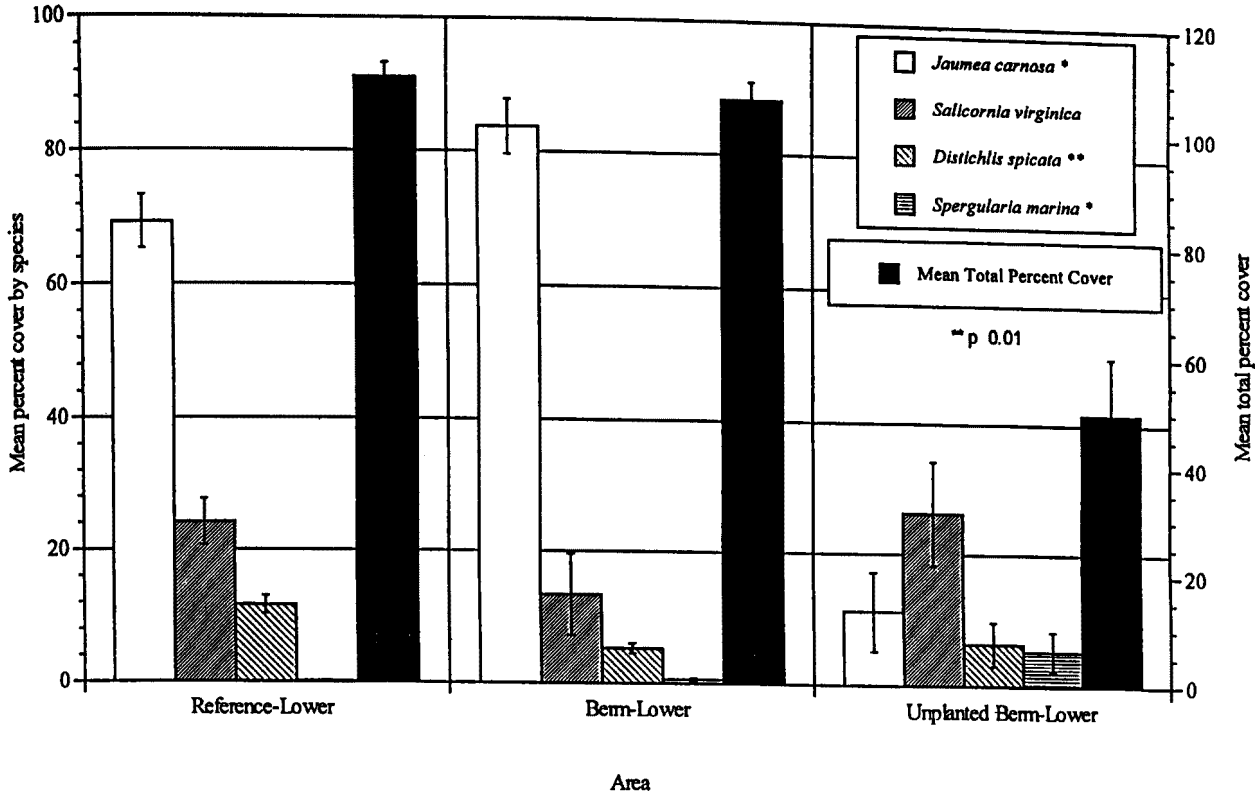

Figure 6. Key species present in the *Jaumea/Salicornia* zone, 1995.

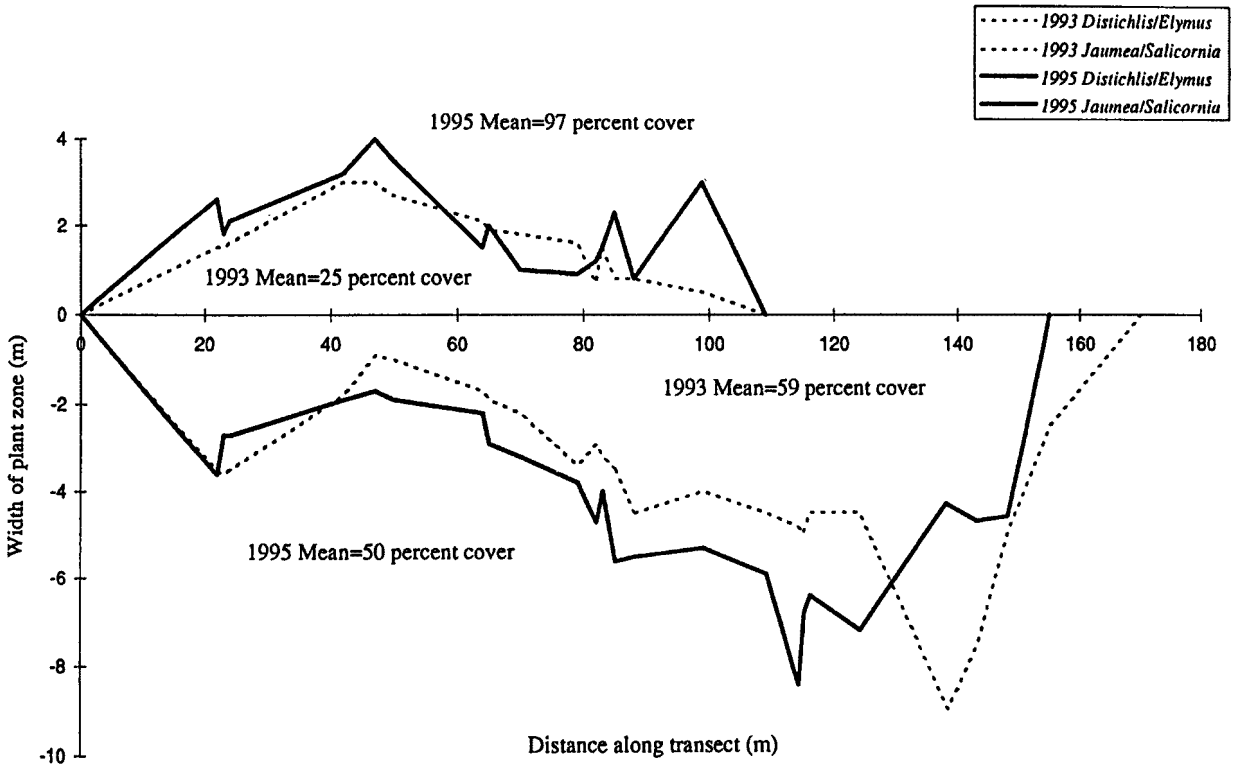

Figure 7. Width of upper (*Distichlis/Elymus*) and lower (*Jaumea/Salicornia*) vegetative zones in the inner berm.

was no significant difference in percent cover of *Jaumea carnosa, Salicornia virginica, Atriplex patula,* or *Elymus mollis* between the inner berm and the reference site. In the lower zone no significant difference was found in percent cover of *Salicornia virginica* between the inner berm and the reference site in 1995. However, several other taxa differed significantly in the lower zone. Total percent cover of all taxa combined differed significantly in the upper zone, but not the lower zone. Changes in total plant cover between 1993 and 1995 are shown in Figure 5.

The 1993 and 1995 width of the upper and lower vegetative zones, *Distichlis/Elymus* and *Jaumea/ Salicornia*, respectively, are plotted on Figure 7. Over a distance of 99 m along the upper transect, a total of 0.05 ac was represented by the *Distichlis/ Elymus* zone in 1995; mean percent cover over this area was 97 percent. Along the lower transect (148 m), a total area of 0.15 ac was represented by *Jaumea* and *Salicornia*; mean percent cover over this area was 49.9 percent.

Discussion

Physical

Before the berm was created, no mud substrate occurred on the west side of Jetty Island. A total of 76.7 ac of sandy habitat was present between the elevations of +2 ft and +12 ft MLLW. A portion (3 ac) of the present-day project area was below +2 ft MLLW before the berm was constructed. Creation of the berm placed the post-construction 2-ft contour approximately 394 ft west of the pre-project 2-ft contour in some places.

When the berm was originally created, it occupied about 15 ac above +12 ft MLLW; currently, only about 13.2 ac remain above this elevation due to erosion of the berm. Thus, the project has resulted in the loss of 13.2 ac of intertidal sandflat.

Biological

As expected, depositional conditions within the 19.2-ac mudflat inside the berm have greatly favored the production of invertebrates including epibenthic prey of juvenile salmonids. Based on the 1995 epibenthic zooplankton abundances and the 1993 bathymetry, the project has greatly exceeded the criterion of providing > 110 percent of pre-project habitat units for juvenile salmonids. Presence of juvenile salmon within the lagoon in 1992 and 1994 has met the criterion for demonstration of salmonid use of the area.

Although no infaunal monitoring was required under the Implementation Plan, qualitative observations confirmed a rapid colonization of the projected embayment formed by the berm. Early in the spring of 1990, ghost shrimp (*Callianassa californiensis*) substantially colonized the mudflat below about +8 ft MLLW. Since 1990, qualitative excavations within the lagoon have shown a continual increase in the apparent density and diversity of the infaunal community with polychaetes, crustaceans, and molluscs all contributing to this assemblage. By summer of 1992, small softshell clams (*Mya arenaria*) were established along the firmer muddy sand along the inside of the berm between about +7 and +8 ft MLLW. Although elevations within the lagoon are relatively high (i.e., > 6 ft MLLW) for Dungeness crab, some use is indicated by an observation of a small (approximately 3 cm) crab moulting in the moist mud of the lagoon.

The abundance of shorebirds, smelt, and flatfish as well as the juvenile salmon feeding on benthic and epibenthic prey within the lagoon demonstrate that the project area is functioning as an important part of the Port Gardner ecosystem.

Natural colonization of vegetation on the top of the berm has been remarkable. Seedlings of some 12 to 15 species became established in the spring of 1990, and the vegetative community has continued to develop rapidly with species such as bighead sedge and beach peavine that are also dominants on drier areas of the main island. During an August 1994 survey, we collected over 40 species of vascular plants on the center of the berm (e.g., above +12 ft MLLW). The berm now supports several shrub and tree species (e.g., Scot's broom and black cottonwood [*Populus balsamifera trichocarpa*]) that have grown to more than 3 m in height.

Dunegrass plantings in 1990 were very successful and continue to expand beyond and between the planted rows along the west side of the top of the berm. A majority of this area planted with dunegrass has eroded away, and it appears that the rhizome mat of the grass is only marginally helpful at retarding this erosion. A similar erosion of the dunegrass community was occurring prior to berm construction along much of the northwest shore of the island. The dunegrass plantings and natural colonization by other

species have significantly retarded wind erosion on the top of the berm, however. Natural colonization of dunegrass from waterborne seeds and rhizomes from eroding areas of the island to the north has developed into a similarly dense growth along the lower elevations (+12 to +14 ft MLLW) on the east side of the berm.

The nearly complete lack of success of the 1990 experimental plantings of saltmarsh plants in the lagoon contrasts sharply with the success of the 1991 plantings and subsequent natural colonization. Quantitative sampling in late summer of 1993 and 1995 demonstrated considerable success of species planted in 1991 and in some areas of natural colonization. The lack of a statistically significant difference in cover in some of the dominant species between the area inside the berm and the reference marsh (*Jaumea, Distichlis, Elymus* in the upper zone; *Jaumea, Salicornia* in the lower zone) is highly encouraging. Over much of the planted areas of the lower zone, the combined cover of *Jaumea* and *Salicornia* exceeded 100 percent, and *Jaumea* was in bloom in early September 1993. Differences in other species and in total cover remain, however, and it is unlikely that the vegetative communities in the two areas (berm inside and reference marsh) will ever be identical because of subtle environmental differences, most notably in salinity regime. However, it is expected that the healthy and expanding saltmarsh fringe on the inside of the berm will continue to develop and function as a significant and beneficial habitat feature that would otherwise be lacking on the west side of Jetty Island. A disturbing finding was the discovery in 1994 of a large number of plants of the invasive exotic *Spartina ?anglica*; the Port has initiated an eradication program.

Sampling of fish and invertebrate use of the lagoon at high tide demonstrates that several functions of a natural saltmarsh (fish use, feeding, refuge; invertebrate production) are also provided by this fringe marsh. Qualitative observations also suggest that important marsh functions of use by shorebirds and waterfowl as well as organic detrital production are also occurring.

Beyond its documented use by immense numbers of juvenile surf smelt and lesser numbers of salmon and other fish, it seems likely that the greatest single ecological benefit of the berm and lagoon has been its suitability for a variety of ecological functions of birds. Virtually all of the bird species reported in the baseline studies have been observed to use the berm and/or lagoon, often in greater numbers than seen elsewhere on the island (Pentec Environmental, Inc. 1996). Construction of the berm added three important bird habitats not previously found on the west side of Jetty Island: unvegetated sandy uplands, saltmarsh, and mudflats. Each of these habitats is used differently by birds, but collectively they constitute a habitat complex similar to some of the better habitats found in more natural estuaries. Although the unvegetated sandy uplands have been reduced in area by vegetative colonization since construction, the two lower spits that have formed on the north end of the berm continue to offer this type of habitat.

Summary

Monitoring of the Jetty Island berm project and evaluation of performance standards through Year 5 indicate that the project has met the criteria established in the Implementation Plan. Monitoring has shown that the area is being used by juvenile salmonids and has met the criterion for demonstrated use in at least two years.

The productivity of the protected embayment exceeded project goals for salmonid habitat by 4,000 percent. As anticipated, the lower energy regime and mud substrate of the protected embayment is providing a more productive habitat for juvenile salmon prey resources than exists on the more exposed sandy beaches outside the berm. Monitoring has shown that the area is being used by juvenile salmonids and has met the criterion for demonstrated use in at least two years. The abundance of juvenile salmon prey resources has enhanced the project area habitat value for juvenile salmonid feeding and for use by juvenile surf smelt, an important forage fish. This gain in productivity within the embayment has more than offset productivity lost by filling 13.2 ac of previously intertidal habitat to greater than +12 ft MLLW.

Year 5 monitoring showed that the percent cover of all plants and of selected plant species in the planted portion of the protected embayment did not differ significantly from those in a reference marsh although other taxa did differ significantly.

The ATAC is considering a proposal by the Port and the Corps to renourish the berm in 1997 to maintain the increased habitat values created.

Acknowledgments

The Port of Everett and the U.S. Army Corps of Engineers, Seattle District, have provided logistical, technical, and field support for this project. Thorough review of project plans and the habitat evaluation model used by the ATAC greatly improved the technical quality of the monitoring program and results. We would like to acknowledge Wilber E. Ternyik of Wave Beach Grass Nursery in Florence, Oregon, who provided the nursery plants; the many state and federal agency representatives who volunteered to assist with the transplanting; and Dr. Ron Thom (Battelle Northwest) for the original design of the vegetation experiment and for technical advice on habitat development in urban estuaries.

Literature Cited

Edgington, E. S. 1987. Randomization tests, 2nd edition. Marcel Dekker.

Nelson, E. 1996. Jetty Island protective berm erosion analysis and renourishment plan. Draft manuscript by the U.S. Army Corps of Engineers, Seattle District. April 5, 1996.

Pentec Environmental, Inc. 1994. Beneficial use of dredged material, Jetty Island habitat development demonstration project. Years 3 and 4 Monitoring Report. Report to the Port of Everett, WA.

Pentec Environmental, Inc. 1996. Beneficial use of dredged materials, Jetty Island habitat development demonstration project. Year 5 Monitoring Report. Report to the Port of Everett, WA.

U.S. Army Corps of Engineers. 1989a. Final environmental assessment, Jetty Island habitat development. Seattle District, Seattle, WA.

U.S. Army Corps of Engineers. 1989b. Implementation plan for habitat development project using dredged material at Jetty Island, Everett, WA. Seattle District, Seattle, WA.

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Restoration and Management of the Salmon River Estuary

Chris McDonald[1]

Abstract: **Located along the north-central Oregon coast, the Salmon River estuary and surrounding lands were set aside by Congress as the Cascade Head Scenic Research Area in 1974. This is the first Scenic Research Area designated in the United States. The goals of the research area are to protect scenic values, provide research opportunities and within the 1085 acre estuary return lands to a more natural condition.**

The U. S. Forest Service bought 80 acres in 1974 to help achieve restoration goals within the estuary. Much of the original estuary had been diked and converted to pasture land or recreation use changing hydrologic and ecological processes. Restoration began in 1978 when dikes were breached within the 80 acre pasture to re-establish a saltmarsh. Studies of the restored saltmarsh began before dike removal and continue today. The Salmon River estuary has the longest continuous record of restoration research of any estuary on the Oregon coast.

Additional land has been acquired and restored throughout the years. The Forest Service owns over 80 percent of the estuarine lands and has restored 230 acres to salt marsh through dike removal. The value of restoring this estuary has increased its importance as other bays and estuaries along the Oregon coast have decreased in natural productivity and diversity. The estuary provides other opportunities for research such as studying estuarine dependence of anadromous fish stocks. Future restoration plans include additional removal of dikes, which will allow 120 acres to return to a more natural condition.

Western Oregon and Washington are known for their scenic beauty, productive forests and riparian ecosystems. Estuarine wetlands are an important but small part of the Northwest's lands and are increasingly being recognized for their contribution to healthy ecosystems. Since European settlement, most of these estuarine lands have decreased in size and productivity as lands have been diked, drained and converted to other uses. The Salmon River estuary is one of the least disturbed along the Oregon coast. Since 1974, the management goal under the U.S. Forest Service Administration has been to restore the estuary to a more natural condition. This 1,085 acre estuary is one of the state's smallest; however, it includes one of the largest wetland restoration projects in Oregon.

The Salmon River estuary is located along the north-central Oregon coast approximately 7 miles north of Lincoln City (Figure 1). The estuary is part of a 75-square-mile watershed surrounded by mostly coniferous forest. Major land uses are forestry, agriculture and recreation. The estuary and associated wetlands are entirely within the 9,670-acre Cascade Head Scenic Research Area (CHSRA), part of the Siuslaw National Forest in the Hebo Ranger District.

People have been part of the Salmon River estuary's landscape for centuries. Perhaps as long ago as 500 years, the Salmon River became home to the Nechesne Indians (Beckham, 1975). Numerous cultural sites can be found along the banks of the river as the Nechesne culture was oriented toward the ocean. Legends of these Indians indicate that they used Cascade Head, a promontory rising above the sea to the north of the Salmon River's mouth, as a vigil site for individuals seeking power and visions.

The estuary and surrounding area began to be populated by European settlers in the mid-1800s. Early settlers cleared the land to raise cattle and grow vegetables. Major changes came to the estuary in the 1920s when the Salmon River Highway and Oregon Coast Highway were constructed. With transportation available, farmers started dairies, and residents of the Willamette Valley and Portland area had access to the coast to fish and enjoy the scenic beauty. Since World War II, the area has grown in popularity.

Historic photos of the estuary show major changes began in the early 1960s when dikes were constructed (Figures 2 and 3). The dikes kept tidal waters from flooding the salt marshes so that they could be used for pastures. In 1961, U.S. Highway 101 was rerouted from a narrow winding road through the forest to a

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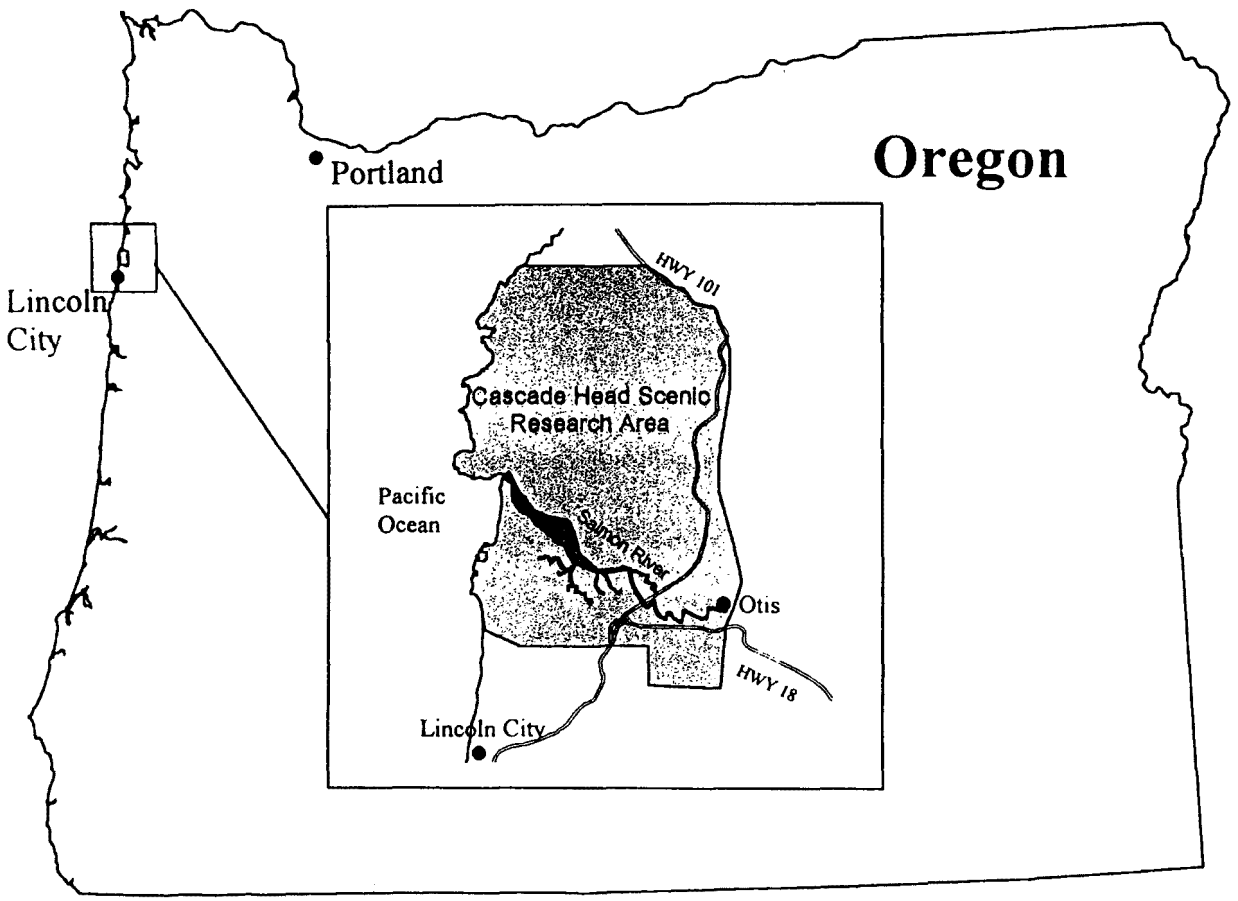

Figure 1. Location of the Salmon River estuary within the Cascade Head Scenic Research Area.

route nearer the coast. The highway crosses the upper portion of the estuary and functions much like a dike. During highway construction, Salmon Creek, a tributary to the Salmon River, was diverted from its natural channel into a ditch. In the early 1970s, an entrepreneur built the Pixieland amusement Park on 55 acres of diked marshland covered with fill. A mobile home park and boat launch area followed elsewhere in the estuary. The Salmon River area appeared to be destined in the late 1970s for development (Figure 3).

Even though much of the estuary was being developed, much of the area remained in a nearly natural state. A small group of residents in the surrounding area became interested in preserving the estuary and adjacent lands from further development. In 1965, they purchased Cascade Head and donated the headland to The Nature Conservancy. This same group of concerned land owners then requested Representative Wendell Wyatt and Senator Bob Packwood to consider formal protection of the estuary and surrounding lands. In 1974, President Gerald Ford signed Public Law 93-535 making the Cascade Head and surrounding lands the first (and only) scenic research area in the United States.

The U.S. Forest Service administers this specially designated parcel. As stated in the legislation, the general management objectives of the CHSRA are *"...to provide present and future generations with the use and enjoyment of certain ocean headlands, rivers, streams, estuaries, and forested areas, to ensure the protection and encourage the study of significant areas for research and scientific purposes and to promote a more sensitive relationship between man and his environment..."*

A Management Plan and EIS were approved in 1976. The writers of the management plan recognized

that the value of this estuary would increase in importance as other bays and estuaries along the Oregon Coast decreased in size and productivity. Both long- and short-term goals were agreed upon at that time. The long-term goal for the estuary is *"revitalization and restoration of the Salmon River estuary and its associated wetlands to a functioning estuarine system free from the influence of man."* The interim management goal is to study the issue of dike removal while allowing existing agricultural use to continue.

The Act states that changes in land ownership would be necessary to meet legislative goals and it specified how lands could be acquired. In 1974, there were no public lands in the estuary and only 35 percent of the estuary remained in a semi-natural state. As landowners became aware of management goals they sold their lands to the U.S. Forest Service. Most lands were acquired from willing sellers, however, some land owners were reluctant to sell, and those lands were condemned. Money for the land acquisitions came from the Land and Water Conservation Fund, a federal fund that dedicates windfall oil profits to federal property acquisition.

The U.S. Forest Service bought 80 acres in 1974 to assist restoration goals within the estuary. Restoration began in 1978 when two-thirds of the outer dikes were breached within a 52-acre pasture. A research project conducted by a Ph.D. student focused on collection of baseline vegetation and soils data before dike breaching, methods of restoring hydrologic connectivity, and a monitoring strategy to follow the effects of breaching the dikes (Mitchell, 1981). Vegetation changes occurred almost immediately as pasture grasses were replaced by salt marsh species. No planting was done. Diking caused the pasture to subside in elevation as soils became compacted and dewatered. The restored surface land is gradually regaining its elevation. Seventeen years of data have been collected on the changes in elevation, vegetation, accretion and elevation, and soil salinity. The Salmon River estuary has the longest continuous record of restoration research of any estuary on the Oregon coast.

Additional estuarine land has been acquired and restored throughout the years. The U.S. Forest Service now owns over 80 percent of the estuarine lands and has restored 230 acres to low salt marsh through dike removal. The value of restoring this estuary has increased its importance as other bays and estuaries along the Oregon coast have been altered. Future restoration plans include additional removal of dikes and tidegates. Several sites west of Highway 101 are being considered for dike removal in 1996. They include a two acre pasture near the mouth of the river, an eight acre pasture with partially breached dikes along the north shore and a 55-acre pasture immediately west of Highway 101. Other long-term restoration goals are to acquire additional lands, reconnect Salmon Creek under Highway 101 and restore hydrologic conditions east of Highway 101 where wetlands have been drained and streams diverted from original channels.

Almost 22 years after the CHSRA legislation and 19 years after the Forest Service management plan was approved, it is recognized that the stringent goal of restoring the Salmon river estuary to a state "... *free from the influences of man...*" may not be entirely feasible. In many cases previous conditions are unknown. Also, it is not realistic to assume that all manmade features will be removed. However, restoration within the salt marsh has been successful. Monitoring started by Diane Mitchell and continued by Dr. Robert Frenkel (Dept. of Geosciences, Oregon State University) and his graduate students has shown that tidal channels are deepening, salt marsh vegetation has been re-established, and sediment accretion is occurring.

Because of its management goals and history, the Salmon River offers opportunities for research. With 57 percent of the Salmon River functioning as a near natural system, it provides the best and longest data set for salt marsh restoration on the Pacific Coast and a prime area for studying the fresh and saltwater interface for anadromous fish. For more information on possible involvement in research in the Salmon River estuary, contact the U.S. Forest Service Pacific Northwest Forest Sciences Laboratory, Corvallis Oregon.

Literature Cited

Mitchell, Diane L. 1981. Salt Marsh Re-establishment Following Dike Breaching in the Salmon River Estuary, Oregon. Corvallis, OR: Oregon State University. 171 pp., Ph.D. thesis.

Beckham, Stephen D. 1975. Cascade Head and the Salmon River Estuary: A History of Indians and White Settlement. Report prepared for the U. S. Forest Service.

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Figure 2. Upper Salmon River Estuary, 1952. Prior significant construction.

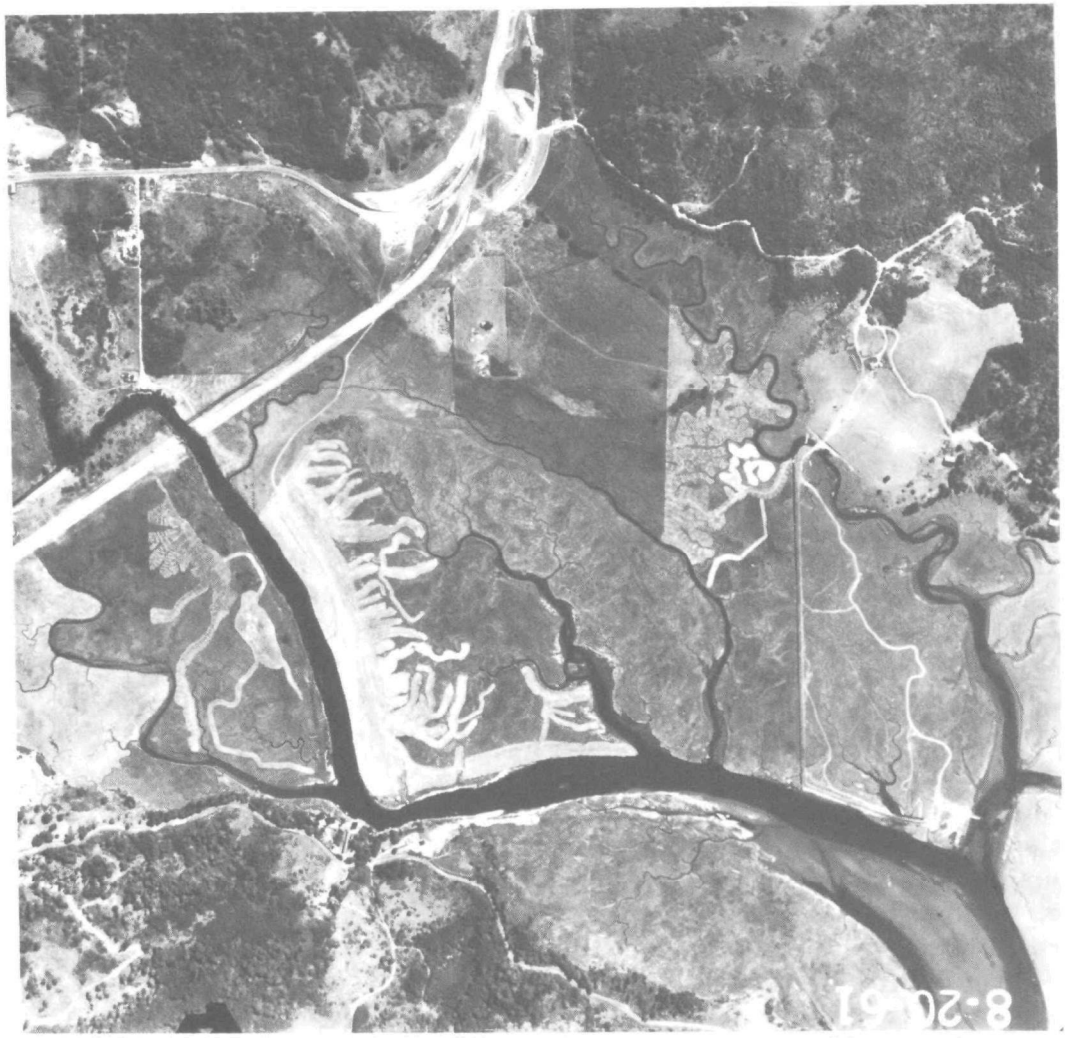

Figure 3. Upper Salmon River Estuary, 1961. Note extensive diking and highway construction.

Salmon River Salt Marsh Restoration in Oregon: 1978-1995

Robert E. Frenkel[1]

Abstract: **The U.S. Forest Service initiated the Salmon River estuary salt marsh restoration in 1978 by removing dikes from a 22 ha pasture. In 1987, a nearby 58 ha pasture was likewise returned to tidal circulation. Vegetation composition, biomass, elevation, sediment accretion, salinity, and substrate texture have been monitored for 15 years since 1978. Adjacent natural salt marshes serve as research controls. Species composition changed rapidly after dike breaching. Pasture species, killed by saltwater, were displaced initially by annual marsh colonizers (*Cotula coronopifolia* and *Spergularia marina*). After 4 years, regionally common native species (*Carex lyngbyei*, *Salicornia virginica* and *Distichlis spicata*) began to replace the annuals and today dominate the restored marshes. Reestablished low marsh differs in species composition from the high marsh controls. Diked pasture elevations were ca. 35 cm lower than adjacent control marsh elevations, caused by subsidence. Subsided restored surfaces increased in elevation at a greater rate than control elevations. Sediment-clogged pasture creeks deepened 20-60 cm upon tidal reconnection. Substrate in the restored marsh accreted ca. 4 cm per decade. Estimated above ground primary productivity based on biomass measurement of the pasture was comparable to that of natural salt marsh controls (ca. 1,200 g/m2/yr) and to other Pacific Northwest high salt marshes. Restored marsh productivity was much greater (2,300 g/m2/yr) than controls. Restoration trends differ between the two restored units. The 58 ha unit lacked an intact drainage system characteristic of the 22 ha unit and surface ponding retarded restoration. The U.S. Forest Service restoration goal was to return diked pasture to its presettlement condition. Restoration was successful in reestablishing apparently functioning salt marshes that are integrated into the estuary; however, long term prospects for complete restoration to pristine conditions are more problematic.**

Introduction

Popular, government and academic interest in wetland and riparian ecosystem restoration has burgeoned in the past decade as we become more cognizant of the enormous losses and degradation of these critically valuable biotic systems (NRC 1992, Mitsch and Gosselink 1993, National Academy of Sciences 1995). Small in proportion to the earth's other impaired and impoverished landscapes, wetlands and riparian systems are recognized as having over arching importance in regulating climate, maintaining water quality, stabilizing hydrologic regimes, and providing crucial wildlife habitat (Mitsch and Gosselink 1993).

Despite the fervor and concern over wetland restoration, there have been few large scale wetland restoration projects conducted over an extended period of time. As the preferred compensatory mitigation strategy, wetland restoration projects have typically been on privately owned land, small in extent and poorly planned, executed and monitored (Kusler and Kentula 1990).

When Congress designated the Cascade Head Scenic Research Area (CHSRA) in 1974 (P.L. 93-535), a unique opportunity arose to restore and monitor a substantial salt marsh area on public land along the north central Oregon coast. The U.S. Forest Service, which administers the specially designated CHSRA, was directed by Congress to restore *". . . the Salmon River estuary and its associated wetlands to a functioning estuarine system free from the influences of man . . . . to its condition prior to the existing diking and agricultural use"* (U.S. Forest Service 1976). About 75 percent of the Salmon River salt marshes had been diked to create pasture, mostly in the early 1960s. Conversion of coastal marshes has been common in Oregon where of the state's original salt marsh area, about 46 percent has been diked or filled since the early 19th century; most conversion has been to pasturage (Boulé and Bierly 1987). Besides accomplishing the Forest Service's goal of returning pasture to near-natural estuarine conditions, restoration of the Salmon River salt marshes by dike breaching serves

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as a long-term demonstration project that should be helpful to wetland managers in planning other coastal restoration programs and projects.

Study Site and Objectives

The Salmon River project consists of two units restored by dike removal, (1) a 21 ha pasture on the north shore was returned to tidal circulation in 1978, and (2) an additional 63 ha of pasture on the south shore were restored in 1987. For comparison, both restored pastures were selected so that they are flanked by relatively intact natural salt marshes that served as "controls" (Figure 1). Restoration was deliberate but passive in that dikes were breached allowing reestablishment of tidal circulation without planting, seeding or surface grading. Together with graduate students, I monitored these restoration sites for 17 and 8 years respectively. Throughout the remainder of the paper I refer to the north shore site as "Mitchell Marsh" and the south shore as "Y-Marsh."

I address four objectives: (1) will salt marsh vegetation reestablish in the absence of planting or seeding; (2) is it possible to restore a diked pasture to original high marsh conditions; (3) what is the rate of restoration; and (4) is the pattern of restoration similar in Mitchell Marsh and Y-Marsh?

Diane Mitchell initiated the north shore study in 1978 and established 115 one-meter square plots along 20 transects in the restored and control units. At each plot she sampled vegetation cover, biomass and elevation before and after dike removal in 1978 and 1979 respectively (Mitchell 1981). Together with Mitchell and Janet Morlan, I continued sampling vegetation cover in 1980, 1981, 1982, 1984, 1988, and 1993. Additionally, I sampled interstitial soil salinity, soil texture, soil organic matter, and accretion above a sand layer placed on the pasture surface in 1978. Mitchell also sampled elevations across 47 cross-sections along 8 marsh creeks. With Morlan, I completely resampled the restored marsh in 1988, 10 years after dike breaching (Morlan 1991, Frenkel and Morlan 1991).

On the south shore in the Y-Marsh, I established 57-meter-square plots along 6 transects and sampled vegetation cover in 1987 prior to dike breaching and again in 1988, 1990, 1991, 1992, 1993, and 1995. I surveyed plot elevations in 1988.

Salt Marsh Reestablishment

Composition and productivity

Upon dike removal, saline (30 to 10 ppt) tides flooded the pasture, stressing upland plants, and within a year, killing them. Detached litter and dead plants prevailed for several years after dike breaching. Change in selected species composition is shown in Figure 2 for Mitchell Marsh from pasture in 1978 to extensive low salt marsh in 1993. We considered three species groups: upland species (e.g., *Holcus lanatus* and *Ranunculus repens*) that were killed immediately; residual species (e.g., *Agrostis alba* and *Potentilla pacifica*) that diminished in cover; and both ephemeral and persistent colonizing species (e.g., *Carex lyngbyei, Cotula coronopifolia, Distichlis spicata* and *Salicornia virginica*) that invaded and increased in cover. Bare ground increased to a maximum the second year after dike breaching after litter was washed away, and diminished rapidly thereafter. Six years after restoration, two low marsh assemblages dominated the Mitchell site, an extensive brackish *Carex lyngbyei* community and a more limited *Salicornia virginica-Distichlis spicata* community in more saline areas.

A similar pattern of reestablishment occurred on the more recently restored Y-Marsh, except that bare ground and litter dominated and persisted for several years after dike removal. Colonization by salt marsh species was slow and initially characterized by annual ephemeral species (*Cotula coronopifolia, Puccinelia pumila* and *Spergularia marina*). By 1993, two assemblages were prominent, a *Deschampsia cespitosa* community and a *Salicornia virginica-Distichlis spicata* community.

During the same interval, species composition in the relatively intact control sites on both the Mitchell Marsh and Y-Marsh changed very little (Figure 3).

I estimated above ground net primary productivity from sorted biomass in the Mitchell Marsh following the method of Kibby et al. (1980). Pasture productivity was similar to that of the control high marsh in 1978, about 1000 g/m^2-yr. After dike breaching, biomass at first diminished and then gradually increased to 2300 g/m^2-yr in 1988, more than double the value of either the intact high marsh or the pasture.

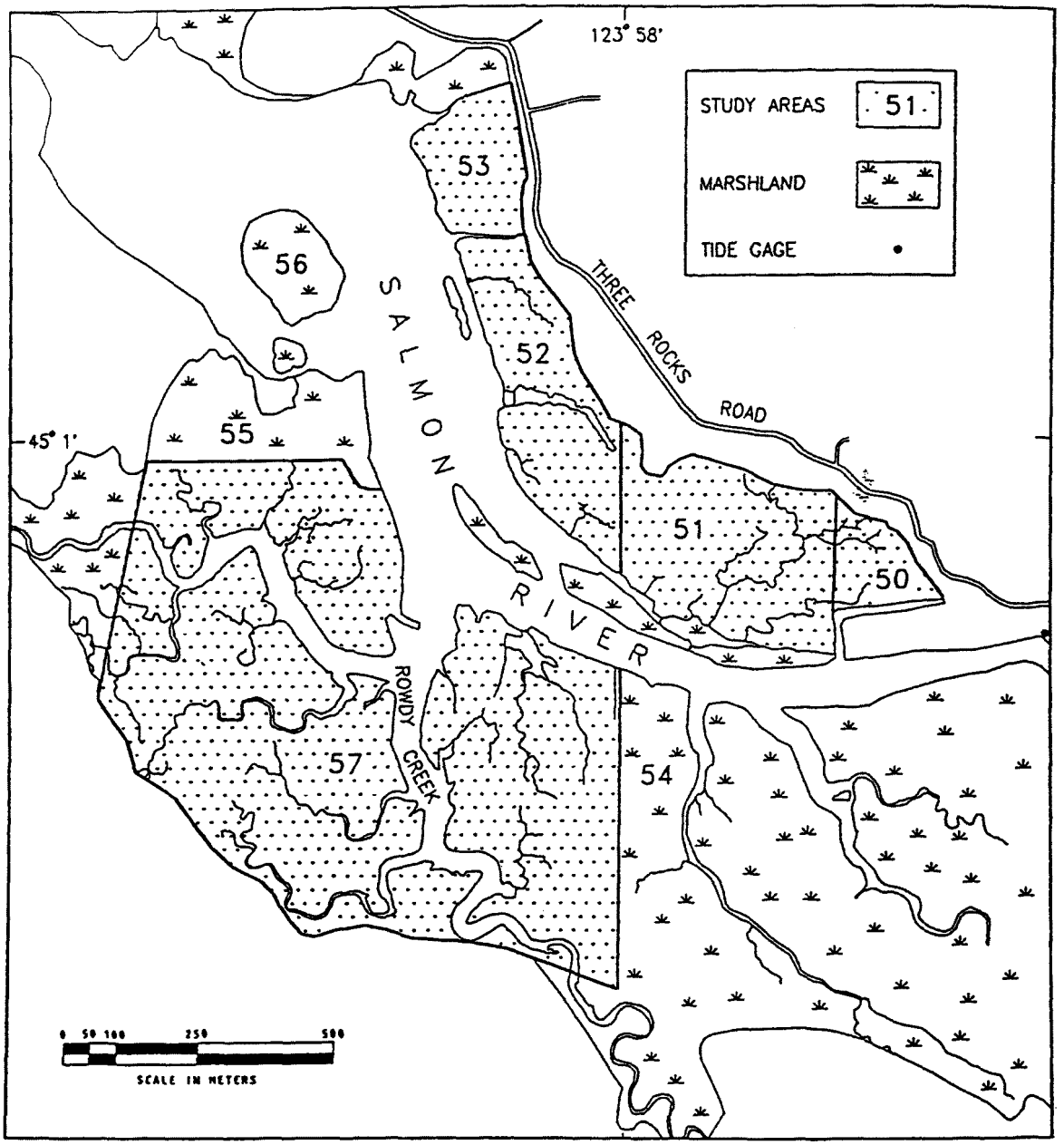

Figure 1. Location of restoration and control area in the Salmon River estuary. Mitchell Marsh: Restoration area 51 and 52, Control area 50 and 53; Y-Marsh: Restoration area 57, Control area 54.

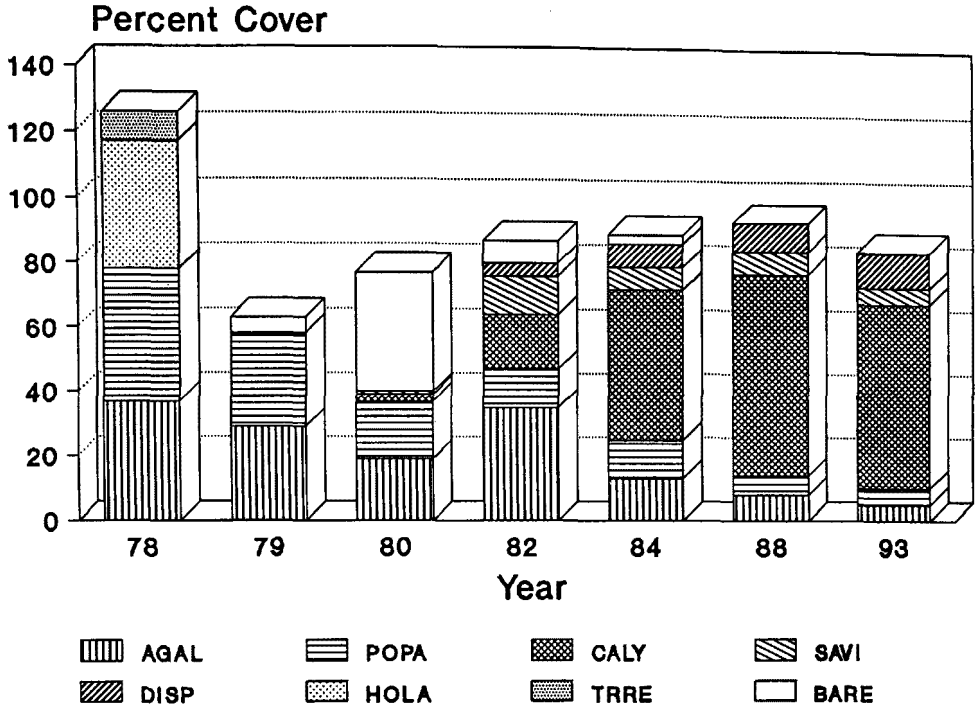

Figure 2. Change in percent cover of principal plant species in the Mitchell Marsh restoration site, 1978-1993 in the Salmon River estuary.
AGAL = *Agrostis alba*, POPA = *Potentilla pacifica*, CALY = *Carex lyngbyei*, SAVI = *Salicornia virginica*, DISP = *Distichlis spicata*, HOLA = *Holcus lanatus*, TRRE = *Trifolium repens*, and BARE = surface not vegetated by vascular plants.

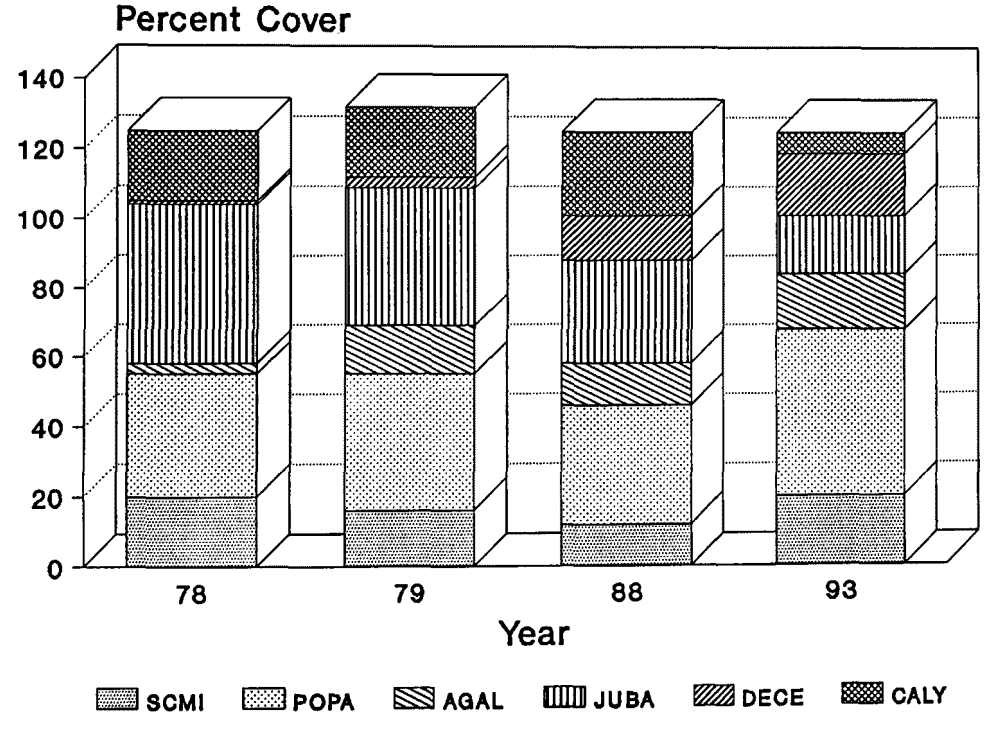

Figure 3. Change in percent cover of principal plant species in the Mitchell Marsh control site, 1978-1988 in the Salmon River estuary.
SCMI = *Scirpus microcarpus*, POPA = *Potentilla pacifica*, AGAL = *Agrostis alba*, JUBA = *Juncus balticus*, DECE = *Deschampsia cespitosa*, and CALY = *Carex lyngbyei*.

Environmental Change

By environmental change I refer to variation in controls that affect marsh restoration including, surface elevation that determines tidal inundation, creek development, sediment accretion, salinity, soil texture, and soil aeration.

During the 17 years that Mitchell Marsh was diked (1961-1978), the original marsh surface subsided 30-35 cm due to loss of buoyancy, oxidation and trampling, while the flanking control marsh surface did not subside (Figure 4). A similar pattern of subsidence occurred on the Y-Dike side that was diked for 27 years prior to dike removal; however the Y-Marsh site was generally about 20 cm lower than the Mitchell Marsh (Figure 4). Upon reestablishment of tidal exchange, the subsided pasture surface increased in elevation at a greater rate than the control marsh surface.

With dike removal, tides again began to flush inactive, clogged tidal creeks. Creeks deepened and narrowed due to erosion, but also because of sedimentation along marsh levees (Figure 5). Marsh creeks show greatest deepening low in the marsh where tidal flushing was greatest, and least deepening at creek heads. In the Y-Marsh, the original tidal creek pattern had been severely altered during the diking period. In places creeks were obliterated leading to ponding and anaerobiosis. Elsewhere the basic creek pattern remained and creeks deepened.

Creek reestablishment was critical to marsh restoration. It is by creeks that sediments and nutrients are exchanged with the estuary. It is by creeks that saline water enters and drives compositional change, and, it is by creeks that anaerobic toxics are removed and aeration enhanced.

Discussion

Salt marsh vegetation was fully restored to Mitchell Marsh and the Y-Marsh within about eight years without planting, seeding, or grading. The only necessary manipulation was dike removal which reestablished tidal exchange.

Composition of restored marsh vegetation, however, is very different now from that prevailing prior to diking and pasture creation in the early 1960s. Based on relatively unaltered natural salt marshes flanking the restored sites, I assumed that pre-dike composition was typical of high marsh with dominance of *Juncus balticus, Deschampsia cespitosa,* and *Potentilla pacifica.* Because the pasture surface subsided 30-40 cm during the diked period, upon return of estuarine circulation, low marsh characterizes the restored sites and strongly reflects salinity and soil texture conditions. Today, an expanse of almost pure *Carex lyngbyei* marks the more brackish and finer textured substrates. Where conditions are more marine, with higher salinities and sandier substrates, *Salicornia virginica* and *Distichlis spicata* dominate.

Above ground net primary productivity in the restored Mitchell Marsh is almost twice that of the pasture and typical high marsh. Enhanced productivity probably reflects released resources and increased tidally born nutrients. Since reestablishment of salt marsh is akin to primary succession, it is not surprising to observe high net productivity (Odum 1969).

To restore high marsh, surface elevations would need to be increased at least 30 cm. Accretion rates are greater in the restored sites (ca. 5 cm per decade) than in adjacent controls (ca. 1-2 cm per decade). Accretion rates, however, are more diminished in high marsh than in lower areas near the river; therefore, I estimate that it would take at least 80 to 100 years to change from a low to high salt marsh.

Drainage and soil aeration also control salt marsh restoration. When marshes are diked, natural drainage patterns disintegrate and are altered by reduced flow, sedimentation, livestock trampling of creek banks, and grading. For high marsh reestablishment, an efficient integrated creek system must redevelop. In Mitchell Marsh, where the creek system was subdued but not destroyed, restoration proceeded fairly rapidly. In the Y-Marsh, surface grading had destroyed the original creek system and drainage was poor. Marsh vegetation restoration was retarded.

Poor drainage and low surface elevations in the Y-Marsh led to persistent ponding which created extreme anaerobic conditions. These were accentuated by the accumulation of dead pasture litter that was not removed by tidal water. After dike breaching, extensive patches of algal mat dominated the Y-Marsh for about four to five years. Vascular plants tolerating these conditions included ephemeral *Cotula coronopifolia, Puccinelia pumila, Spergularia marina,* and *Triglochin coccinea.* More typical salt marsh species were found along creek margins or where soil aeration was greater. Gradually, soil aeration increased, algal mats broke up, and ephemeral

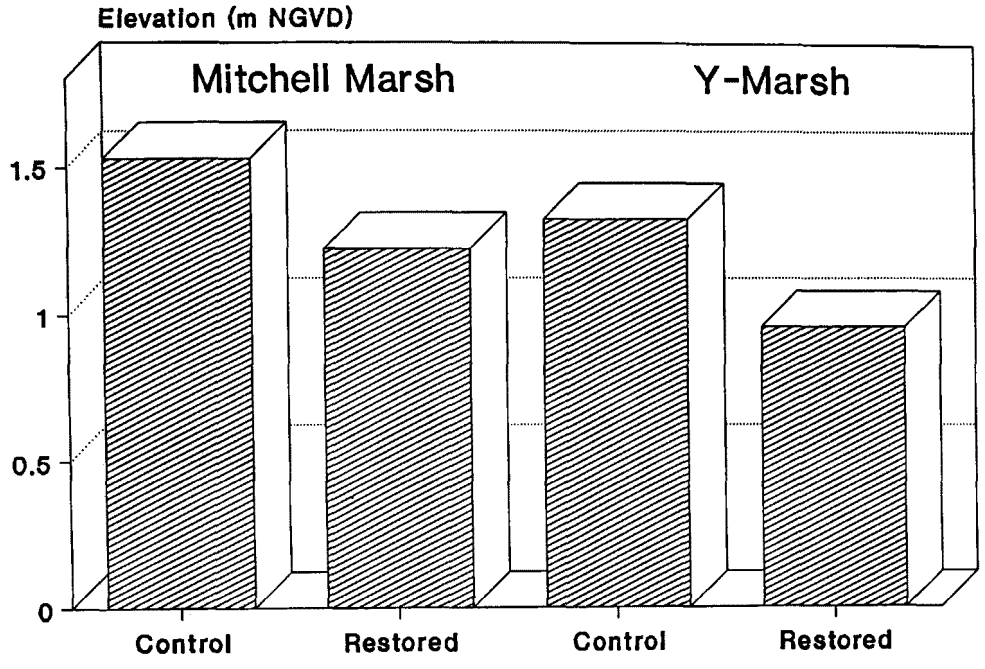

Figure 4. Elevation distribution 1988 of the restoration site and controls in Mitchell Marsh and Y-Marsh in the Salmon River estuary.

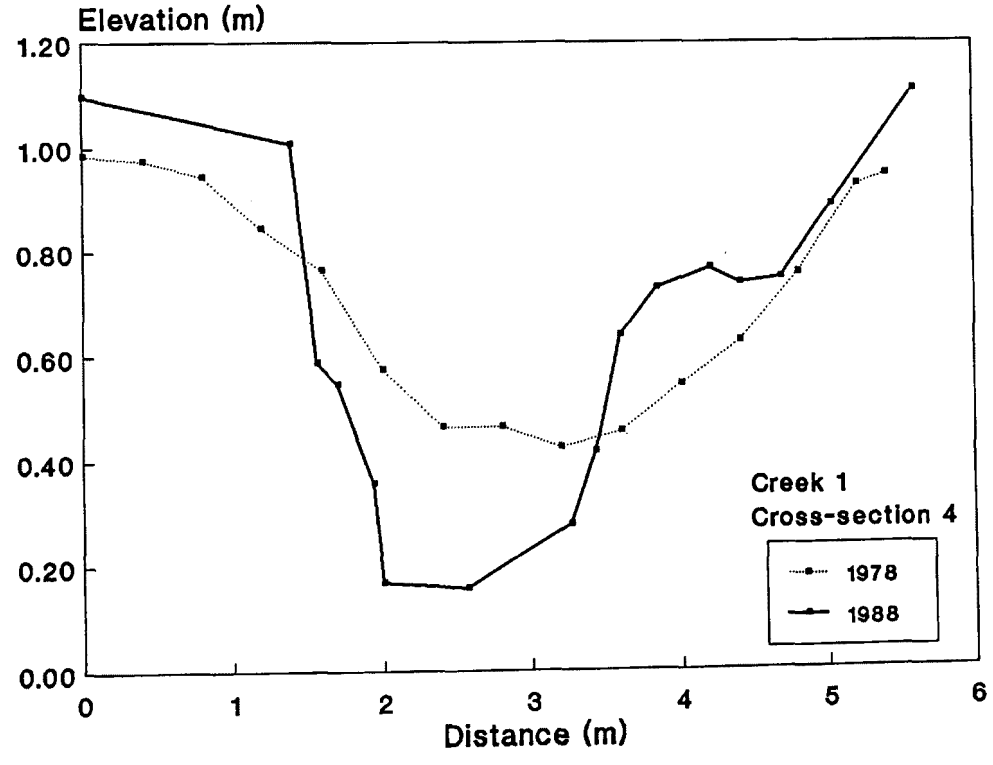

Figure 5. Elevation across a typical marsh creek cross-section in the Mitchell Marsh restoration site in 1978 and 1988 showing creek deepening associated with reestablished tidal circulation.

species have begun to be replaced by common regional salt marsh species.

Conclusions

I addressed four objectives. First, after dike removal, typical salt marsh vegetation reestablished in the course of about eight years in the absence of planting and seeding. Although I did not measure marsh function directly, species composition, enhanced productivity, presence of salmonids in marsh creeks (C. Simenstad pers. comm., June 30, 1993), and abundant waterfowl provide evidence of a functioning marsh.

Second, because the diked pasture subsided 30-40 cm relative to the original high marsh, low salt marsh vegetation prevailed after dike breaching. It is unlikely that original salt marsh conditions will be restored within 100 years, if then.

Third, rates of restoration as measured by vegetation composition vary greatly. Under favorable conditions, Mitchell Marsh returned to low marsh within about six years after which composition changed very slowly. In the Y-Marsh, where surface drainage had been severely altered causing ponding and anaerobiosis, marsh vegetation took about eight years to reestablish and even then was changing annually.

Fourth, the pattern of restoration was quite different in Mitchell Marsh than in the Y-Marsh. Although surface subsidence was the same in the two restored areas, initial elevation, drainage and anaerobiosis were strikingly different. Under anaerobic conditions prevailing in the Y-Marsh, restoration of vascular plants was retarded, algal mats covered large areas, annual ephemeral plants dominated for several years before more characteristic marsh plants became established.

Despite these differences in restoration pattern and rate, the Salmon River salt marsh restoration is a success. The project demonstrates that one of the best opportunities to stem the national and regional loss of coastal wetlands is to return agriculturally diked coastal wetlands to natural salt marsh through dike removal. This opportunity is particularly advantageous on large tracts of public land. It is under these circumstances that restoration can be carefully planned and monitored over decades.

Acknowledgements

The Salmon River estuary research project could not have taken place without the careful meticulous field and analytical work of Diane Mitchell and Janet Morlan, for this I am especially grateful. I also wish to acknowledge funding assistance from the U.S. Forest Service and U.S. Environmental Protection Agency, Region 10.

Literature Cited

Boulé, M. E. and K. F. Bierly. 1987. History of estuarine wetland development and alteration: What have we wrought? Northwest Environmental Journal 3:41-61.

Frenkel, R. E. and J. C. Morlan. 1991. Can we restore our salt marshes? Lessons from the Salmon River, Oregon. Northwest Environmental Journal 7:119-135.

Kibby, H. V., J. L. Gallagher, and W. D. Sanville. 1980. Field guide to evaluate net primary production of wetlands. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR, USA. (EPA-600/8-80-037).

Kusler, J. A. and M. E. Kentula (eds.). 1990. Wetland Creation and Restoration: The Status of the Science. Island Press, Washington, DC.

Mitchell, D. L. 1981. Salt marsh reestablishment following dike breaching in the Salmon River estuary, Oregon. Ph.D. Dissertation, Oregon State University, Corvallis, OR.

Mitsch, W. J. and J. G. Gosselink. 1993. Wetlands, 2nd ed. Van Nostrand Reinhold, New York, NY.

Morlan, J. C. 1991. Ecological status and dynamics of a salt marsh restoration in the Salmon River estuary, Oregon. M.S. Thesis, Oregon State University, Corvallis, OR

NRC (National Research Council Committee on Restoration of Aquatic Ecosystems). 1992. Restoration of Aquatic Ecosystems: Science, Technology, and Public Policy. National Academy Press, Washington, DC.

National Academy of Sciences. 1995. Wetlands Characteristics and Boundaries. National Academy Press, Washington, DC.

Odum, E. P. 1969. The strategy of ecosystem development. Science 164:262-270.

U.S. Forest Service. 1976. Final environmental statement for the management plan: Cascade Head Scenic Research Area. U.S. Department of Agriculture, Forest Service document USDA-FS-FES(Adm)-76-06 on file at the Siuslaw National Forest, Corvallis, OR.

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Batiquitos Lagoon Enhancement Project: Concept to Construction

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Ralph Appy[2]
Adam Whelchel[1]

Abstract: **The 593 acre Batiquitos Lagoon Enhancement Project is a joint restoration project of the City of Carlsbad and the Port of Los Angeles. The $55 million project is being funded by the Port of Los Angeles as mitigation for the loss of deep water marine habitat in the outer Los Angeles harbor area. The enhancement project involves the re-establishment of a tidal entrance to the blocked lagoon mouth and dredging approximately 3.1 million cubic yards of accumulated sediments from the lagoon basin to increase the tidal prism and restore coastal lagoon habitats. The project has been under construction since 1994 using electric dredges. A portion of the dredged material is being used for beach replenishment and a portion for the construction of nesting areas for the California least tern and western snowy plover. Use of the nesting sites by these birds has exceeded historic levels and contributes significantly to the coastal populations of these federally listed species. The project will be completed in 1997.**

Introduction

The Batiquitos Lagoon Enhancement Project (Figure 1) is the largest coastal wetland restoration project in California and provides an example of the effort and planning required to implement large scale projects. It is one of a series of six lagoons on the northern coast of San Diego County, several of which are also proposed for restoration. Most of the other proposed restoration projects involve restoration of tidal action as was done at Batiquitos.

The need for restoration at Batiquitos Lagoon was clearly recognized in 1976 when the Department of Fish and Game issued its report on the current conditions in the lagoon (Mudie *et al.* 1976). The lagoon had been tidally influenced during historic times; however, the construction of road and railroad bridges and causeways restricted tidal flows. With increasing agricultural and urbanization pressures on the lagoon's watershed, sediments filled the lagoon, decreasing its tidal prism. In addition, portions of the lagoon were filled for development, diked for solar salt production, and freshwater flows damned and diverted. These influences resulted in a lagoon environment that was frequently closed and sometimes completely fresh. In summer months, the closed system dried out and hypersaline mudflats were prevalent. In less than 150 years, the lagoon had been converted from a tidally influenced, marine community to one that altered annually between freshwater and hypersaline conditions.

The lagoon continued to support wildlife, particularly migratory birds (California Coastal Conservancy 1987). In addition, the federally listed, endangered California least tern *(Sterna antillarum browni)* and threatened western snowy plover *(Charadrius alexandrirrus nivosus)* were observed and sometimes nested on the mudflats of the lagoon. The concern expressed by most observers was that despite the existing values, the frequent long-term closure of the lagoon coupled with the decreasing tidal prism due to siltation would eventually eliminate the existing wetland functions and values.

In 1987, the Port of Los Angeles, the City of Carlsbad, the California State Lands Commission, the California Department of Fish and Game, the US Fish and Wildlife Service, and the National Marine Fisheries Service entered into a formal Memorandum of Agreement to enhance Batiquitos Lagoon for the mitigation of Port fills in San Pedro Bay. The state and federal resource agencies determined that the enhancement of Batiquitos Lagoon would be an appropriate mitigation to offset the adverse environmental impacts that would result from filling activities in the

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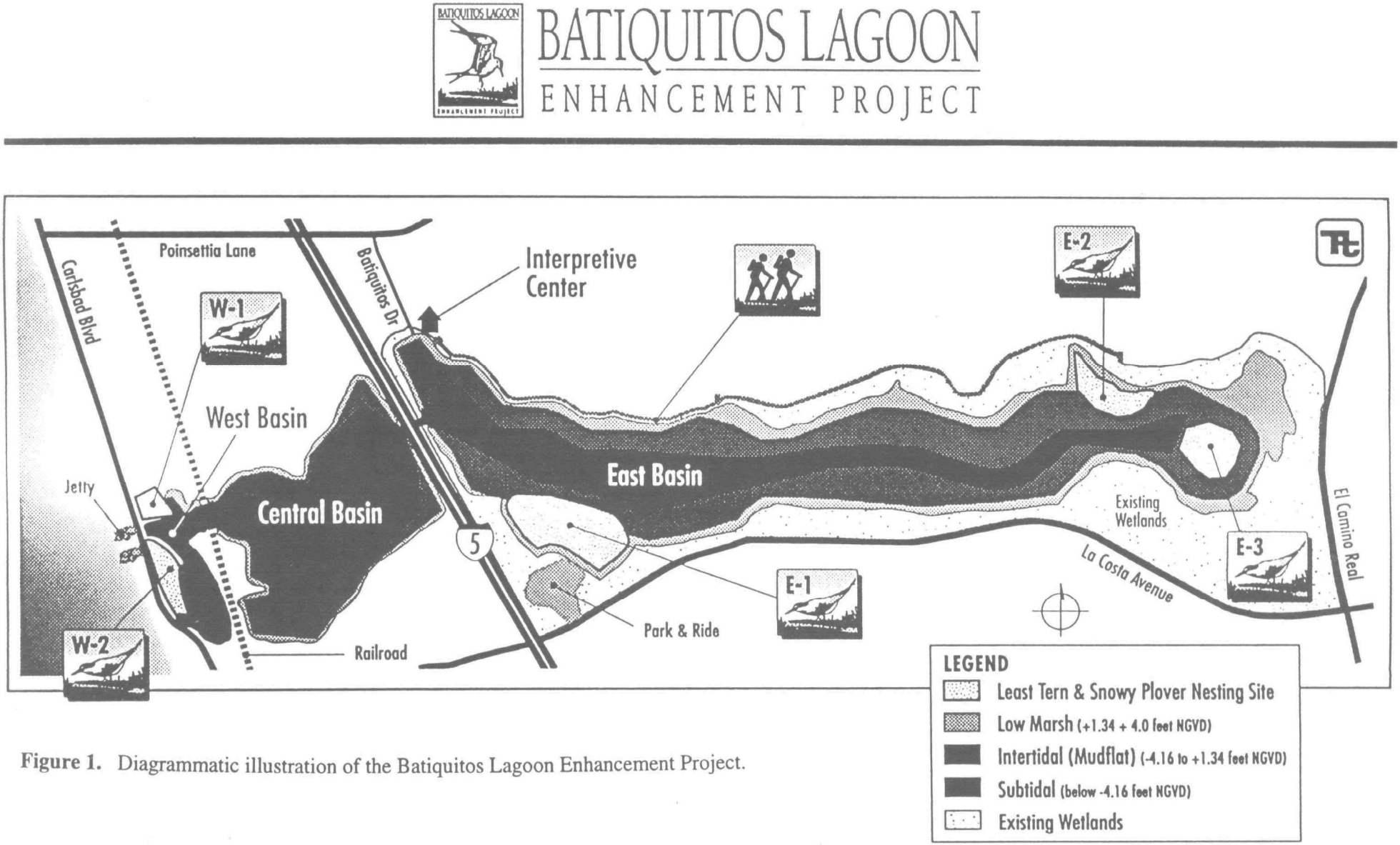

Figure 1. Diagrammatic illustration of the Batiquitos Lagoon Enhancement Project.

shallow waters in Los Angeles Harbor. The mitigation was both off-site and out-of-kind, but was deemed valuable because "shallow, estuarine coastal embayment habitat in southern California, with its relatively high value to marine fishes and migratory birds, has been reduced in area at a greater rate than that of deep water habitat".

A joint EIR/EIS document was prepared for the enhancement of the lagoon (City of Carlsbad 1990). The EIR/EIS evaluated a number of alternatives for the enhancement project. These alternatives ranged from no action to large scale dredging of the lagoon. The City and the California Coastal Commission eventually determined that a project referred to as "Mitigated Alternative B" had the best opportunity to restore tidal action to the lagoon and to provide for improved habitat for marine and estuarine fish and wildlife.

Engineering studies and specifications were prepared by the City and the Port and construction on the project commenced in April of 1994. The purpose of this paper is to summarize the construction procedure and to document the use of newly constructed nesting areas by least terns and western snowy plovers.

Construction Summary

The restoration of Batiquitos Lagoon consists of several major construction activities (Figure 1).

- Dredging the lagoon's central basin and use of the sandy material for beach nourishment and construction of shorebird nesting sites.
- Dredging the lagoon's east basin and the use of the silty material as fill within the central basin.
- Dredging of the west basin and use of the sandy material for nesting sites and beach nourishment.
- Construction of five California least tern and western snowy plover nesting sites totaling 38 acres.
- Construction of two 300-foot rock jetties extending into the ocean from the mouth of the lagoon.
- Replacement and/or rock revetment of bridges crossing the lagoon
- Revegetation of salt marsh vegetation disturbed during construction of the project.

The enhancement project has a unique approach to disposal of bottom materials that are unsuitable for beach nourishment. Within the west and central basin, sandy materials are prevalent. However, in the east basin, the material is largely fine silt and unsuitable for placement on area beaches. As a result, the construction involves an over-excavation of sand from the central basin and the placement of the fine silts into the pit created. Approximately 1.6 million cubic yards of sand will be placed on area beaches at the conclusion of the project. The sand was also used to create the nesting areas for shorebirds. These areas range in size from 1 acre to 19 acres and were created to be above tidal and flood water influence. Once the fine materials from the east basin are placed within the central basin, it will be capped with sandy material from the dredging in the western basin. At the end of the project, approximately 3 million cubic yards of material will have been moved. The change in habitat acreage as a result of the project is provided in Table 1.

The project is also being constructed in a manner to avoid impacts to existing natural resources. Dredging is being conducted using electric dredges to assure quiet and reduce air pollution. All dredging is halted during the least tern/western snowy plover nesting period (April 1 to September 15). Construction activities must also avoid the state listed Belding's savannah sparrow *(Passerculus sandwichensis beldingi)* habitat during the summer nesting period. All salt marsh vegetation disturbed by the project must be transplanted or replaced on a one to one ratio. These measures and others are being supervised by an onsite environmental monitor throughout the construction.

Utilization of Nesting Sites During Construction

One of the conditions established in the permits issued for the project was that suitable nesting sites for California least tern and western snowy plover be constructed during the project and the utilization of these sites monitored (Figure 2). During the first year of construction, one 2 acre site (W-1) was constructed, managed, and monitored. During the second year of construction, two additional sites (W-2–5.2 acres and E-l–19.1 acres) were made available. Each of the sites was constructed with sand either trucked in or dredged from the lagoon. The sand contained shell materials. Two of the sites (W-1 and W-2) were surrounded on landward facing

Table 1. Approximate acreage of habitats to the enhanced by the project.

Habitat type	Existing	After construction
Open nontidal water/seasonal mudflats	415	0
Subtidal (-1.6 ft MLLW)	0	148
Intertidal flats (-1.6 to 3.9 ft MLLW)	0	144
Salt marsh (3.9 to 7.0 ft MLLW)	111	174
Nontidal nesting flats (+7 ft MLLW)	0	38
Other transitional habitats	67	94
Total acreage	593	593

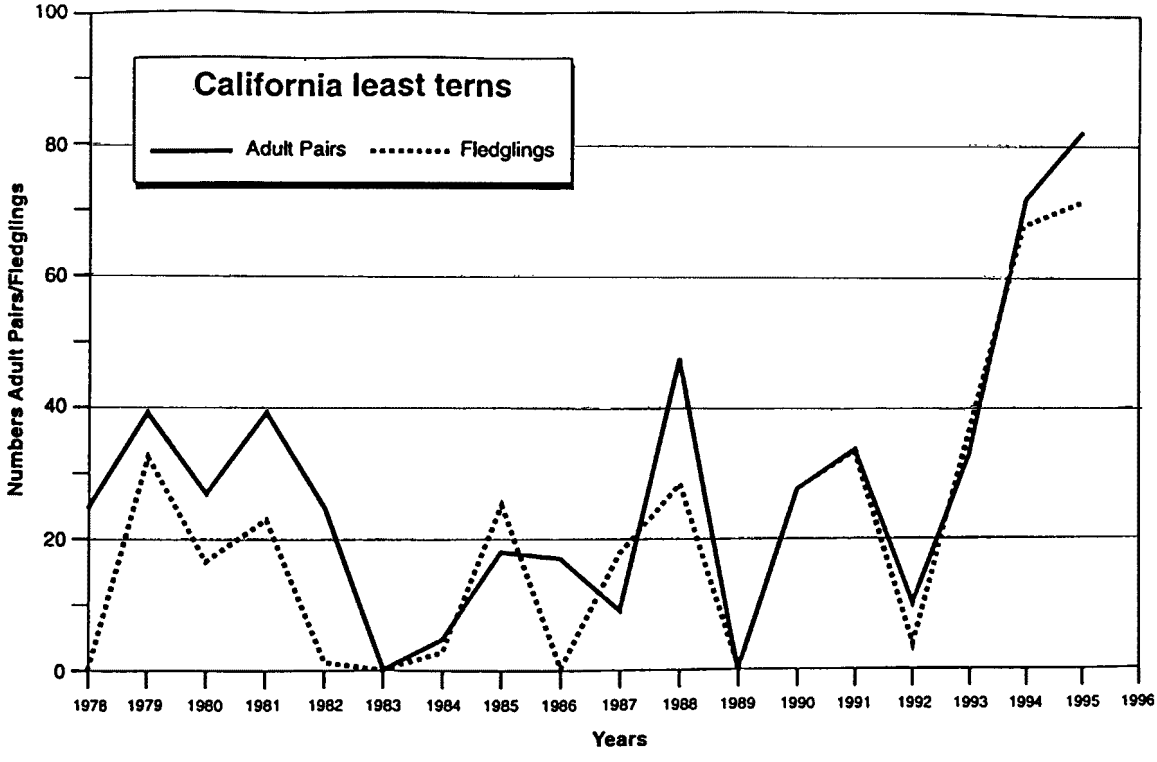

Figure 2. Utilitization of Batiquitos Lagoon Breeding Colony given in numbers of adult pairs and fledglings from 1978 through 1995.

portions with chain-link fencing up to 7 ft in height. On the waterfacing sides, they were bounded with 3 foot high polyethylene mesh fencing to retain least tern chicks inside the nesting area and to provide some predator control. Site E-1 was not fenced except at the access road.

To facilitate monitoring, a grid system of various dimensions was established on each of the three nesting sites. On W-1 and W-2, a 20x20 meter grid system was utilized and on E-1 a 60x60 meter grid system was used. The grid system assisted in locating and monitoring nest sites and chicks and was supplemented with marking each nest location with wooden slats (tongue depressors). Monitoring of the nesting areas was conducted daily either by observations outside the nesting area or by walk-throughs. Counts were kept on number of eggs laid, hatched, abandoned or depredated, and chicks fledged.

During the 1994 breeding season, approximately 72 adult pairs of the California least tern were observed nesting at the Batiquitos Lagoon breeding colony (Whelchel and Keane 1994). Nesting at Batiquitos Lagoon in 1994 represented approximately 5 percent of the total nesting pairs in San Diego County and 3 percent of the State total. Nesting was initiated on May 13th and ended on June 15th. The 72 adult pairs initiated a total of 80 nests. A total of 147 eggs were laid; of these, 129 hatched (88 percent hatching success). Mean clutch size was 1.84. Based on fledgling counts, an estimated 68 chicks fledged from Batiquitos Lagoon. Fledgling dates ranged from June 22 to July 25.

During the 1995 nesting season, 82 adult pairs of the California least tern were observed at Batiquitos Lagoon (Whelchel and Keane 1995). These birds initiated a total of 93 nests. Nest initiation ranged from May 22 to June 20. The distribution of nest initiations between the three newly created nesting sites was 60 (W-I), 30 (E-I), and 3 (W-2). A total of 165 eggs were laid; of these 146 hatched (88 percent hatching success). The mean clutch size was 1.77. Based on fledgling counts and recaptures, an estimated 71 chicks fledged (0.87 fledgling per pair).

In addition to least tern, nesting activity for the western snowy plover was also recorded. This species did not utilize the constructed nest site in 1994, but did utilize E-1 during the 1995 nesting season (Whelchel and Keane 1995). Fifteen nesting attempts were documented between April 3 and July 25. A total of 40 eggs were laid and of these 32 hatched (80 percent hatching success rate). The modal clutch size was 3 eggs with a mean clutch size of 2.7. Based on fledgling surveys, an estimated 24 chicks fledged between June 28 and September 19.

Predation at the nesting sites was reduced through the efforts of an active trapping and monitoring program conducted by Animal Damage Control. Most predation attacks on least terns were from American kestrels, great horned owls, and a feral cat. Most predation attacks on western snowy plovers were due to common ravens.

Discussion

The construction of the Batiquitos Lagoon Enhancement Project represents the culmination of a 20-year planning effort. Restoration of degraded systems is made difficult by the conflicts that arise due to surrounding land uses, recreational demands on beaches where inlets are proposed, infrastructure and transportation corridor constraints, protection of federally and state listed species, and implementation costs. The cooperation of federal and state agencies coupled with the economic resources of the Port of Los Angeles and the desire of the City of Carlsbad to restore its natural resources was the combination that made this project possible. The project benefits, especially to beach nourishment and habitat enhancement, provided the public incentives to accept the project.

The use of the nesting areas by federally listed shorebirds has far exceeded expectations. This may be due in part to the protection afforded these areas by both passive and active predator control methods. However, the wide-spread use of the site during construction suggests that the design and implementation of the project has been successful. Only long-term monitoring to be conducted after the completion of the construction will tell whether the restoration itself functions as designed.

Literature Cited

California State Coastal Conservancy. 1987. Batiquitos Lagoon Enhancement Plan. Revised Draft. Oakland, CA.

City of Carlsbad. 1990. Batiquitos Lagoon Enhancement Project. Final EIR/EIS.

Memorandum of Agreement. 1987. Agreement among the City of Los Angeles, the City of Carlsbad, the California Department of Fish and Game,

the California State Lands Commission, the National Marine Fisheries Service, and the United States Fish and Wildlife Service to Establish a Project for Compensation of Marine Habitat Losses Incurred by Port Development Landfills within the Harbor District of the City of Los Angeles by Marine Habitat Enhancement at Batiquitos Lagoon.

Mudie, P. J., B. M. Browning and J.W. Speth. 1976. The Natural Resources of San Dieguito and Batiquitos Lagoon. Coastal Wetland Series #12. Department of Fish and Game, State of California.

Whelchel, A. W. and K. Keane. 1994. California Least Tern Breeding Survey, Batiquitos Lagoon, San Diego County, 1994 Season. Batiquitos Lagoon Environmental Monitor, City of Carlsbad, CA.

Whelchel, A. W. and K. Keane. 1995. Western Snowy Plover Breeding Survey, Batiquitos Lagoon, San Diego County, 1995 Season. Batiquitos Lagoon Environmental Monitor, City of Carlsbad, CA.

Whelchel, A. W. and K. Keane. 1995. California Least Tern Breeding Survey, Batiquitos Lagoon. San Diego County, 1995 Season. Batiquitos Lagoon Environmental Monitor, City of Carlsbad, CA.

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Restoration of *Spartina* Marsh Function: An Infaunal Perspective

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Although rarely a focus of wetlands restoration monitoring, sediment-dwelling fauna (infauna) play a number of important roles in salt marsh function. They cycle organic matter by breaking down detritus and mixing sediment; provide food for fish, birds, and other predators; link primary producers, microbes, and fungi with higher order consumers; and represent a significant portion of the faunal diversity in salt marshes. Here we will summarize some key results from studies of macrofauna (animals retained on a 300 micron sieve) in restored salt marsh systems on both the Pacific and Atlantic coasts of the United States. We consider infaunal succession, measures and rates of recovery relative to natural marsh reference sites, factors that influence recovery, and possible causes of compositional differences between created and natural marshes.

North Carolina Sites

Data presented here are from studies of the Port Marsh, a 2.2-acre created *Spartina alterniflora* marsh located on the Newport River in Morehead City, N.C. An adjacent natural marsh was used as a reference site. Experiments were initiated in June 1990 to (1) test the effects of 10 different soil treatments on ecosystem recovery, and (2) evaluate successional stages, mechanisms and recovery potential of flora and fauna in created *Spartina alterniflora* habitats relative to an adjacent natural marsh. The treatments applied included a series of organic matter amendments (straw, alfalfa, peat) with and without inorganic N additions, as well as planted and unplanted, rototilled and unrototilled controls, each replicated three times (one 2 x 7 m plot in each of three blocks). Samples were taken from upper and lower elevations of each plot in the created marsh and the natural reference marsh at 1 wk, 1 mo, 3 mo, 6 mo, 12 mo, 15 mo, 18 mo, 22 mo, 27 mo, 47 mo, and 52 mo after initiation.

Unamended Treatments

Recovery of the unamended sediments (planted and unplanted but not rototilled) during the first 4 years after marsh establishment is described in Levin et al. (in press). Throughout much of the study macrofaunal densities were reduced in the planted (unamended) and unplanted (bare) plots relative to the natural marsh, especially at upper tidal elevations (Figure 1). A similar pattern was observed for species richness. Three taxa, the polychaetes *Streblospio benedicti* and *Capitella* sp., and turbellarians, accounted for over 75 percent of the created marsh macrofauna during the first several years after marsh establishment. Even when macrofaunal densities and species richness were comparable in the created and natural systems, species composition differed greatly. Just over 2 years after marsh creation, the artificial marsh had 92 percent of the natural marsh species and 88 percent of the individuals present, yet the composition of the two systems was markedly different. The natural marsh was dominated by oligochaetes (61 percent), which are burrowing, subsurface-deposit feeders, while in the created marsh the oligochaetes were virtually absent, having been replaced, proportionally, by *Streblospio benedicti*, a tube-dwelling, surface-deposit feeder. *S. benedicti* and other surface-deposit feeders may be more available to fish and birds as food, but are less likely to aerate

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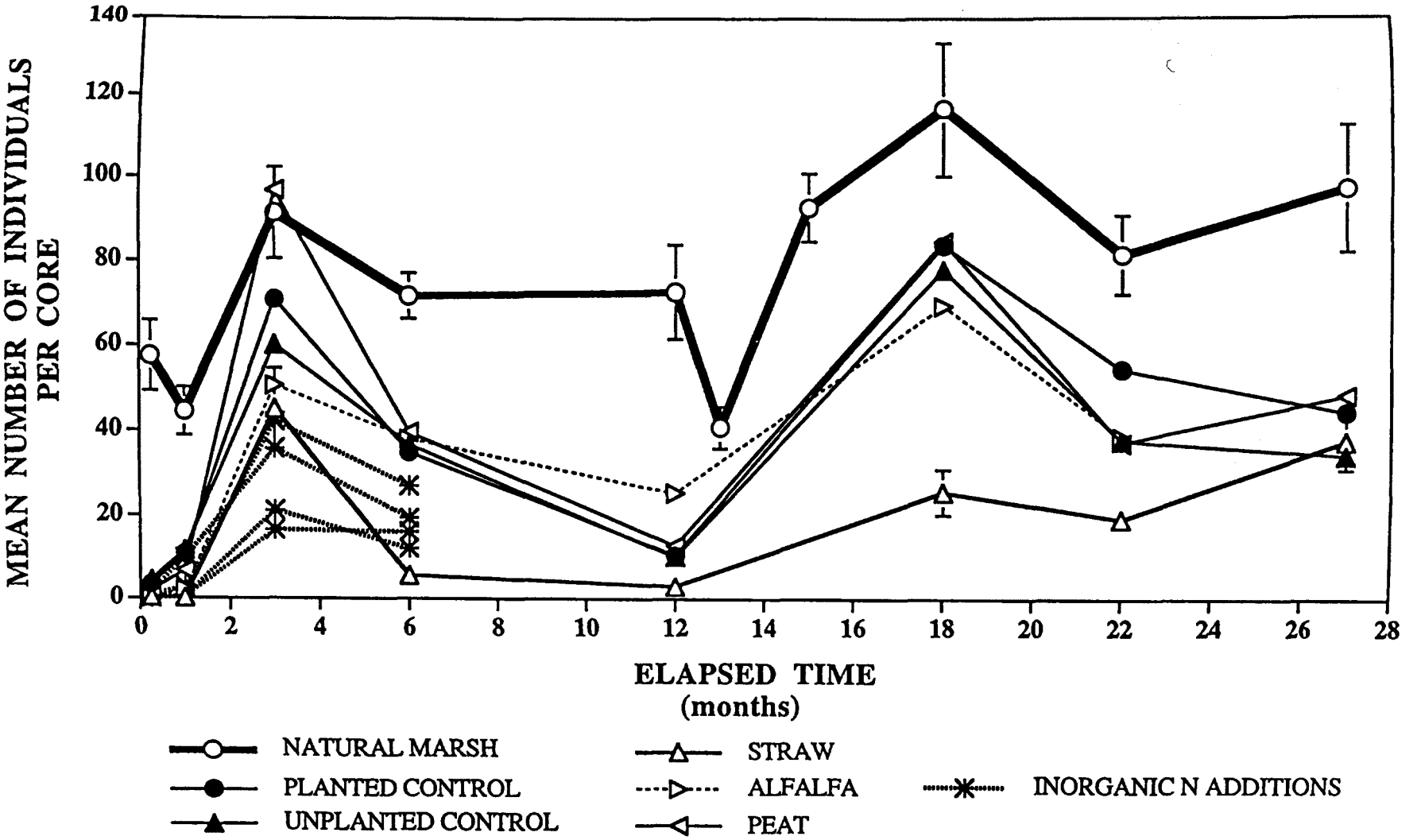

Figure 1. Mean number of macrofauna (>300 microns) collected per 18 cm^2 core in the Port Marsh and an adjacent natural marsh. Data are given for selected organic amendment treatments (straw, alfalfa, and peat incorporated into the soil at 3000 g/m^2 with no additional nitrogen), planted and unplanted controls, and an adjacent natural marsh. Data are also presented for inorganic nitrogen additions including unamended and straw-, alfalfa-, and peat-amended plots. Time indicates months since establishment of the man-made marsh. Error bars are one standard error, and are included only on natural marsh and straw treatments, for clarity of presentation.

sediments or contribute to mechanical breakdown of plant detritus. A primary lesson from these results is that total abundance data and species richness measures alone cannot reflect infaunal functional recovery without the additional consideration of species composition and trophic roles.

Organic Amendments

We initially expected that addition of organic matter and nutrients to marsh soils should expedite marsh recovery by enhancing growth of *Spartina*. Addition of straw, alfalfa, and peat initially reduced the rates of macrofaunal colonization (Figure 1) and species richness relative to the unamended control plots. However, after 6 months all organic additions except straw had infaunal densities statistically indistinguishable from control treatments. Negative effects of straw on macrofaunal abundance and species richness persisted for at least 22 months (Figure 1). The straw treatment exhibited the lowest redox levels of any treatment during the experiment, and the lack of oxygen may have contributed to poor infaunal establishment. Additions of inorganic N, tracked during the first 6 months only, appeared to further reduce macrofaunal colonization in plots with organic amendments (Figure 1). The source of this effect is unknown.

Elevation

The lower tidal heights came to resemble the natural marsh more quickly than the higher elevations did in terms of both abundance and species richness. The greater inundation time at low elevations may enhance recruitment.

Vegetation

Specific associations between fauna and above or belowground *Spartina* biomass have been reported for a variety of infaunal taxa. We therefore note with interest that the planted and unplanted control plots exhibited similar patterns of macrofaunal colonization in most instances (Figure 1), although we did not sample *Spartina* culms directly. Also, proximity to *Spartina* culms (measured to the nearest 10 cm) did not appear to affect densities of total macrofauna or dominant species in the created marsh. Because the unplanted plots and adjacent walkways were surrounded by planted *Spartina* on all sides but one (the creek edge), we cannot infer that the observed successional patterns would apply to unvegetated areas larger than 6 x 7 m.

Life Histories

The role of life history in marsh succession also was examined by categorizing each species into one of four development or dispersal modes, and then evaluating the composition of the system over time with respect to these categories. Planktotrophic larval development was rare among macrofauna in the natural (reference) marsh, while the created system was dominated by planktotrophic forms or direct developers with swimming adult stages, especially during the first several years. Conversely, in the natural system a substantially larger proportion of its fauna exhibited direct development with no dispersal stage, relative to the created marsh, although representation of direct developers increased with time in the man-made system. Thus, dispersal potential may partially explain the early differences in faunal composition between the created and natural marshes.

Southern California Sites

Preliminary studies have examined macrofauna of three restored wetland areas: a newly created system in the Northern Wildlife Preserve of Mission Bay (unvegetated during the first 3 months); a 5-year-old restored *Salicornia virginica* marsh within Anaheim Bay; and in 10-year-old restored *Spartina foliosa* and *S. virginica* habitats within the Connector Marsh of San Diego Bay. In the 5- and 10-year-old systems, total macrofaunal densities and species richness were either similar or were higher in the restored than natural transects (Table 1). In all three of these systems oligochaetes were the most abundant taxon. The oligochaete *Paranais litoralis* was a dominant colonizer of the restored marshes and in some cases was significantly more abundant or proportionally better represented in the restored than reference marshes. At high densities, adults of this species swim in the water column, providing an effective dispersal mechanism. In sharp contrast, poorly dispersing tubificid and enchytraeid oligochaetes dominated natural sediments in most of the Pacific and Atlantic coast marshes studied. *S. benedicti* and *Capitella* spp., both dominant early colonizers in North Carolina, were present but relatively rare in both created and natural habitats studied in southern California. In southern California, the young Mission Bay restored marsh was initially colonized by peracarid

Table 1. Macrofauna (>300 microns) and oligochaete representation in restored (Rest.) and natural (Nat.) salt marshes of southern California. Restored systems in Paradise Creek (San Diego Bay), Anaheim Bay, and Mission Bay were 10 yr., 5 yr., and 3 mo. of age, respectively, at the time of sampling. Natural marsh data from Mission Bay March 1996 have not yet been processed. P values indicate t tests comparing restored and natural values within a system. Percent data were arcsin-transformed for statistical analyses, then back-transformed for presentation.

	Macrofaunal Density no. per 18.05 cm^2 core			***P. litoralis* Density no. per 18.05 cm^2 core**			**% Oligochaetes**			***% P. litoralis***		
Paradise Creek	Rest.	Nat.	P	Rest.	Nat.	P	Rest.	Nat.	P	Rest.	Nat.	P
San Diego Bay (Feb. 1994)												
Spartina foliosa transect	231.2	310.0	NS	119.0	99.3	NS	95.4	96.5	NS	60.8	28.2	0.002
Salicornia virginica transect	159.5	47.0	0.049	89.6	16.5	0.026	93.1	79.4	NS	46.4	42.1	NS
n = 10 cores												
Anaheim Bay (Feb. 1994)												
Salicornia virginica transect	211.1	59.9	NS	142.9	23.6	0.087	33.1	86.7	0.068	31.9	28.6	NS
n = 10 cores												
Mission Bay (Mr. 1996)												
Creekbed (unvegetated)	147	No Data		79.5	No Data		54.1	No Data		54.1	No Data	
n = 2 cores												

crustaceans (amphipods and tanaids–all direct developers) and annelids that rafted into the system on drift plant material.

Conclusions and Applications

1. Organic amendments such as peat or alfalfa can retard macrofaunal colonization initially, but should be used if they enhance *Spartina* growth, since no adverse long-term effects were observed. This may not be true of straw. Leaching of organic amendments prior to their application may amcliorate the early inhibition of infaunal recruitment.
2. Our results demonstrate that although faunal densities and species richness measures in a created marsh can resemble those of a natural system fairly quickly, these similarities can belie compositional differences that may be of consequence. In North Carolina, restored marshes exhibited a notable absence of oligochaetes, while in southern California, restored marshes exhibited an unusually high representation of a naidid oligochaete species with high adult dispersal ability. Although oligochaetes comprise over 3/4 of the macrofauna in most salt marshes, too little is known about their functional roles to assess effects of reduced or elevated oligochaete densities.
3. Results from both North Carolina and California suggest that species' dispersal abilities affect macrofaunal succession. However, planktotrophic recruits dominated newly created marshes of North Carolina, whereas adult dispersers appear to dominate those of California. In southern California, where wetlands habitats are isolated by stretches of high-energy coastline, adult dispersal may be more effective than that of long-lived planktonic larvae. On both coasts species lacking effective dispersal mechanisms were slowest to recover (Moy and Levin, 1991; Levin et al., in prep.). These taxa could be stimulated by 'seeding' restoration sites with adults. Construction of restored systems that directly border natural marshes also may facilitate recruitment by these species.
4. From the studies conducted, we do not know how long full recovery of macrofaunal communities will take in restored salt marshes. Recovery seems to be both marsh- and taxon-specific, with time alone being no guarantee of functional equivalence (Sacco et al., 1994). We suspect that the time frame will match that required for restored marsh sediment properties (organic matter, water content, and particle size) to approach natural levels.

Literature Cited

Levin, L. A., Talley, D. M., Thayer, G. (in press). Succession of macrobenthos in a created salt marsh. Marine Ecology Progress Series.

Moy, L. D., Levin, L. A. 1991. Are *Spartina* marshes a replaceable resource? A functional approach to evaluation of marsh creation efforts. Estuaries 14: 1-16.

Sacco, J., Seneca, E., Wentworth T. 1994. Infaunal community development of artificially established salt marshes in North Carolina. Estuaries 17: 489-500.

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The Sum Exceedance Value as a Measure of Wetland Vegetation Hydrologic Tolerance

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Abstract: **Although qualitative information exists on wetland community hydropatterns and wetland species hydrologic ranges, it has not been adequately quantified in the Midwest. The lack of quantitative baseline information hampers current efforts to restore, create, and manage wetlands in the Midwest. To quantify wetland species occurrence along a hydrologic gradient, we used a method called the sum exceedance value (SEV). The SEV is an integration of the wetland hydropattern that incorporates the magnitude, timing, and duration of surface flooding and soil saturation within the growing season. We installed and monitored staff gauges and groundwater monitoring well transects within wetland communities of several high quality Chicago-area natural areas (Grass Lake Marsh, Lake County; Wadsworth Prairie, Lake County; and West Chicago Prairie, DuPage County) to obtain wetland community hydropatterns. We obtained plant species locations, corresponding cover classes, and plot elevations during yearly July vegetation sampling and surveying. Hydropatterns and corresponding SEVs were determined for all vegetation plots based on bi-weekly water levels and vegetation plot elevations. A range of SEVs was computed for each species based on elevational abundance within each study area. Monitoring results indicate that wetland species distribution is strongly correlated with elevation and the sum exceedance value. From these results, we present mean SEVs for wet prairie, sedge meadow, shallow marsh and deep marsh communities. We also present preliminary preferred, tolerated, and avoided SEV ranges for all species analyzed for each wetland community type at these three sites. This research has many applications in wetland management, restoration, and creation. The methods and data can be used for selected community or species management, differentiating community and hydrologic disturbance from natural dynamics, and identifying target communities, target hydropatterns, design criteria, planting locations, and performance standards for wetland restoration/ creation projects.**

Introduction

Background

Hydrology is a critical component of all wetland ecosystems (Mitch and Gosselink 1993). Although many environmental factors (light, soils, temperature, etc.) influence vegetation distribution, one dimensional environmental gradients are often seen as an oversimplification of many environmental effects on species distribution (Curtis and McIntosh 1951). Hydrology, though, is an extremely strong and possibly overriding selection pressure influencing the distribution of vegetation at the wet end of the soil moisture continuum (Mitch and Gosselink 1993; Spence 1982). Because of the strong influence inundation exerts on the plant community, plant species are often stratified along a hydrologic gradient based upon species' adaptations to particular hydropatterns or

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inundation regimes (Johnson 1987; Shipley 1991; Zimmerman 1987). Temporal and spatial variation in hydrology is key to the spatial and seasonal variability in the vegetation distribution of our wetland communities (Gosselink and Turner 1978; Niering 1987a; Niering 1987b; Weller 1987).

Several terms are used to describe wetland hydrologic signatures. For clarification purposes in this paper we will use the term hydropattern to describe the yearly pattern of water levels within a wetland, similar to Mitch and Gosselink's (1993) definition of the hydroperiod. We will define the hydroperiod instead as the period of inundation within a particular wetland. The hydropattern is the "hydrologic signature of each wetland type" and strongly influences the type of wetland community, its diversity, productivity, and distribution of associated plants and animals (Mitch and Gosselink 1993). Typical marsh hydropatterns differ from typical deep water swamp hydropatterns in shape and flooding depth and duration. The hydropattern within one part of a wetland can differ from the hydropattern in another portion of the wetland based on topography, variation in hydrologic inputs and outputs, etc (Zimmerman 1987). A wetland's hydropattern can also vary temporally, based on yearly differences in precipitation patterns (Gosselink and Turner 1978; van der Valk 1989; Doss 1993).

The hydropattern variation between and within communities presents difficulties in managing, restoring, and creating wetland communities. For the wetland manager, differentiating between natural rainfall driven vegetation dynamics and vegetation change caused by an unnatural hydrologic disturbance is difficult without knowledge of the sites' hydropattern, precipitation dynamics, and wetland vegetation hydrologic preferences. For the wetland restorationist, identifying planting locations is difficult without knowledge of the target community's hydropatterns, seasonal water elevations, basin morphology, and wetland vegetation hydrologic preferences. Identifying the target community's hydropattern and wetland vegetation occurrence along the hydrologic gradient is critical for successful vegetation establishment in restoration and creation projects. Because the success of these projects is often dependent on the vegetation, successful wetland vegetation establishment is critical. Wetland vegetation is typically planted with minimal information on species' preferred hydropatterns and water depths. The wetland project manager has to make management, planting, construction, or design decisions without adequate information on the hydropatterns and vegetation hydrologic tolerance of natural wetland communities. Consequently, most of these projects are prone to failure (Kentula et al. 1992).

Purpose

Although experienced wetland managers and naturalists are familiar with species' qualitative hydropattern and seasonal water depth tolerances, this information has not been adequately quantified, hampering wetland protection, restoration, and management efforts (Maurizi and Poillon 1992; Galatowitch and van der Valk 1994). Species tolerance to selected features of the hydropattern have been included in the literature (Nelson and Anderson 1983; Gunderson et al. 1986; Picket and Bazzaz 1978; Robel 1962; Squires and van der Valk 1992; van der Valk et al. 1994; Wentz et al. 1974, and Whitlow 1979; Thunhorst 1993). Examples of these features include peak water levels, mean water levels, flooding duration, the rate of drawdown, low water levels, drainage class etc. The information provided on species' hydrologic tolerances includes anecdotal water level and flooding evidence, greenhouse studies with possible applications to natural systems, and detailed statistical gradient analysis not readily usable for wetland managers and restorationists. Typically, these studies describe vegetation response to depth, duration, or timing of inundation or saturation independently and not their combination. The information does not facilitate easy application or comparison between sites and studies. Additionally, reliable measurements of various wetland hydropattern features (water depth, duration of flooding, rate of drawdown, timing of drawdown, etc.) for high-quality communities have not been accurately quantified. In instances where some knowledge is available, these disparate measurements of hydropattern features do not adequately represent the hydropattern itself and do not allow comparisons of its influence on species distribution within or between sites and communities.

We have been conducting a baseline study of the hydrology, vegetation, and vegetation response to natural yearly hydropattern changes in several Illinois wetland communities. We are attempting to quantify the hydropattern and wetland vegetation distribution in response to the hydropattern. In our early attempts to quantify the hydropattern, we used several measurements including peak water level, low water level, depth of flooding, duration of flooding, timing of drawdown, and rate of drawdown. Unfortunately, we

felt that these measurements did not adequately characterize the hydrologic influence on the plant community given the variation in the hydrologic components (depth, duration, timing, etc.) from one year to the next. We needed an increased level of precision that included a comprehensive measure of the hydropattern. Following work conducted by Gowing and Spoor (In Press) and Gowing, Spoor, and Mountford (In Press) in the United Kingdom, we are analyzing an additional measure of the wetland hydropattern, the sum exceedance value (SEV), in an attempt to better characterize vegetation response to hydrologic change. The SEV is an integrated feature of the hydropattern that incorporates the magnitude, timing, and duration of surface flooding or soil saturation (Spoor 1993).

The Sum Exceedance Value

The SEV (Figure 1) is a simple, quantifiable parameter that integrates water table depth and duration of flooding/saturation over a growing season, similar in concept to the "degree-day" measure used to assess temperature stress:

$$SEV = \sum_{i=1}^{n} (d_{30})$$

'n' is the number of days present in the growing season. 'SEV_{30}' is the Sum Exceedance Value Index for an entire growing season. 'd_{30}' is a daily value of the elevation of the water above a datum located 30 cm below the soil surface and is defined to be zero when the water table is below the datum. We selected 30 cm as an assumed average wetland plant rooting depth (National Research Council 1995). The SEV is a valuable index for quantifying the hydropattern because the SEV describes how much and for how long water depths are present in a given soil column. For example, using a 30 cm datum (0.30 m), if the water were 5 cm (0.05 m) below the soil surface every day for a 200 day growing season, SEV would be 50 m.day, or (0.30 - 0.05 m) * 200 days. In comparing SEV values for two different wetlands (or two different locations in the same wetland) over growing seasons of identical length, high values indicate greater wetness. SEV values can be easily calculated at any ground surface elevation within the wetland from water table data using a spreadsheet if the elevation and appropriate wells and staff gauges are surveyed. From species occurrence and elevation data, an SEV range can be calculated for a particular species at a site.

Field studies with agricultural crops, and more recently with wetland plants, have documented strong relationships between the soil moisture indices like the SEV and plant productivity and abundance. Soil moisture indices have been used successfully in agronomy to assess water stress and associated changes in productivity in corn, soybeans, grain sorghum, and other crops (Shaw 1974; Sudar et. al. 1979; Howell et al. 1976; Hardjoamidjojo et al. 1990; Evans et al. 1990). Recent work by Spoor (1993), Gowing and Spoor (In Press), and Gowing, Spoor, and Mountford (In Press) demonstrated strong relationships between the SEV and wetland plant distribution in United Kingdom wet grasslands.

Although the SEV reduces a site's growing season hydropattern to one number, this number incorporates something many of the other measurements do not: the depth, duration and timing of inundation/saturation combined. For a particular species, researchers and managers can compare species SEV distributions between years, compare species SEV distributions between sites, predict vegetation responses to hydrologic change, and determine the effects of modeled hydropatterns on vegetation distribution for wetland mitigation projects (Konhya et al. 1995).

Methods

We conducted field work and data analysis to identify and quantify hydropatterns of Illinois' native wetland communities and plant species. Hydropatterns were quantified using the SEV from site elevation and water levels. Methods include site selection, hydrologic monitoring, surveying, vegetation sampling, SEV calculations, and identifying species' hydrologic preferences.

Site Selection

We randomly selected possible study sites from the state list of natural areas as identified in the Illinois Natural Area Inventory (White 1978). Such sites provide the best information on natural wetland communities in Illinois because the plant communities are relatively intact and human disturbance has been relatively light. Over fifty high quality wetland sites of various types were initially identified from the Illinois Natural Areas Inventory (White, 1978). This list

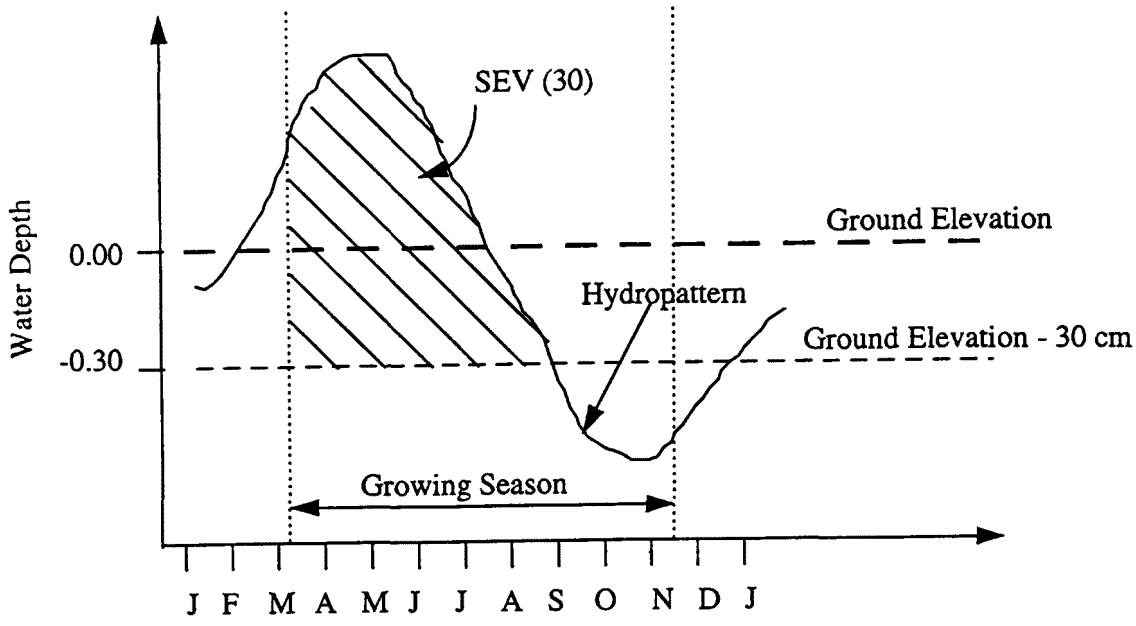

Figure 1. Representation of sum exceedance value derived from hydropattern, growing season, and 30 cm below ground elevation.

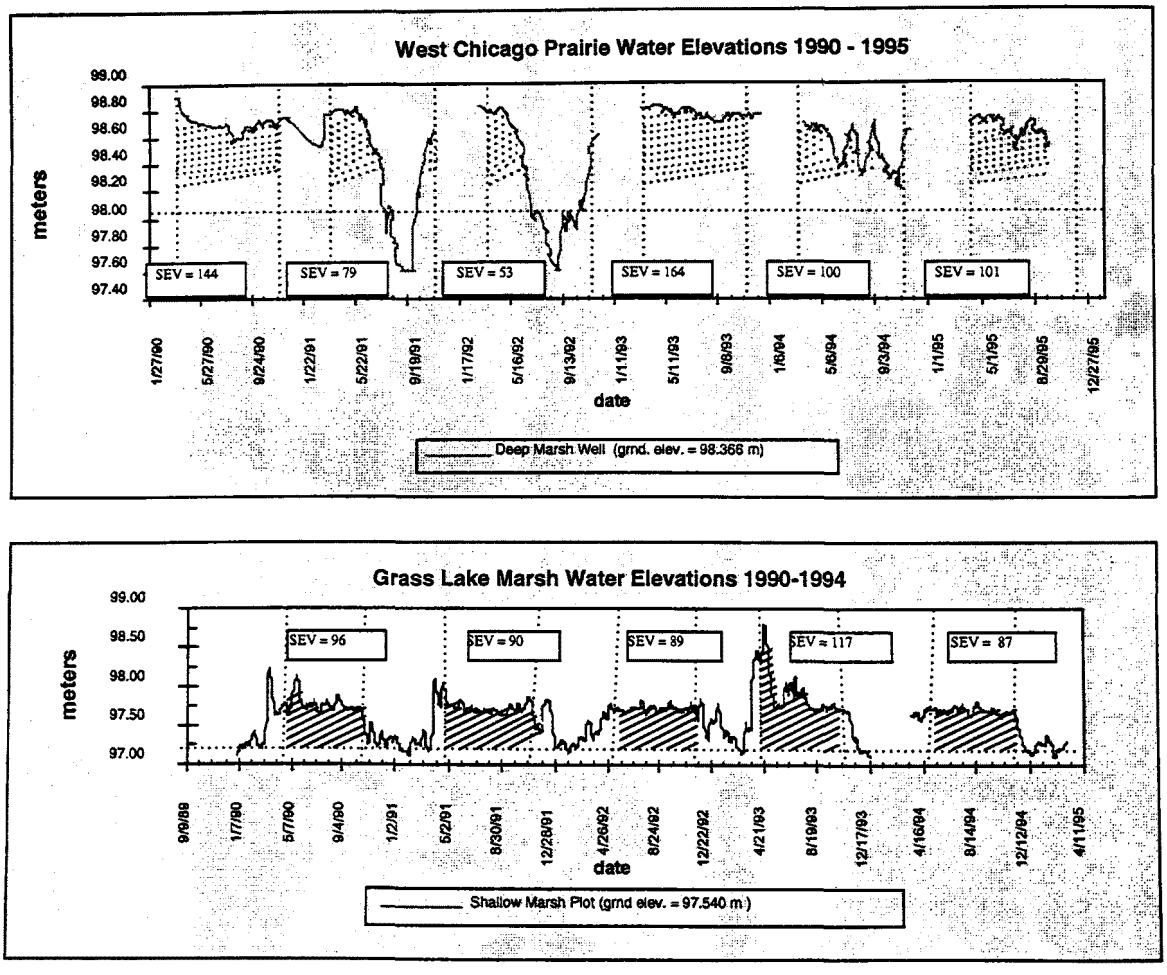

Figure 2. West Chicago Prairie and Grass Lake Marsh hydrographs from water level data.

was screened for sites with minimal hydrologic disturbance and, preferably, existing hydrologic data or hydrologic monitoring capability (e.g., existing wells or staff gauges). Ten sites on the short list were randomly selected and visited to determine feasibility for field sampling. Field surveys were made of each site to obtain baseline information on soils, plant community composition and diversity, general hydrology, and other important features. Additional information on plant communities, soils, ownership, disturbance history, and management activities was obtained from soil surveys, topographic maps, reports, the Illinois Natural Areas Inventory, and other sources. Of the sites included within the study, three will be discussed in this paper.

Study Sites

All three study sites described in this paper (Grass Lake Marsh Natural Area, Wadsworth Prairie Nature Preserve, and West Chicago Prairie Natural Area) are located within Illinois' Northeastern Morainal Division (Schwegmann 1973). Grass Lake Marsh Natural Area (Lake County) consists of shallow and deep water marsh communities as described by White (1978) and Ebinger et al. (1977). The shallow marsh community is dominated by *Sparganium eurycarpum* (bur-reed), *Scirpus fluviatilis* (river bulrush), and *Carex lacustris* (lake sedge). The deep marsh community is dominated by *Typha* spp. (cattail) and *Sparganium eurycarpum*. The site is adjacent to Grass Lake of the Chain O' Lakes, with water levels influenced by Grass Lake and the adjacent Fox River.

Wadsworth Prairie Nature Preserve (Lake County) is located near the Des Plaines River and consists of a mosaic of marsh, sedge meadow, and wet prairie communities as described by White (1978) and Nuzzo (1994). Previous hydropattern and vegetation information was documented by Bremholm (1993). The marsh community is dominated by *Acorus americanus* (sweet flag), *Sparganium eurycarpum*, *Scirpus fluviatilis*, *Carex lacustris*, and *Typha* spp. The sedge community is dominated by *Carex stricta* (tussock sedge). The wet prairie community is dominated by *Carex stricta* and *Calamagrostis canadensis* (bluejoint grass).

West Chicago Prairie (DuPage County) is located at the headwaters of Salt Creek and also consists of a mosaic of marsh, sedge meadow, and wet prairie wetland communities as described by White (1978) and Lampa (1993). The marsh community is dominated by *Scirpus acutus* (hard-stem bulrush) and *Ludwigia palustris* (marsh purslane). The sedge community is dominated by *Carex stricta*. The wet prairie community is dominated by *Carex stricta* and *Calamagrostis canadensis*. Water table, soils and plant community relationships have been described by Kelsey and Hootman (1992).

Hydrologic Monitoring

Hydrologic data for Grass Lake Marsh was provided from gauges at the Fox River and Chain O' Lakes maintained by the United States Geological Survey and Illinois Department of Transportation (IDOT 1970-1984; USGS 1995). Specifically, water level data for the vegetation transects was generated by correlating current water level data from the nearby station at Lake Villa (U.S. Geological Survey, 1994) with data available from the Fox River, located at the edge of the marsh. The correlation has a root mean square of 0.94. Hydrologic data for West Chicago Prairie was provided from monitoring wells maintained by Pat Kelsey at The Morton Arboretum (Lampa 1992, Kelsey 1995, and Kelsey and Hootman 1992). Water level data for the vegetation transects at this site was taken from existing well transects.

At Wadsworth Prairie, we installed groundwater monitoring wells and staff gauges along randomly selected transects parallel to the water level gradient. Each transect consisted of 5 to 7 shallow wells, staff gauges in open water areas, and two deep wells at either end of the transect to provide information on site stratigraphy and underlying aquifers. Each transect begins topographically higher and ends lower, crossing several plant communities. One shallow groundwater monitoring well was placed within the center of each community. Wells were constructed from one-inch diameter PVC pipe with a 30 cm long screen at the bottom, inserted in a hand-augured borehole lined with sand and capped with bentonite. Wells were permanently capped at the bottom and temporarily capped at the top to prevent rainfall entry. All of the shallow groundwater monitoring wells were made of the same materials and placed at the same depth. Wells were constructed uniformly to ensure that differences in water level readings between any two wells were actually differences in the water table, and not caused by differences in well construction. Because all of the study sites were State Nature Preserves or Natural Areas, wells were installed during the non-growing season (March) and soil cores were discarded off site to minimize siltation within

the wetlands. Wells were surveyed and purged 2 to 3 weeks after installation. Water levels at staff gauges and wells were recorded at least bi-weekly.

Vegetation Sampling

Herbaceous vegetation sampling was conducted in July of each year along permanent, randomly established transects parallel to the well transects. Vegetation was sampled according to the quadrat-transect method (Mueller-Dombois and Ellenberg 1974). Plant species were identified and corresponding cover was collected from $1/4m^2$ plots spaced every two meters along the transects. Species cover was then converted to midpoints based upon corresponding cover classes (Table 1) from (Daubenmire 1959). Plant species nomenclature follows Mohlenbrock (1985).

Table 1. Cover classes, cover class ranges, and range midpoints for vegetation plots.

Cover Class	Cover Class Range	Range Midpoint
1	0-1%	0.5%
2	1-5%	3.0%
3	5-25%	15%
4	25-50%	37.5%
5	50-75%	62.5%
6	75-95%	85%
7	95-100%	97.5%

Surveying

Elevations of vegetation plots and bench marks were determined using a Sokkia C75 level, surveying tripod, and range poles. Elevations were taken from the center of the plot and read to the thousandth of a meter. In plots with several distinct elevations (logs or sedge tussocks for example), readings were taken at each elevation with vegetation. All species were then assigned a cover class for each elevation.

Calculating the SEV

The SEV is a numerical representation of the hydropattern of any given elevation within the wetland. Hydropatterns were generated from shallow groundwater monitoring wells and staff gauges for each vegetation community or zone. Hydropatterns for each vegetation plot were modeled from the hydropatterns of the nearest wells or staff gauges and the elevation of that particular vegetation plot. Therefore, a hydropattern was assigned to the species within each plot. By integrating the hydropattern within the growing season to 30 cm below the soil surface, an SEV was calculated for each plot, and, therefore, for each species in each plot. The growing season for these sites was selected as the time between the last 28° F freeze (50 percent probability) in spring and first 28°F freeze (50 percent probability) in the fall. Of the three sites, West Chicago Prairie has the longest growing season, starting on April 12th and ending on October 23rd (USDA 1979). Grass Lake Marsh and Wadsworth Prairie have shorter growing seasons, starting on April 23rd and ending on October 28th (USDA 1970).

Identifying Species' Preferred Hydrologic Ranges

Although a species could occur over a wide range of SEVs at a particular site, we attempted to identify the SEVs where a species was particularly abundant (high coverage) or rare (low coverage) within a subset of its entire SEV range. We anthropomorphically assigned a species SEV range with high coverage its "preferred" range, medium coverage its "tolerated" range, and low or absent coverage its "avoided range." Identifying these three ranges for each species based on coverage required some data manipulation. First, cover data was plotted versus SEV classes for each species at each site. The observed coverages were divided into small, 2 m.day SEV intervals and a frequency histogram of species coverage within each interval was calculated. Second, to identify a species' preferred, tolerated, and avoided ranges at a particular site, we standardized the observed coverage for each SEV range by calculating a ratio of the observed coverage over a hypothetical uniform coverage (as if the species was uniformly distributed throughout the site irrespective of SEVs). A species uniform coverage was calculated as its total coverage divided by the number of SEV classes. Third, confidence intervals (95 percent) for the coverage ratio mean were identified for each species. Fourth, SEVs were categorized as "preferred, "tolerated," or "avoided" based on the location of the coverage ratio in relation to the confidence intervals. SEVs for coverages above the upper

confidence limit were designated as the "preferred" SEV range. SEVs for coverages within the 95 percent confidence limits were designated as the "tolerated" SEV range. SEVs for coverages below the lower confidence limit were designated as the "avoided" SEV range.

To determine whether the observed coverage distribution along the SEV gradient was significantly different from a hypothetical uniform distribution, the null hypothesis H_0: "Species coverage follows a uniform distribution along the SEV gradient" was tested against the hypothesis H_1: "Species coverage distribution along the SEV gradient is different from a uniform distribution." The Kolmogorov-Smirnov nonparametric test of distribution was used to compare the observed versus the uniform distributions at the 0.01 level. The test statistic is the maximum deviation of the observed coverage distribution versus the uniform coverage distribution. Significant deviation from the uniform coverage distribution are indicative of the hydrologic preferences explained above.

Results and Discussion

The results of our monitoring and associated discussion include preliminary community hydropatterns, preliminary community SEVs, preliminary species SEVs and some notes on SEV comparisons between sites and years.

Hydropatterns

Long-term water level elevations for marsh communities at West Chicago Prairie and Grass Lake Marsh are provided in Figure 2. The years 1990 and 1993 appear to have higher growing season water levels at both sites. At Grass Lake Marsh, 1993 was the wettest year, with an SEV of 117 m.day. Other years between 1990 and 1994 at Grass Lake Marsh showed very little variation in SEV, ranging from 87 to 96 m.day. At West Chicago Prairie, 1993 was also the wettest year, with a SEV of 164 m.day. Both 1991 and 1992 were the driest years during this period at West Chicago Prairie, with SEVs of 79 and 53 m.days, respectively. West Chicago Prairie hydropatterns include a marked drawdown during the late summer of most years. This drawdown is most pronounced during drier years, as expected. Such late summer drawdowns do not appear to occur at Grass Lake Marsh. Lower water levels occur during the winter in most years. This seemingly unnatural hydropattern is possibly caused by maintenance of artificially high water levels on the Fox River by the Corps of Engineers (presumably for summer navigation). The water levels are drawn down during the winter, presumably in preparation for spring rains. Although this hydropattern is different from the other sites and probably modified, the vegetation at the site must still adapt to the timing, depth, and duration of flooding. Because the Grass Lake Marsh community is of natural-area quality, we decided to include the site, even with its potentially modified hydropattern. Inclusion of this site increases project hydropattern variety, and allows species SEV comparisons between different hydropatterns.

Hydropatterns and estimated SEVs for wetland communities at Wadsworth Prairie 1995, West Chicago Prairie 1994, and West Chicago Prairie 1995 are provided in Figure 3. The water levels are relative to the soil surface elevation and 30 cm below the soil surface. Differences in the hydropatterns between the communities are discernible. As expected, the marsh communities have the highest water levels throughout the year, followed by the sedge meadow and wet prairie communities. Water level differences between the wetland communities are relatively small. For West Chicago Prairie specifically, water level elevation differences ranged from 12 to 18 cm between wet prairie (WP) and sedge meadow (SM), 13 to 17 cm between sedge meadow and shallow marsh (SMA), and 6 to 23 cm between shallow marsh and deep marsh (DMA) throughout the year.

Community Sum Exceedance Values

The relatively small differences between community hydropatterns translate into correspondingly larger differences in community SEVs because these water level differences are cumulative over the entire growing season. Although there were site differences in hydropatterns for each community, the community SEV ranges appear well differentiated. Among the sites, the mean SEV_{30} for wet prairie was 20.30 ± 9.01 m.day, for sedge meadow was 45.94 ± 10.64 m.day, for shallow marsh was 71.86 ± 10.91 m.day, and for deep marsh was 95.90 ± 8.88 m.day.

Quantifying a community hydropattern using the SEV allows for comparison to other communities and studies. For example, the mean SEV_{30} for wet prairie of 20.30 ± 9.01 m.day is comparable to an SEV_{40} of 28 m.day as documented by Spoor (1993) from a United Kingdom lowland wet grassland.

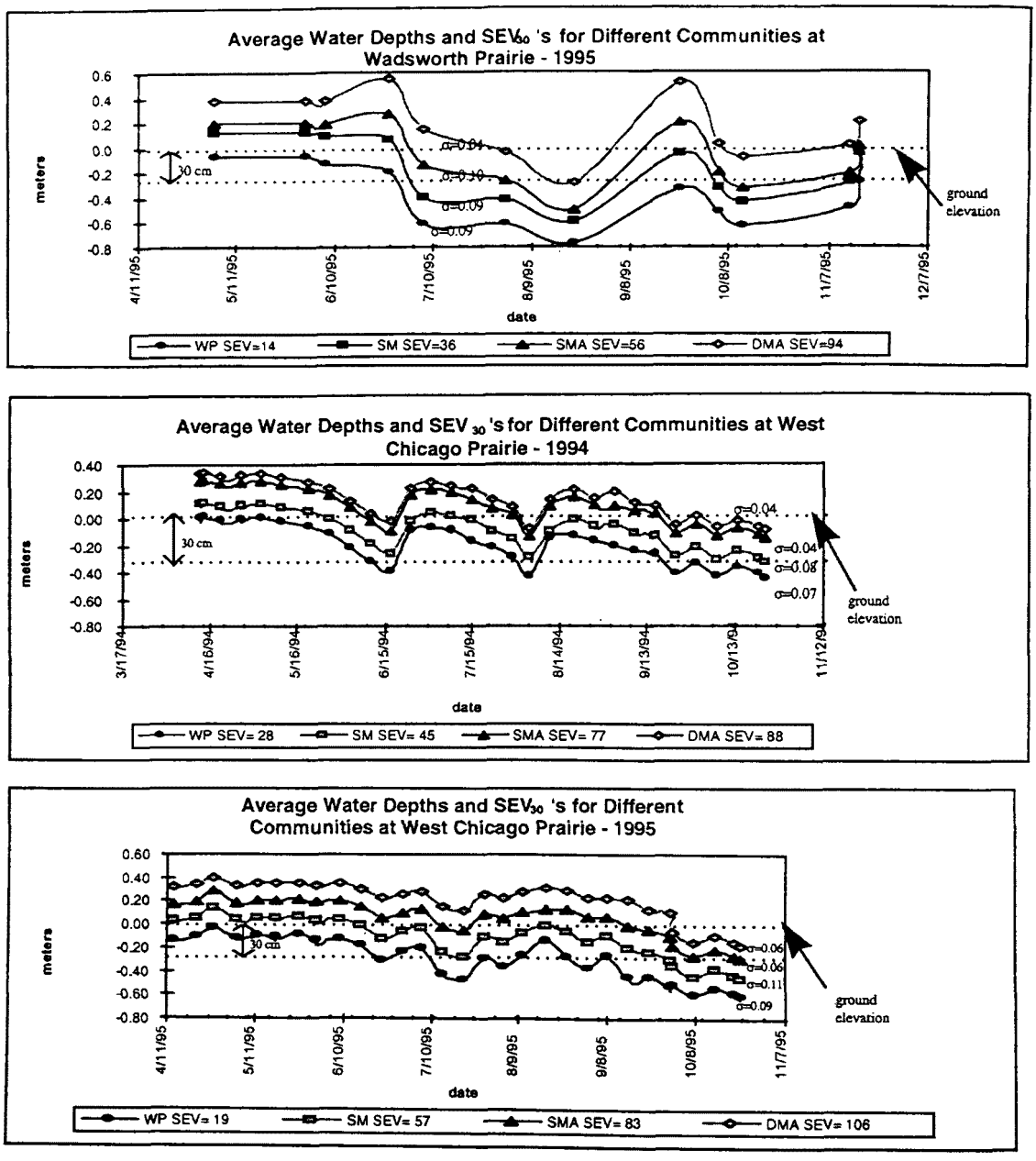

Figure 3. Water depths and sum exceedance values for wet prairie (WP), sedge meadow (SM), shallow marsh (SMA), and deep marsh (DMA) communities at Wadsworth Prairie, West Chicago Prairie 1994, and West Chicago Prairie 1995.

Individual Species' SEV Ranges

SEV ranges were also computed for individual species at the three sites. Using *Helianthus grossoserratus* (saw-tooth sunflower) at West Chicago Prairie 1994 as an example (Figure 4), a ratio of observed over expected coverage is plotted versus the SEV and preferred, tolerated, and avoided ranges were compiled in relation to the confidence interval as described in the methods section. Based on the coverage distribution we recorded for *Helianthus grosseserratus* at West Chicago Prairie in 1994, we designated its "preferred" SEV range as 1 to 28 m.day, "tolerated" range as 28 to 38 m.day, and "avoided" range greater than 38 m.day. It is important to note that the SEV ranges we have designated as preferred, tolerated, and avoided, are based on this limited data and are preliminary. These ranges are a starting point. As additional data from wetter and drier conditions is added from additional sites and years, SEV ranges for various species are expected to change.

Analyzing individual species' SEV ranges allows us to make preliminary comparisons between many species occurring along a theoretical or real hydrologic gradient. Preliminary species' SEV ranges based on our field data are provided for species at Grass Lake Marsh in 1994 (Figure 5), West Chicago Prairie in 1994 (Figure 6), West Chicago Prairie in 1995 (Figure 7), and Wadsworth Prairie in 1995 (Figure 8). The abrupt delineation between species ranges is not meant to be indicative of natural conditions, but was used for illustrative purposes only. The line graph of *Helianthus grosseserratus* in Figure 4 is condensed into the bar graph for the same species in Figure 6.

Evaluating each site individually, species occurrence along the hydrologic gradients appear to make biological sense. As would be expected looking at species from West Chicago Prairie 1994 (Figure 6), *Helianthus grosseserratus* (a common species of wet prairie communities) is located at the drier end of the SEV gradient (< 38m.day) than species like *Scirpus acutus* (hard-stem bulrush), a common species of deep marsh communities, that occurred at higher SEVs (> 49 m.day).

Within a single community, species' preferred SEV ranges also follows this pattern. For the marsh community, the order of species abundance from the drier end to the wetter end of the SEV gradient is relatively consistent between sites. Deep water marsh plants like *Scirpus acutus* are consistently located at higher SEV ranges with average preferred SEV ranges of 66 - 120 m.day. On the other hand, species like *Carex lacustris* (lake sedge) are located at lower average preferred SEV ranges with average preferred SEV ranges of 27 to 58 m.day, among all three sites. Additionally, the sequence of species between these SEV endpoints like *Acorus americanus*, *Sagittaria latifolia* (common arrowhead), *Eleocharis smallii* (marsh spikerush), and *Sparganium eurycarpum*, seems to make biological sense, and support previously published community descriptions (Weller 1987; Squires and van der Valk 1993).

We used the Kolmogorov-Smirnov non-parametric test of distribution ($\propto = 0.01$) to test species' observed versus expected values to determine if a species' observed distribution is different than an expected uniform distribution. Based on these tests, almost all of the species were not uniformly distributed throughout the site. Almost all of the species were significantly more abundant at some SEVs than others. These results suggest that the SEV method can quantify a species location along a hydrologic gradient.

Three species, though, failed to reject the test of uniform distribution, *Polygonum amphibium* (water smartweed), *Scirpus fluviatilis*, and *Acorus americanus*. *Polygonum amphibium* failed to reject the test at Wadsworth Prairie 1995 and West Chicago Prairie 1994, while it did reject the test at West Chicago Prairie 1995. This disturbance-tolerant species rapidly germinates on moist soils. As water levels drop over the growing season, this species is able to germinate, growing quickly at a range of elevations within a site, and therefore at a broad range of soil exceedance values. Why, then, was this species not found uniformly distributed at West Chicago Prairie in 1995? A possible explanation includes variation within the site's hydropattern. Water levels during 1995 (Figure 2) did not drop as much as the previous year, providing less habitat for this species to colonize. In this case, SEV does not adequately explain this species occurrence along a hydrologic gradient. The rate, depth, and timing of drawdown might better explain its coverage distribution.

Scirpus fluviatilis and *Acorus americanus* both failed to reject the null hypothesis of uniform distribution at Grass Lake Marsh. Although *Scirpus fluviatilis* is somewhat tolerant to disturbance and fluctuating water levels, *Acorus americanus* is not as tolerant. A partial explanation to these species distribution may exist in the site's topography. Whereas

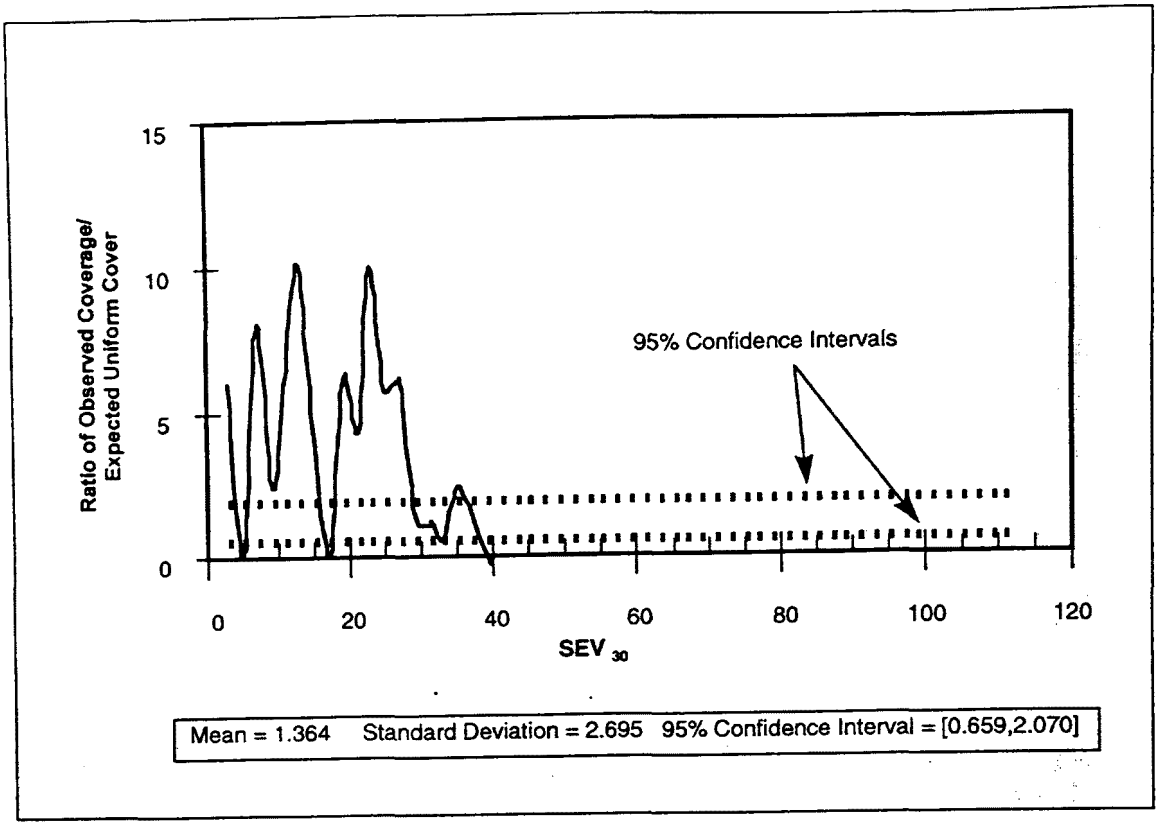

Figure 4. Standardized coverage versus sum exceedance values for *Helianthus grosseserratus* at West Chicago Prairie 1994.

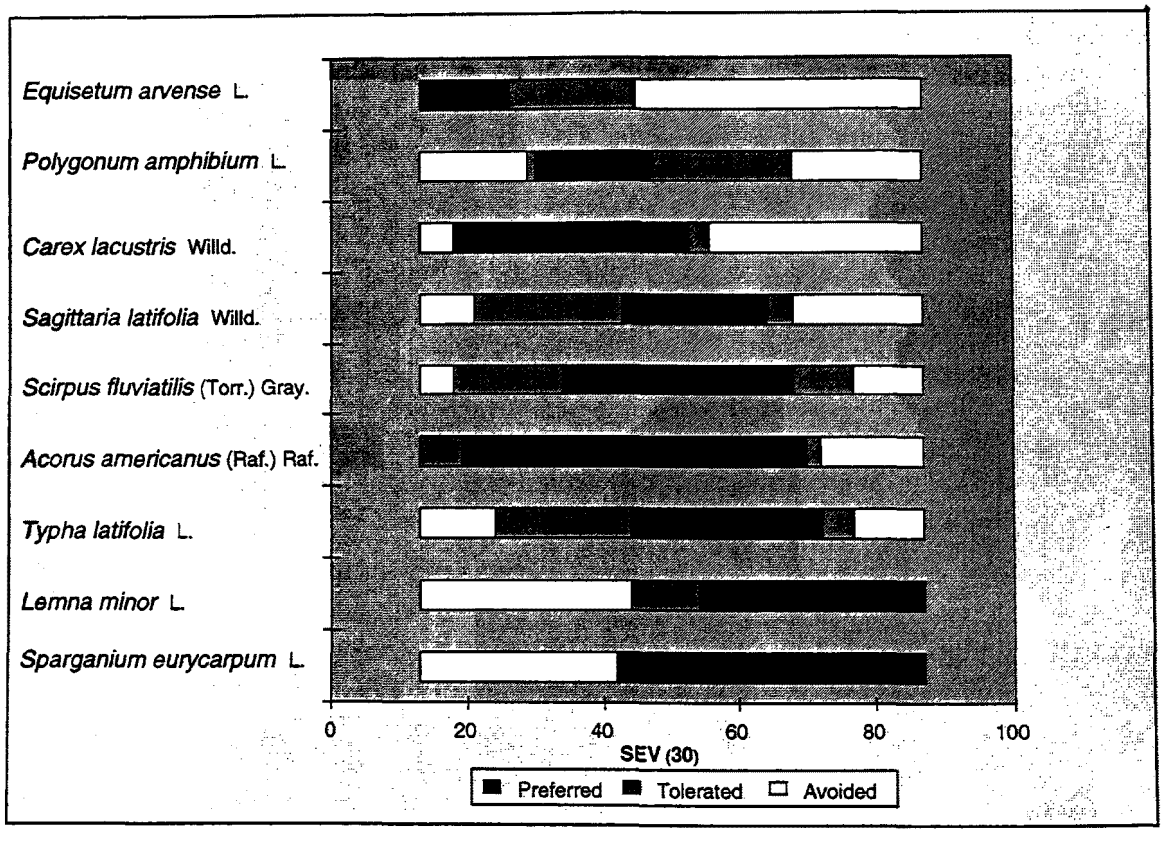

Figure 5. Sum exceedance value ranges for selected species at Grass Lake Marsh 1994.

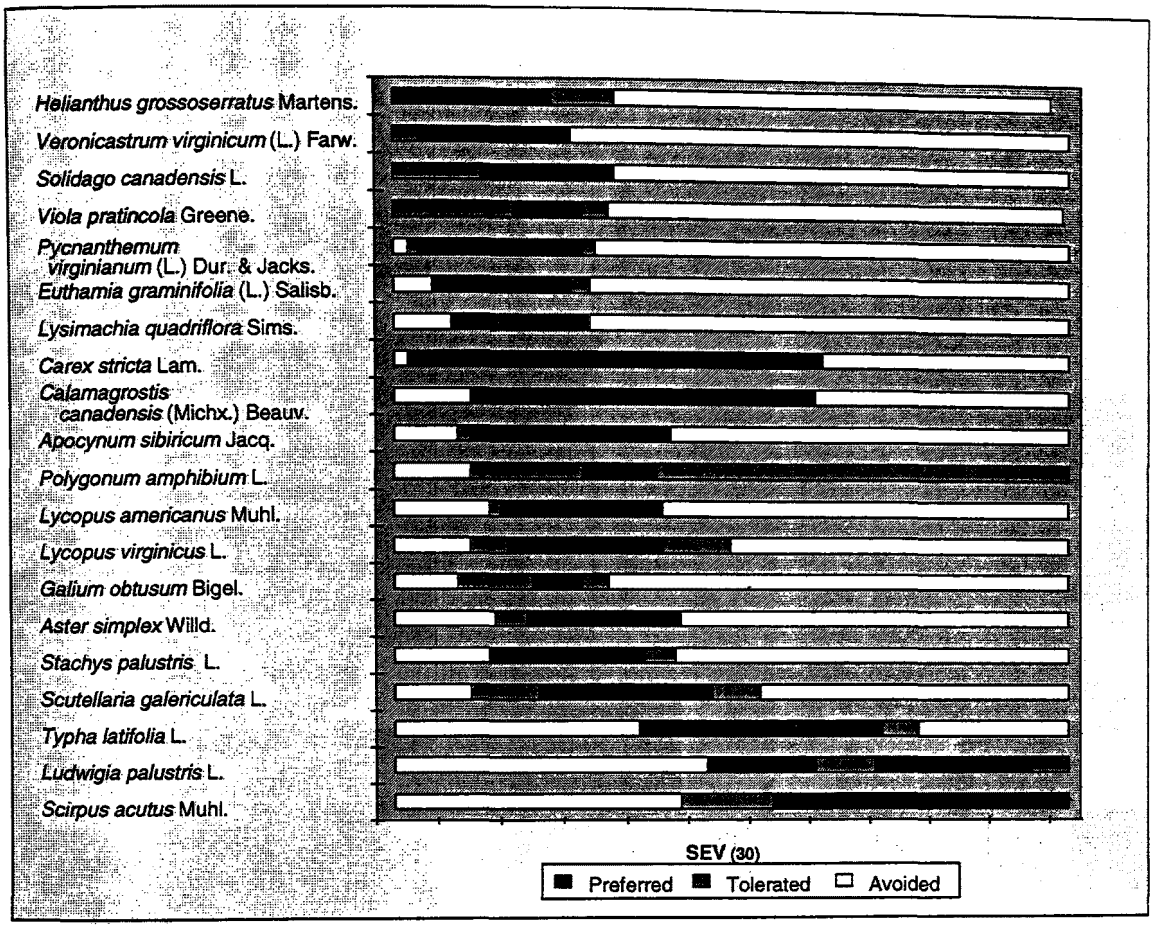

Figure 6. Sum exceedance value ranges for selected species at West Chicago Prairie 1994.

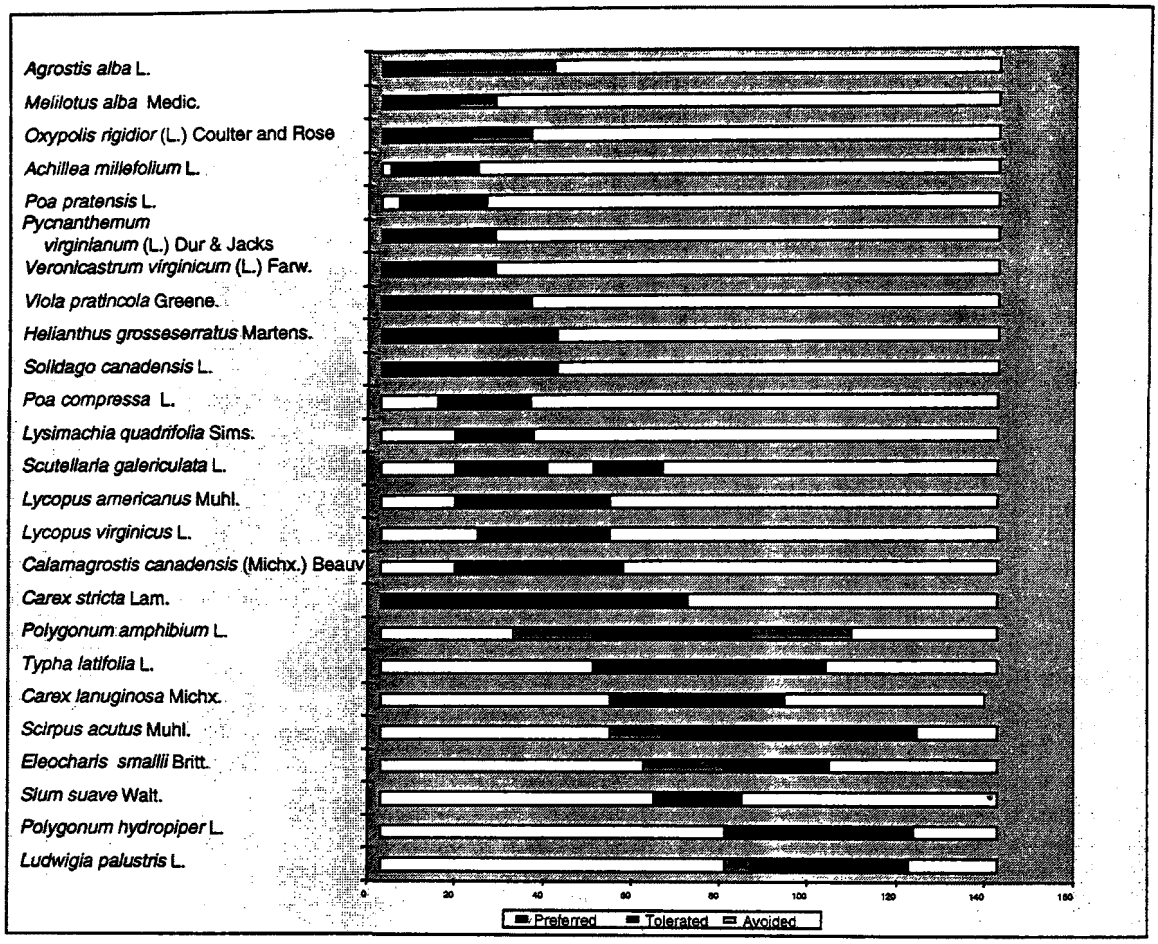

Figure 7. Sum exceedance value ranges for selected species at West Chicago Prairie 1995.

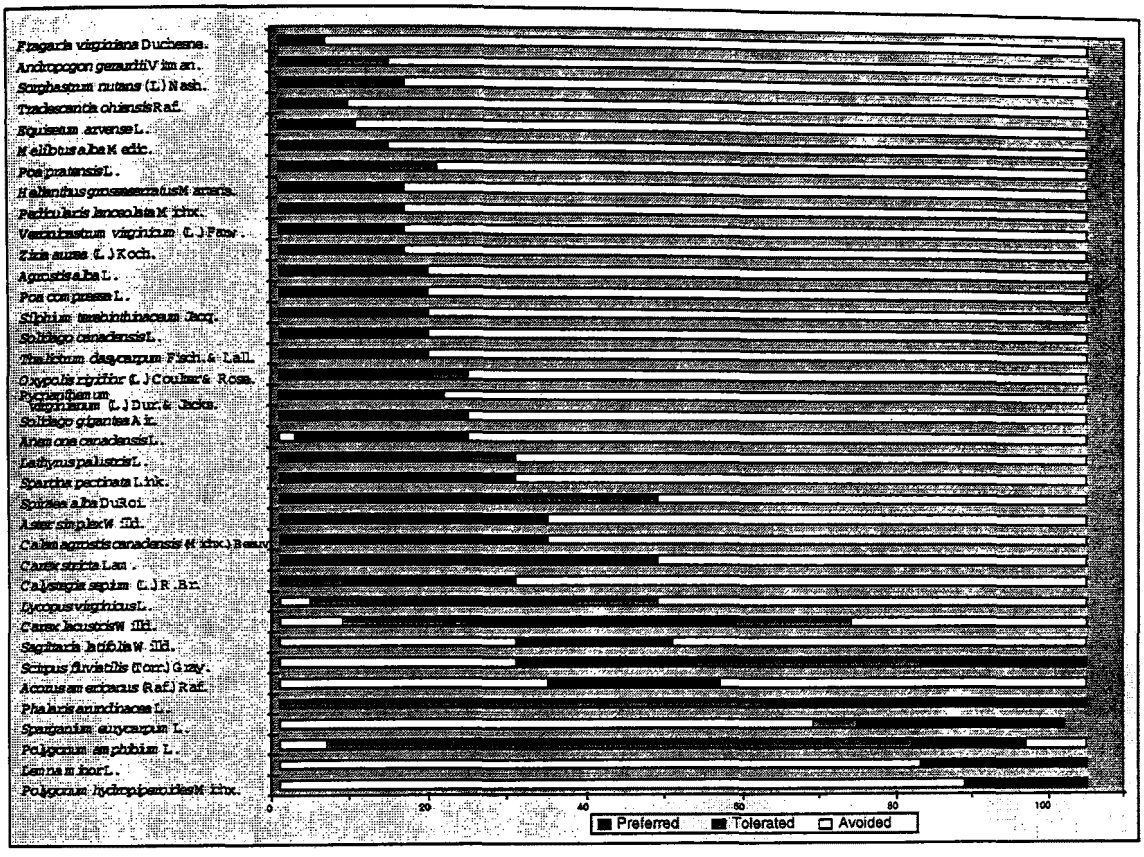

Figure 8. Sum exceedance value ranges for selected species at Wadsworth Prairie 1995

the marsh communities SEV ranges at the other sites were approximately 40 to 110 m.day, the marsh community SEV range at Grass Lake Marsh was approximately 20 to 85. The site is exceptionally flat (0.25 meters of elevation change over a 200 meter transect). Additionally, limitations in the range of our surveying equipment inhibited us from sampling the full range of SEV values within the marsh communities, unlike our representative sampling at West Chicago Prairie and Wadsworth Prairie.

Conclusions

There are several issues we need to address as we continue data analysis and field work. Obviously, these results are preliminary, representing only a select number of communities and sites, over only a few years. To increase the applicability of the data, we need to increase community, geographic, and temporal coverage. Recording seedlings (as we did), in addition to mature individuals during vegetation sampling, may not provide the clearest picture of a species' "preferred" SEV range. Including seedlings (that may not survive the entire growing season) increases a species' observed SEV range. Additionally for some species, the SEV for the entire growing season may not explain species distributions as well as an SEV of the first half of the growing season, the previous years SEVs, etc.

Based on our preliminary results, the SEV appears to be a usable method to quantify the hydropattern and species and community occurrence along a hydrologic gradient. Sum exceedance values for the wet community averaged 20.90 m.day, sedge meadow community averaged 45.94 m.day, shallow marsh community averaged 71.86 m.day, and deep marsh community averaged 95.90 m.day. Sum exceedance values for species along the hydrologic gradients measured appear to reflect their intuitive location along a hydrologic gradient.

Regardless, we feel that in addition to the importance of documenting the vegetation and hydrology for our remaining high-quality wetland communities, this work has immediate practical applications in wetland management, restoration, and creation. For wetland restoration and creation projects, a baseline wetland vegetation data set is valuable for identifying target communities, target hydropatterns, and species planting locations (combined with site hydropattern information) with respect to water levels, at any time for both wet and dry years. The data can also be used when establishing project design criteria and performance standards. Additionally, quantitative information on species' hydrologic preferences provides wetland managers an additional tool in managing for target species and communities, and discerning between natural wetland community dynamics and long-term, disturbance-caused, hydrologic change.

Acknowledgments

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Literature Cited

Bremholm, T. 1993. Evaluation of techniques for establishment of sedge meadow vegetation. Master Thesis. Iowa State University. 93 pp.

Curtis, J. T. and R. P. McIntosh. 1951. An upland forest continuum in the prairie-forest border region of Wisconsin. Ecology 32:479-496.

Doss, P. K., 1993. The nature of a dynamic water table in a system of non-tidal, freshwater coastal wetlands. J. Hydrology 141:107-126.

Daubenmire, R. 1959. A canopy coverage method of vegetational analysis. Northwest Science 33:43-64.

Ebinger, J. E., R. D. Andrews, H. C. Nilsen. 1977. Papers on the ecology of the Chain O' Lakes region, Illinois. Eastern Illinois University. Charleston, IL.

Evans, R. O., R. W. Skaggs and R. E. Sneed. 1990. Normalized crop susceptibility factors for corn and soybean to excess water stress. Transactions of the American Society of Agricultural Engineers 33(4):1153-1161.

Galatowitch, S. M. and A. G. van der Valk. 1994. Restoring prairie wetlands: an ecological approach. Iowa State University Press. 246 pp.

Gosselink, J. G. and R. E. Turner. 1978. The role of hydrology in freshwater wetland ecosystems. pp. 63-78, In R. E. Good, D. F. Whigham and R. L. Simpson (ed). Freshwater Wetlands: Ecological Processes and Management Potential. Academic Press, Rutgers University, New Brunswick, NJ.

Gowing, D. J. G., Spoor, G. and Mountford, J. O. In press. The influence of minor variations in hydrological regime on plant communities - implications for water management. In C. Joyce and M. Wade (eds). European Lowland Wet Grasslands: Biodiversity, Management and Restoration. International Center for Landscape Ecology. Loughborough, England.

Gowing, D. J. G. and Spoor, G. In press. The effect of water table depth on the distribution of plant species on lowland wet grassland. In B. Sherwood P. Jose, and R. Bailey (eds.) United Kingdom Floodplains. 1995. The Linnaean Society. London, England.

Gunderson, L. H., J. R. Stenberg and A. K. Herndon. 1986. Tolerance of five hardwood species to flooding regimes. pp. 119-133, In R. Herrmann (ed). Interdisciplinary Approaches to Freshwater Wetlands. Michigan State University Press.

Hardjoamidjojo, S., R. W. Skaggs and G. O. Schwab. 1990. Corn yield response to excessive soil water conditions. Transactions of the American Society of Agricultural Engineers Vol. 33.

Howell, T. A., E. A. Hiler, O. Zokezzi and R. J. Ravello. 1976. Grain sorghum response to inundation at three growth stages. Trans. American Society of Agricultural Engineers, 19(5):876-880.

Illinois Department of Transportation. 1970-1984. Division of Water Resources. Water Resources Data for Illinois: water years 1970 - 1984.

Johnson, W. C., T. L. Sharik, R. A. Mayes, and E. P. Smith. 1987. Nature and cause of zonation discreteness around glacial prairie marshes. Canadian Journal of Botany 65: 1622-1632.

Kelsey, P. D. and R. G. Hootman. 1992. Relationships between water tables, plant communities. and hydric soils: West Chicago Prairie, West Chicago, IL. Soil Survey Horizons 33(3):53-58.

Kelsey, P. D. 1995. Unpublished data on West Chicago Prairie water tables.

Kentula, M. E., R. P. Brooks, S. E. Gwin, C. C. Holland, A. D. Sherman and S. J. C. 1992. An Approach to Improving Decision Making in Wetland Restoration and Creation. U.S. Environmental Protection Agency. EPA/600/R-92/150.

Koyha, K. D., D. T. Shaw and K. W. Weiler. 1995. Hydrologic design of a wetland: advantages of continuous modeling. Ecological Engineering 4(2):99-116.

Kusler, J. A., and G. Brooks (eds.). 1987. Proceedings of National Wetlands Symposium: Wetland Hydrology. Omnipress, Chicago, IL.

Lampa, W. 1992. Results of a three year study of wetlands at the West Chicago Prairie. Unpublished report. 16 PP + appendices.

Maurizi, S., and F. Poillon. 1992. Restoration of aquatic ecosystems. National Academy Press, Washington, DC. 732 pp.

Mitsch, W. and J. Gosselink. 1993. Wetlands. Van Nostrand Reinhold. NY. 722 pp.

Mohlenbrock, R. H. 1986. Guide to the Vascular Flora of Illinois. Southern Illinois University Press. Carbondale. 507 pp.

Mueller-Dombois, D. and H. Ellenberg. 1974. Aims and methods of vegetation ecology. John Wiley and Sons. NY.

National Research Council. 1995. Wetlands characteristics and boundaries. National Academy Press, Washington, DC. 307 pp.

Nelson, D. C. and R. C. Anderson. 1983. Factors related to the distribution of prairie plants along a moisture gradient. American Midland Naturalist 109(2):367-375.

Niering, W. A. 1987a. Hydrology, disturbance, and vegetation Change. pp. 50-54, In J. A. Kusler and G. Brooks (eds.). National Wetlands Symposium-Wetland Hydrology.

Niering, W. A. 1987b. Wetlands hydrology and vegetation dynamics. National Wetlands Newsletter 9(2):9-10.

Nuzzo, V. 1994. Management plan for Wadsworth Prairie Nature Preserve and Wadsworth Savanna. Lake County, Illinois. Unpublished report to the Lake County Forest Preserve District.

Pickett, S. T. A. and F. A. Bazzaz. 1978. Organization of an assemblage of early successional species on a soil moisture gradient. Ecology 59(6):1248-1255.

Robel, R. J. 1962. Changes in submerged vegetation following a change in water level. Journal of Wildlife Management 26(2):221-224.

Schwegmann, J. E. (ed.) 1973. Comprehensive plan for the Illinois Nature Preserves System, Part 2. The Natural Divisions of Illinois, Illinois Nature Preserves Commission, Rockford. 32 pp.

Shaw, W.H. 1974. Weighted moisture stress index for corn in Iowa. Iowa State J. Research 49(2): 101-114.

Shipley, B., P. A. Keddy, and L. P. Lefkovitch. 1991. Mechanisms producing plant zonation along a water depth gradient: a comparison with the exposure gradient. Canadian Journal of Botany 69: 1420-1424.

Spence, D. H. N. 1982. The zonation of plants in freshwater lakes. Advances in Ecological Research. 12:37-125.

Spoor, G. In Preparation. Water regime requirements of wetland plant species. Report in progress, Silsoe College, Cranfield Institute of Technology, Silsoe, England. MK45 4DT.

Squires, L. and A. G. van der Valk. 1992. Water-depth tolerances of the dominant emergent macrophytes of the Delta Marsh, Manitoba. Canadian Journal of Botany 70:1860-1867.

Sudar, R.A., et. al. 1979. A predictive model of water stress in corn and soybeans. American Society of Agricultural Engineers Paper No. 79-2004.

Thunhorst, G. A. 1993. Wetland planting guide for the northeastern United States. Environmental Concern, Inc. St. Michaels, MD. 179 pp.

United States Department of Agriculture, Soil Conservation Service. 1979. Soil Survey of DuPage and part of Cook counties, Illinois. Agricultural Experiment Station Report No. 108 217 pp. + maps.

United States Department of Agriculture, Soil Conservation Service. 1970. Soil Survey of Lake County. Illinois Agricultural Experiment Station Soil Report No. 88. 82 pp + tables and maps.

United States Geological Survey. 1995. Water Resources Data for Illinois Open File.

van der Valk, A, (ed.) 1989. Northern prairie wetlands. Iowa State University Press, Ames, IA. 400 pp.

van der Valk, A. G., L. Squires and C. H. Welling. 1994. Assessing the impacts of an increase in water level on wetland vegetation. Ecological Applications 4(3):525-534.

Weller, M. W. 1987. Freshwater Marshes. University of Minnesota Press. Minneapolis. 147 pp.

Wentz, W. A., R. L. Smith and J. A. Kadlec. 1974. State-of-the-art survey of marsh plant establishment techniques: induced and natural; volume II, a selected annotated bibliography on aquatic and marsh plants and their management. US Army Engineers Waterways Experiment Station, Vicksburg, MS.

White, J. 1978. Illinois natural areas inventory technical report. Vol. 1: Survey methods and results. Illinois Natural Areas Inventory, Urbana. 426 pp.

Whitlow, Thomas H., and Richard W. Harris. 1979. Flood tolerance in plants: a state-of-the-art review. US Army Corps of Engineers. NTIS, E-79-2.

Zimmerman, J.H. 1987. A multi-purpose wetland characterization procedure, featuring the hydroperiod. pp. 31-49 In Kusler, J. A., and G. Brooks (eds.). Proceedings. National Symposium Wetland Hydrology, Association of State Wetland Managers, Chicago, IL.

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A Successful East-Central Illinois Kettle Marsh Restoration and Its Developing Wetland Plant Community

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Abstract: **The restoration of a kettle marsh in east-central Illinois is detailed and the development of its plant community is documented. Underground drainage tiles were dismantled in 1990, restoring wetland hydrology to the study area. Wetland vegetation rapidly colonized the site, as indicated by five years of vegetation sampling data beginning in 1991. Ninety-five plant species have been recorded in the wetland, of which 68 percent are hydrophytic and 78 percent are native to central Illinois. General trends in plant community development are discussed, including the effects of extremes in rainfall on species composition. Exotic and aggressive plant species have generally not been a major concern, although some efforts aimed at cattail (*Typha* spp.) control have been attempted. A controlled burn conducted in 1995 appeared to be more detrimental than beneficial.**

Introduction

Wetlands once totaled about 10†million acres in Illinois or about 23 percent of the land surface (Bell 1981). However, it is estimated that over 90 percent of this wetland area has already been lost (Suloway and Hubbell 1994), with some estimates putting the loss as high as 99.5 percent (Bell 1981). Most of the remaining wetland habitat is severely degraded, less than 6,000 acres remain in their natural, undisturbed state (IDENR 1994). Most of this loss and degradation has resulted from Illinois agriculture. The conversion of Illinois wetlands to prime farmland has been accomplished principally through drainage. Historically encouraged by legislation, over 5 million acres of wetlands had been drained by Illinois drainage districts as of 1960 (Bell 1981).

In response to this wetland crisis, numerous wetland creation and restoration projects have been undertaken across Illinois. Many of these attempts involve the conversion of agricultural land back to wetland. Included in this collection are scattered, small-scale projects, as well as larger undertakings along Illinois rivers, where thousands of acres of farmland are to be converted or reverted back to wetland. All of these restoration or creation projects, no matter the scale, are of importance in trying to restore or replace the functions and values lost by the destruction and degradation of Illinois wetlands.

The restoration project described here is a small-scale attempt to replace lost wetland habitat through an approach low in cost, labor, and maintenance. Also, by systematically monitoring the restoration and subsequent community development processes, important information can be gained for use in similar restoration efforts and for the promotion of restoration science in general.

Site History and Methods

Illinois Natural History Survey researchers have been studying a restored marsh community in east-central Illinois. This restored wetland is the result of a cooperative effort between the Illinois Natural History Survey, the Champaign County Forest Preserve District, and regional volunteer organizations, such as the Nature Conservancy and the Grand Prairie Friends. Additional funding was provided by the Illinois Department of Transportation.

Prior to the restoration process, the study area was principally fallow, abandoned pasture, having not been grazed for 10 years or more. Underground drainage tiles, in place for at least several decades, directed water to an excavated ditch running through the center of the area. Some wetland plant species, including cattail (*Typha* spp.) and willow (*Salix* spp.), persisted along this ditch, although much of the vegetation likely consisted of non-wetland species. In 1990, the drainage tile system was dismantled, restoring "natural" hydrology to the site. Some excavation was required in this process, creating a shallow depression at the eastern edge of

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the site. Water now collects in this low-lying basin of approximately 7 acres, persisting on the site throughout the year. Only in late summer does the entire wetland occasionally dry out.

Hydrophytic vegetation began to return to the site almost immediately after the dismantling of the drainage tile system. For the purposes of this study, hydrophytic plant species are those classified, in Illinois, as facultative or wetter by Reed (1988). Some native plant species were also introduced into the wetland via seed, rootstock, and/or plantings, but with virtually no success. These efforts are not thought to have significantly affected the vegetation composition of the restored wetland.

Vegetation sampling in the wetland restoration was initiated in 1991 and has continued on a yearly basis through 1995. Seven vegetation transects were established at 30†m intervals, crossing the length of the wetland. Herbaceous vegetation was sampled within one-quarter meter plots at 5 m intervals along each transect. Within each plot, plant species were identified and the estimated percent aerial coverage was determined (Daubenmire 1959). These percent aerial coverages were then converted to standard cover classes (Table 1). Mid-points of the ranges of aerial coverage were used in further calculations.

Table 1. Cover classes used in herbaceous vegetation sampling (Daubenmire 1959).

Cover Class	Range of Coverage (%)	Mid-point of Range (%)
6	95-100	97.5
5	75-95	85.0
4	50-75	62.5
3	25-50	37.5
2	5-25	15.0
1	1-5	3.0
+	< 1	0.5

Dominance for an individual plant species is calculated as the total aerial coverage for all quadrats divided by the area sampled. Relative dominance for a species is the dominance for that species divided by the total dominance for all species, times one hundred. Frequency for a given species is calculated as the number of quadrats in which the species occurs divided by the total number of quadrats sampled. Relative frequency for a species is the frequency for that species divided by the total frequency values for all species, times one hundred. The importance value for any species is defined as its relative dominance plus its relative frequency, divided by two. All formulas and calculations follow those defined in Cox (1985).

A controlled burn conducted on the surrounding prairie and savanna restorations was allowed to extend into the restored wetland in November 1994. Vegetation sampling results from 1995 show any effects of this fire. Also, a one-time attempt to physically remove cattail from the restoration was conducted in 1992.

Results and Conclusions

Ideally, restorationists and site managers would like to predict the success of wetland creations and restorations and the development of their associated plant communities. Unfortunately, this is not an exact science. Studies have documented wetland community development on these types of sites (Reinartz and Warne 1993, Wilhelm and Wetstein 1993), but generalizations are difficult. In a broad sense, newly developing wetlands are often characterized by opportunistic, invading plant species that may give way to species characteristic of more stable wetlands as time passes (Mitsch and Gosselink 1986, Odum 1988). Regional, local, and site-specific influences, however, all affect wetland community development (Willard and Hiller 1990). Weather, species interactions (e.g., competition, allelopathy), immigration, and emigration all contribute to this constant change to which plants are continually trying to adapt (van der Valk 1981). Nonetheless, some patterns and trends in vegetation establishment, change, and community development are evident, particularly in individual cases.

Colonization and Vegetation Establishment

In this restoration, development of the wetland plant community proceeded at a rapid pace from the very beginning. In 1991, one year after the restoration of "natural" hydrology, 43 different plant species were present (Table 2). Species composition in this first-year community was also favorable. 72 percent of the plant species were classified as hydrophytic and 73 percent were native to central Illinois. As indicated by importance values (Appendix A), the most prevalent plant species sampled in that first year are generally considered to be valuable or desirable wetland species, with the

possible exception of common ragweed (*Ambrosia artemisiifolia*). Other prevalent plant species included beggar-ticks (*Bidens tripartita*), spikerushes (*Eleocharis palustris* and *E. obtusa*), marsh yellow cress (*Rorippa islandica*), water plantain (*Alisma plantago-aquatica*), millet (*Echinochloa crus-galli*), sedges (*Carex frankii* and *C. tribuloides*), and duckweed (*Lemna minor*). The quick return of wetland vegetation to the restoration is thought to be attributable to the spread of seed from the previously existing wet ditch, although waterfowl and other water birds may have also been the source of some seed. The drained area had likely been dry too long for the survival of any viable wetland seeds in the below-ground seed bank, although the seeds of many wetland species may remain viable beneath the soil surface for many years (Galatowitsch and van der Valk 1994).

Trends in Vegetation Composition and Prevalence

Changes in the developing plant community, some of which appear to be directly related to rainfall, have been noted from year to year. Although detailed hydrologic data was not collected until recently, general, anecdotal information is known. Of particular note are the years of 1992 and 1993. Rainfall during 1992 was very slight, leading to dry conditions and very low water levels in the restored wetland. As a result of this drought, three obligate wetland plant species, water plantain, arrowhead (*Sagittaria latifolia*), and duckweed, disappeared from sampling results (Table 3). Other plants, such as ragweed, swamp marigold (*Bidens aristosa*), and smartweed (*Polygonum* spp.), increased greatly during this time, apparently responding to the drier conditions and the disappearance of competing plant species. Plant species diversity dropped in 1992, as did the percent of hydrophytic species and the percent of species native to central Illinois (Table 2), all likely attributable to a lack of water.

Heavy rains returned in 1993, resulting in very wet conditions at the wetland restoration site. Species diversity again declined, but the percent of hydrophytic species and the percent of species native to central Illinois both increased (Table 2). Excessively wet conditions may have caused the extirpation of some plant species favoring drier conditions. The three obligate wetland plant species that disappeared in 1992 all returned in 1993, while common ragweed, swamp marigold, and smartweed all dropped substantially. Common ragweed, in fact, disappeared completely from sampling results.

Some general trends in vegetation change and plant community development not thought to be directly related to fluctuation in rainfall have also been noted. Marsh yellow cress, a common moist-soil herb, has shown a large decline throughout the five-year sampling period. Exhibiting an importance value of 10.75 in the first year of sampling, it has decreased rapidly, to an importance value under one in the last three years of sampling. Marsh yellow cress was successful in colonizing and exploiting the newly restored wetland, but is apparently not able to thrive in the developing wetland plant community. It may not compete well with dense growth of taller plant species such as beggar-ticks, cattail, and swamp marigold. The opposite trend has been exhibited by swamp marigold, a tall, moist-soil annual. Initially exhibiting an importance value of only 1.34 in 1991, it has grown steadily in prevalence throughout. Conditions at the restoration site are apparently favorable for survival and growth of this species, enabling its continued population growth through reseeding.

Invasive and Exotic Vegetation

Potentially troublesome and undesirable species have generally not been a major problem thus far. Cattail has been the only species to cause moderate concern. Although native and an important part of many wetland communities, cattails can spread rapidly, excluding other plant species and decreasing diversity such that a virtual monoculture is sometimes the result (Baldassarre and Bolen 1994, Galatowitsch and van der Valk 1994). This may be of particular concern in the developing plant community of a restored wetland where the establishment of other species may be prevented by cattail (Galatowitsh and van der Valk 1994). Cattail, however, is an important component of the restored marsh and is the only common, large emergent in the wetland. It is a valuable food source for muskrat (*Ondatra zibethica*) and important habitat for birds such as bitterns, rails, grebes, and waterfowl (Baldassarre and Bolen 1994, Herkert 1992, Martin et al 1951). Reed canary grass (*Phalaris arundinacea*) is another species found within the wetland restoration that has the potential to be problematic.

Capable of tolerating extensive periods of inundation, it can sometimes dominate shallow marshes (Galatowitsch and van der Valk 1994).

Cattail and reed canary grass were initially of no concern in the restored wetland. Cattail, however, began to show a steady increase. The physical

Table 2. Number of plant species, percent of species hydrophytic, and percent of species native to central Illinois in the restored wetland, 1991-1995.

	Number of Species	% of Species Hydrophytic[1]	% of Species Native to central Illinois[2]
1991	43	72	73
1992	37	65	65
1993	29	93	83
1994	34	82	82
1995	53	64	70

[1] Reed (1988).
[2] Taft et al. (1993).

Table 3. Importance values for selected plant species at the restored wetland, 1991-1995.

	1991	1992	1993	1994	1995
Alisma plantago-aquatica	8.82	0.00	6.33	12.07	0.60
Ambrosia artemisifolia	4.73	22.46	0.00	0.67	5.56
Bidens spp. (total)	23.91	29.79	12.67	21.49	24.84
Bidens aristosa	1.34	6.59	3.28	6.03	13.73
Bidens tripartita	22.57	23.20	8.91	15.33	11.11
Carex spp. (total)	7.48	4.32	6.26	2.97	5.06
Echinochloa crus-galli	4.41	2.15	0.57	1.58	1.03
Eleocharis spp. (total)	22.51	19.56	41.99	27.88	25.03
Eleocharis palustris	18.06	17.83	41.34	25.08	22.47
Leersia oryzoides	2.15	0.96	1.04	1.66	3.35
Lemna minor	3.27	0.00	11.73	17.58	7.93
Phalaris arundinacea	0.16	0.12	1.24	0.44	2.41
Polygonum spp. (total)	0.91	9.67	0.88	1.76	8.42
Rorippa islandica	10.75	2.09	0.00	0.77	0.82
Sagittaria latifolia	0.42	0.00	2.69	4.76	0.35
Typha spp. (total)	0.25	0.12	4.50	3.27	4.32

removal of cattail in 1992 was effective, but was deemed too labor intensive for long-term maintenance. Although cattail had not really reached problematic levels, the controlled burn conducted on the adjacent prairie and savanna restorations was allowed to burn into the wetland restoration to help control cattail and, secondarily, reed canary grass.

Fire Effects

Almost no positive effects were evident from the controlled burn. Cattail and reed canary grass both showed increased importance values after the fire (Table 3). Cattail can apparently be controlled by fire only under limited circumstances. For cattail to be killed, the fire must be hot enough, or the marsh must be dry enough, to burn into the sediment or peat layer of the wetland (Beule 1979, Smith and Kadlec 1985).

Reed canary grass can be controlled by repeated exposures to fire (Hutchison 1990, Galatowitsh and van der Valk 1994). This one-time burn, however, appears to have had the opposite effect or, at best, no effect. Although still a relatively minor component of the wetland plant community, reed canary grass did rise to an importance value of 2.41 in 1995, almost six times the previous year's level. Observations indicated that this was attributable to the spread of existing colonies. No new colonies were observed.

Two important wetland plant species, water plantain and arrowhead, showed a definite decline after the burn. Water plantain, a common, moist-soil, emergent, had been prevalent in the wetland restoration throughout, with the exception of the drought year (1992). However, 1995's sampling results indicated a tremendous drop, from an importance value of 12.07 in 1994 to only 0.60. Similarly, the emergent arrowhead showed a large decrease after the burn. Arrowhead had an importance value of 4.76 in 1994, but dropped dramatically to 0.35 in 1995.

Populations of several plants, besides cattail and reed canary grass, increased after the fire. Smartweeds, common ragweed, swamp marigold, rice cutgrass (*Leersia oryzoides*), and sedges (*Carex* spp.) all showed greater importance values in 1995 sampling results. Whether or not all of these increases are directly attributable to the fire is highly debatable. Sedges appeared to fluctuate throughout the five-year sampling period and 1995's increase may have just been an upward correction from the unexplained decline in 1994. As previously discussed, smartweeds, common ragweed, and swamp marigold all also increased in the drought year of 1992. These plants appear to be disturbance-tolerant, capable of tolerating and exploiting disturbances such as drought and fire, where other species may not be able to survive or at least compete effectively. Smartweeds, common ragweed, and swamp marigold are all annuals which produce an abundance of seed, thereby allowing rapid colonization of newly exposed ground after disturbance.

Overall, plant species diversity rose greatly following the burn, from 34 species in 1994 to 53 in 1995. Species diversity often increases following a disturbance such as fire (Triner and Klimstra 1965, Ricklefs 1983). The increase in species diversity, unfortunately, was not due to an influx of native, hydrophytic species, as both the percent of species hydrophytic and the percent of species native to central Illinois decreased. Invasion by exotic, upland species accounted for these trends. Although species diversity increased after the burn, few, if any, other positive effects were noted. Two highly desirable species (water plantain and arrowhead) declined greatly, while cattail and reed canary grass both increased. Decreases in the percent of species hydrophytic and the percent of species native to central Illinois were other negatives possibly attributable to the fire. This one-time burn appears to have been more of a detriment to the wetland plant community than a benefit. Fire may not be an effective management tool in this marsh wetland community and no additional burns are planned at this time.

Vegetation Sampling Summary

Five years of vegetation sampling has, thus far, revealed a total of 95 plant species, representing 67 different genera (Appendix B). Hydrophytic species accounted for 68 percent of the total. Seventy-eight percent of species were native to central Illinois. Only a few of the sampled plants (*Vernonia* spp., *Asclepias incarnata,* and *Carex* spp.) may have originated, at least partially, from limited plantings and seedings in the wetland. One other sampled species, drooping coneflower (*Ratibida pinnata*), was originally introduced into the adjacent prairie and savanna restorations. Artificially established vegetation, however, is not thought to have significantly contributed to the composition of the wetland community. Community composition is attributable to natural processes of vegetation establishment, competition, and succession.

Summary and Conclusions

Although this wetland restoration is termed a success by its researchers, the evaluation of wetland restoration and creation success is often quite difficult. What exactly does determine success? Ideally, specific goals directly related to the proposed functions of the restored or created wetland should be formulated before the process actually begins (Army Corps 1993). This restoration project, unfortunately, was not designed with scientific experimentation and evaluation in mind. Evaluation of success is largely subjective in this case. Wetland success evaluation often includes monitoring selected aspects of vegetation, hydrology, and/or wildlife utilization (Erwin 1990). Monitoring of this restoration has mainly focused on the previously discussed vegetation sampling. This sampling revealed a rapidly-developing, diverse flora dominated by native, hydrophytic vegetation, similar to what might be found in a natural wetland community. Assuming no major hydrologic changes and no dramatic increases in potentially problematic vegetation, this functioning wetland community will likely persist into the future. This newly restored wetland is already being used by a wide variety of mammals, reptiles, amphibians, and birds, including two Illinois-endangered species, the northern harrier (*Circus cyaneus*) and the pied-billed grebe (*Podilymbus podiceps*) (Herkert 1992). In 1994, a pied-billed grebe even nested successfully in the wetland.

Continued sampling and monitoring over the next few years will clarify and more accurately determine any patterns of vegetational change in the wetland. Included in this determination will be the true, long-term effects of fire, if any, on vegetation composition. For example, will water plantain and arrowhead both recover after being severely affected by the fire? Recently installed staff gauges and ground water monitoring wells will permit accurate monitoring and documentation of hydrology, hopefully allowing the relationship between hydrology and vegetation composition to be successfully delineated. Continued vegetation sampling will be supplemented by detailed data analysis of vegetation composition, community change, and hydrologic/vegetational relationships. Further sampling and analysis of this restored wetland will provide restorationists and site managers with background information that can hopefully be of use in similar types of restoration efforts. Long-term monitoring of this developing wetland community will provide data not commonly available.

This restoration effort has successfully created a functioning kettle marsh wetland in an area of Illinois where wetlands have readily been drained and degraded. This quality restoration has realistically been achieved with relatively minor effort and cost. Proper site selection was key. This restoration site was a landscape depression with artificially drained, hydric soils, meaning that it was likely wetland in the past. Natural hydrology was easily restored by disrupting the underground drainage tiles through minimal excavation, unlike many other restorations or creations where extensive excavation is required. Also unlike many other projects, natural vegetation establishment processes were the primary source of the wetland plant community. Other restorations often invest much time and money in the planting and seeding of desired vegetation, often with less than optimal results. Relying principally on natural revegetation, this restored wetland has developed a diverse wetland community dominated by native, hydrophytic species in a very short time. As demonstrated, proper site selection can allow for a low cost approach to restoration, requiring only minimal labor and low maintenance. The broad-scale attractiveness of this type of restoration is obvious, especially when compared to more intensive approaches. Although small in scale, the contribution that this restored wetland makes to the natural resources of Illinois is nonetheless valuable. Similar restoration efforts can be repeated on a statewide or regional basis to significantly help offset the loss and degradation of wetland habitat.

Literature Cited

Baldassarre, G. A., and E. G. Bolen. 1994. Waterfowl ecology and management. John Wiley & Sons, New York, NY.

Bell, H. E., III. 1981. Illinois wetlands: their value and management. State of Illinois, Institute of Natural Resources, Chicago, IL. Document No. 81/33.

Beule, J. D. 1979. Control and management of cattails in southeastern Wisconsin wetlands. Wisc. Dept. Nat. Resour. Tech. Bull. 112, Madison, WI.

Cox, G. W. 1985. Laboratory manual of general ecology. 5th ed. Wm. C. Brown Publishers, Dubuque, IA.

Daubenmire, R. F. 1959. Canopy coverage method of vegetation analysis. Northw. Sci. 33:43-64.

Erwin, K. L. 1990. Wetland evaluation for restoration and creation. pp. 429-449, In J. A. Kusler and

M. E. Kentula (eds) Wetland Creation and Restoration - the Status of the Science. Island Press, Washington, DC.

Galatowitsch, S. M., and A. G. van der Valk. 1994. Restoring prairie wetlands: an ecological approach. Iowa State Univ. Press, Ames, IA.

Herkert, J. R. (ed). 1992. Endangered and threatened species of Illinois: status and distribution, Volume 2 - animals. Illinois Endangered Species Protection Board, Springfield, IL.

Hutchison, M. 1990. Vegetation management guideline - reed canary grass (*Phalaris arundinacea*). Vol. 1, No. 19. Illinois Nature Preserves Commission, Springfield, IL.

Illinois Department of Energy and Natural Resources and Nature of Illinois Foundation. 1994. The changing Illinois environment: critical trends. Summary report of the critical trends assessment project. Illinois Department of Energy and Natural Resources and Nature of Illinois Foundation, Springfield, IL.

Linde, A. F. 1969. Techniques for wetland management. Research Report 45, Dept. of Natural Resources, Madison, WI.

Martin, A. C., H. S. Zim, and A. L. Nelson. 1951. American wildlife and plants - a guide to wildlife food habitats. McGraw-Hill Book Company, New York, NY.

Mitsch, W. J. and J. G. Gosselink. 1986. Wetlands. Van Nostrand Reinhold, NY.

Odum, W. E. 1988. Predicting ecosystem development following creation and restoration of wetlands. pp. 67-70, In J. Zelazny and J. S. Feierabend (eds) Increasing Our Wetland Resources, National Wildlife Federation Conference Proceedings. Washington, DC.

Reed, P. B., Jr. 1988. National list of plant species that occur in wetlands: Illinois. U.S. Fish and Wildlife Service, National Wetlands Inventory, Washington, DC. NERC-88/18.13.

Reinartz, J. A. and E. L. Warne. 1993. Development of vegetation in small created wetlands in southeastern Wisconsin. Wetlands 13(3):153-164.

Ricklefs, R. E. 1983. The economy of nature. 2nd ed. Chiron Press, New York, NY.

Smith, L. M. and J. A. Kadlec. 1985. Fire and herbivory in a Great Salt Lake marsh. Ecology 66:259-265.

Suloway, L., and M. Hubbell. 1994. Wetland resources of Illinois: an analysis and atlas. Illinois Natural History Survey, Champaign, Illinois. Special Publication 15.

Taft, J., G. S. Wilhelm, D. Ladd, and L. A. Masters. Floristic quality assessment for Illinois. Erigenia. In preparation.

Triner, E. J. and W. D. Klimstra. 1965. Effects of burning and fallowing on vegetation. Tran. Ill. Acad. Sci. 58(2):102-114.

van der Valk, A. G. 1981. Succession in wetlands: a Gleasonian approach. Ecology 62(3):688-696.

Wilhelm, G. and L. Wetstein. 1993. Vegetational monitoring of a wetland restoration in northern Illinois. pp. 755-765, In M. C. Landin (ed) Wetlands - Proceedings of the 13th Annual Conference, Society of Wetland Scientists. June 1992, New Orleans, LA.

Willard, D. E. and A. K. Hiller. 1990. Wetland dynamics: considerations for restored and created wetlands. pp. 459-466 In J. A. Kusler and M. E. Kentula (eds) Wetland Creation and Restoration - the Status of the Science. Island Press, Washington, DC.

U.S. Army Corps of Engineers. 1993. Guidelines for developing mitigation proposals. Chicago District, U.S. Army Corps of Engineers, Chicago, IL.

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Appendix A. Top importance values for vegetation at the restored wetland, 1991-1995.

1991	1992	1993	1994	1995
Bidens tripartita	Bidens tripartita	Eleocharis palustris	Eleocharis palustris	Eleocharis palustris
(22.57)	(23.20)	(41.34)	(25.08)	(22.47)
Eleocharis palustris	Ambrosia artemisiifolia	Lemna minor	Lemna minor	Bidens aristosa
(18.06)	(22.46)	(11.73)	(17.58)	(13.73)
Rorippa islandica	Eleocharis palustris	Bidens tripartita	Bidens tripartita	Bidens tripartita
(10.75)	(17.83)	(8.91)	(15.33)	(11.11)
Alisma plantago-aquatica	Bidens aristosa	Alisma plantago-aquatica	Alisma plantago-aquatica	Lemna minor
(8.82)	(6.59)	(6.33)	(12.07)	(7.93)
Ambrosia artemisiifolia	Polygonum punctatum	Typha latifolia	Bidens aristosa	Ambrosia artemisiifolia
(4.73)	(5.86)	(4.50)	(6.03)	(5.56)
Echinochloa crus-galli	Polygonum pensylvanicum	Carex frankii	Sagittaria latifolia	Leersia oryzoides
(4.41)	(3.58)	(4.25)	(4.76)	(3.35)
Carex frankii	Carex tribuloides	Bidens aristosa	Typha latifolia	Typha latifolia
(3.55)	(2.24)	(3.28)	(2.63)	(3.21)
Lemna minor	Echinochloa crus-galli	Sagittaria latifolia	Amaranthus tuberculatus	Polygonum punctatum
(3.27)	(2.15)	(2.69)	(2.21)	(2.71)
Carex tribuloides	Rorippa islandica	Bromus inermis	Carex frankii	Eleocharis acicularis
(2.70)	(2.09)	(1.55)	(2.18)	(2.55)
Eleocharis obtusa	Carex frankii	Juncus acuminatus	Eleocharis acicularis	Phalaris arundinacea
(2.54)	(1.54)	(1.55)	(1.96)	(2.41)

Appendix B. Complete plant species list for the restored wetland, 1991-1995.

Scientific Name	Common Name	Hydrophytic[1]	Native to central Illinois[2]
Abutilon theophrastii	velvetleaf	No	No
Acalypha spp.	three-seeded mercury	--	Yes
Acalypha virginica	three-seeded mercury	No	Yes
Acer saccharinum	silver maple	Yes	Yes
Agropyron repens	quackgrass	No	No
Alisma plantago-aquatica	water plantain	Yes	Yes
Amaranthus rudis	tamarisk waterhemp	Yes	Yes
Amaranthus spp.	pigweed	--	--
Amaranthus tuberculatus	waterhemp	Yes	Yes
Ambrosia artemisiifolia	common ragweed	No	Yes
Asclepias incarnata	swamp milkweed	Yes	Yes
Aster pilosus	hairy aster	No	Yes
Aster simplex	panicled aster	Yes	Yes
Bidens aristosa	swamp marigold	Yes	Yes
Bidens frondosa	beggar-ticks	Yes	Yes
Bidens spp.	beggar-ticks	Yes	Yes
Bidens tripartita	beggar-ticks	Yes	Yes
Bromus inermis	awnless brome grass	No	No
Carex annectens	sedge	Yes	Yes
Carex frankii	sedge	Yes	Yes
Carex spp.	sedge	--	--
Carex tribuloides	sedge	Yes	Yes
Carex vulpinoidea	fox sedge	Yes	Yes
Carya ovata	scaly-bark hickory	No	Yes
Chamaesyce humistrata	milk spurge	No	Yes
Chamaesyce spp.	spurge	--	--
Chenopodium album	lamb's quarters	No	No
Cirsium arvense	Canada thistle	No	No
Cirsium spp.	thistle	--	--
Cirsium vulgare	bull thistle	No	No
Conyza canadensis	horseweed	No	Yes
Cornus spp.	dogwood	--	Yes
Crataegus mollis	red haw	Yes	Yes

Appendix B. Complete plant species list for the restored wetland, 1991-1995.

Scientific Name	Common Name	Hydrophytic[1]	Native to central Illinois[2]
Daucus carota	Queen-Anne's-lace	No	No
Dichanthelium spp.	panic grass	--	Yes
Echinochloa crus-galli	barnyard grass	Yes	No
Eclipta prostrata	yerba de tajo	Yes	Yes
Eleocharis acicularis	needle spikerush	Yes	Yes
Eleocharis erythropoda	spikerush	Yes	Yes
Eleocharis obtusa	spikerush	Yes	Yes
Eleocharis palustris	spikerush	Yes	Yes
Erechtites hieracifolia	fireweed	No	Yes
Erigeron annuus	annual fleabane	No	Yes
Erigeron spp.	fleabane	--	Yes
Eupatorium altissimum	tall boneset	No	Yes
Fragaria virginiana	wild strawberry	No	Yes
Galinsoga quadriradiata	Peruvian daisy	No	No
Geum canadense	white avens	Yes	Yes
Geum laciniatum	rough avens	Yes	Yes
Glechoma hederacea	ground ivy	No	No
Gleditsia triacanthos	honey locust	Yes	Yes
Impatiens capensis	jewelweed	Yes	Yes
Juncus acuminatus	rush	Yes	Yes
Juncus tenuis	path rush	Yes	Yes
Lactuca serriola	lettuce	Yes	No
Lactuca spp.	wild lettuce	--	--
Leersia oryzoides	rice cutgrass	Yes	Yes
Lemna minor	duckweed	Yes	Yes
Lindernia dubia	false pimpernel	Yes	Yes
Lycopus americanus	common water horehound	Yes	Yes
Lysimachia nummularia	moneywort	Yes	No
Oxalis stricta	yellow wood sorrel	No	Yes
Penthorum sedoides	ditch stonecrop	Yes	Yes
Phalaris arundinacea	reed canary grass	Yes	No
Phyla lanceolata	fog-fruit	Yes	Yes
Plantago lanceolata	buckhorn	Yes	No

Appendix B. Complete plant species list for the restored wetland, 1991-1995.

Scientific Name	Common Name	Hydrophytic[1]	Native to central Illinois[2]
Plantago rugelii	red-stalked plantain	Yes	Yes
Poa pratensis	Kentucky bluegrass	No	No
Polygonum amphibium	water smartweed	Yes	Yes
Polygonum hydropiper	smartweed	Yes	No
Polygonum hydropiperoides	wild water pepper	Yes	Yes
Polygonum lapathifolium	pale smartweed	Yes	Yes
Polygonum pensylvanicum	giant smartweed	Yes	Yes
Polygonum persicaria	spotted lady's thumb	Yes	No
Polygonum punctatum	dotted smartweed	Yes	Yes
Polygonum ramossissimum	bushy knotweed	No	Yes
Polygonum scandens	climbing buckwheat	Yes	Yes
Polygonum spp.	smartweed	--	--
Portulaca oleracea	purslane	No	No
Potamogeton nodosus	American pondweed	Yes	Yes
Ratibida pinnata	drooping coneflower	No	Yes
Rorippa islandica	marsh yellow cress	Yes	Yes
Rumex altissimus	pale dock	Yes	Yes
Rumex crispus	curly dock	Yes	No
Sagittaria graminea	grass-leaved arrowhead	Yes	Yes
Sagittaria latifolia	arrowhead	Yes	Yes
Salix exigua	sandbar willow	Yes	Yes
Scirpus acutus	hardstem bulrush	Yes	Yes
Scirpus tabernaemontanii	softstem bulrush	Yes	Yes
Scrophularia spp.	figwort	--	Yes
Setaria faberi	giant foxtail	No	No
Setaria glauca	yellow foxtail	Yes	No
Sida spinosa	prickly sida	No	No
Solanum carolinense	horse nettle	No	Yes
Toxicodendron radicans	poison ivy	Yes	Yes
Trifolium pratense	red clover	No	No
Trifolium repens	white clover	No	No
Typha angustifolia	narrowleaf cattail	Yes	No

Appendix B. Complete plant species list for the restored wetland, 1991-1995.

Scientific Name	Common Name	Hydrophytic[1]	Native to central Illinois[2]
Typha latifolia	cattail	Yes	Yes
Verbena hastata	blue vervain	Yes	Yes
Vernonia spp.	ironweed	--	Yes
Veronica spp.	speedwell	--	--
Viola pratincola	common blue violet	Yes	Yes
Viola spp.	violet	--	--
Vitis riparia	grape	Yes	Yes

[1] Reed (1988).
[2] Taft et al. (1993).

Restoration of Twelve Heavy-Seeded Hardwood Species in South Carolina Coastal Plain Wetlands

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D. D. Hook

Abstract: **The restoration of altered forested wetland ecosystems involves the determination of which species are best able to survive under different flood regimes, soils, and competition from forbs and grasses. In 1991 a study was established to measure the survival and growth of twelve, heavy-seeded hardwood tree species on poorly drained, somewhat poorly drained, moderately well drained, and well drained drainage classes, on three different geologic terraces across the South Carolina coastal plain. Species were selected for their wildlife and forest product values. On each terrace twelve blocks of species were planted in 7x7 rows and were replicated twice more in each drainage class. The interior 5x5 rows were measured, and the second and fourth rows contained tree shelters. Survival and growth were significantly affected by tree shelters and soil drainage class. The first three years of data indicate that tree shelters significantly increased the survival of all twelve species. Growth responses among seedlings of the same species were also significant, but less dramatic than survival. Drainage class was significant in species survival and growth by the third year, with generally more survival and growth found in the well drained and moderately well drained drainage classes. Additionally, tree shelters provided protection from wildlife browsing and protection from competing forbs and grasses. Finally, a suite of species for each drainage class was determined using methodology which incorporated best species performance by total height and percent survival.**

Introduction

It was estimated in the mid-1980s that 103.3 million acres of wetlands existed in the continental United States, which actually represents more than a 50 percent decrease in wetland acreage since settlement began (Dahl 1990). Wetlands are important to society because of the various functions and values which they provide (Walbridge 1993). Restoring altered forested wetland ecosystems requires selecting those species which appear to be best adapted to survive specific flood regimes, soils, and competition from forbs and grasses.

Although a considerable amount is known about the flood tolerance of many tree species (Teskey and Hinckley 1977, Hook 1984a, Kozlowski 1984, Tiner 1991), specific information on species-site suitability is lacking when they are planted on wetland sites where hydrologic regimes and soils have been altered (Hook 1984b). Thus, there is a real need to develop better guidelines for matching species and site traits to improve restoration efforts.

Information on the effects of tree shelters on survival and growth of tree species is scant and virtually nothing is known about their effects on wetland sites (Manchester et al. 1988, Potter 1989, Tuley 1983 and 1985, Minter et al. 1992, Bardon and Countryman 1992, Lantagne et al. 1990, Ponder 1992, McConnel 1992).

The effects of soil drainage class and tree shelters on the survival and growth of twelve heavy-seeded hardwood species in the South Carolina coastal plain was the focus of this study. It was hypothesized that species would not differ, with soil drainage class or tree shelters, in percent survival and height growth.

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Materials and Methods

Sites

This project was carried out on three separate geologic terraces across the South Carolina coastal plain. The study locations were the Savannah River Site (SRS; elevation > 30 m mean sea level [MSL]) in New Ellenton, SC; the Santee Experimental Forest (SEF; < 15 m MSL) in Cordesville, SC; and the Webb Wildlife Center (WWC; between 15-30 m MSL) located in Garnett, SC. Two locations (SRS and WWC) contained four different drainage classes (well drained (WD), moderately well drained (MWD), somewhat poorly drained (SPD), and poorly drained (PD) soil while SEF contained only three soil drainage classes (there was no well drained drainage class). As implied, drainage classes were chosen based on their relative wetness. Differences in these classes were based on several factors including hydrology and soil characteristics, including soil aeration and depth to mottling. Depth to mottling on the well drained class was greater than or equal to 30 inches; 16 to 29 inches on the moderately well drained class; 9 to 15 inches on the somewhat poorly drained class; and 3 to 6 inches on the poorly drained site. The soil series for each terrace and drainage class are listed below.

Santee Experimental Forest - Flatwoods Coastal Plain, Upper Terrace

- Moderately Well Drained - Duplin
- Somewhat Poorly Drained - Wahee
- Poorly Drained - Bethera

Savannah River Site - Hilly Coastal Plain

- Well Drained - Lucy and Troup
- Moderately Well Drained - Hornsville
- Somewhat Poorly Drained - Smithboro
- Poorly Drained - Rembert

Webb Wildlife Center - Flatwoods Coastal Plain, Lower Terrace

- Well Drained - Emporia
- Moderately Well Drained - Emporia
- Somewhat Poorly Drained - Elonia
- Poorly Drained - Elonia

Experimental Design

Each drainage class contained three replications of twelve species planted in 7x7 blocks. Only the inner 5x5 rows were measured with the second and the fourth rows containing tree shelters; for a total of 25 trees per block, with 10 trees containing tree shelters. The study was set up as a split-split plot design. For analysis purposes only seedlings 1-20 (rows 1-4) were used to balance the number of sheltered versus non-sheltered seedlings. The species involved in the study were: water hickory (*Carya aquatica* (Michx. f.) Nuft.), swamp tupelo (*Nyssa sylvatica* var. *biflora* (Walt.) Sarg.), swamp chestnut oak (*Quercus michauxii* Nuft.), mockernut hickory (*Carya tomentosa* (Poir.) Nuft.), white oak (*Quercus alba* L.), southern red oak (*Quercus falcata* Michx.), overcup oak (*Quercus lyrata* Walt.), cherrybark oak (*Quercus pagoda* Ell.), blackgum (*Nyssa sylvatica* Marsh.), persimmon (*Diospyros virginiana* L.), sugarberry (*Celtis laevigata* Willd.), and dogwood (*Cornus florida* L.). Species were chosen based on their relative abundance within these areas, their wildlife value and their forest products value. For ease in reading the graphs, Table 1 lists the species with their species code along with their relative flood tolerance (McKnight et al. 1980).

Measurements

All trees were measured for survival and total height. Total height was obtained by using a meter stick placed beside the individual. Visual observation determined whether the tree was alive or dead. Third year height growth and diameter were also measured, but the results are not presented herein. Additionally, wildlife browsing and dieback were noted for each individual.

Species Suites

The methodology behind the determination of the suite of species best suited for a particular drainage class was based on four categories: non-sheltered percent survival, sheltered percent survival, non-sheltered total height and sheltered total height. The cutoff point for a species falling into the percent survival category was at least 45 percent survival. The cutoff point for a species falling into the total height category was at least 40 cm in height. These cutoffs were chosen based on relative performance of individual species within each class. For every terrace and drainage class, in each of the four categories, a

Table 1. Species Codes and Relative Flood Tolerance of the Twelve Study Species (McKnight et al. 1980).

Species	*Species Code*	*Flood Tolerance*
Blackgum	BG	Moderately Tolerant
Cherrybark Oak	CO	Weakly Tolerant to Intolerant
Dogwood	D	Intolerant
Hackberry	H	Moderately Tolerant to Intolerant
Mockernut Hickory	MH	Intolerant
Overcup Oak	OC	Moderately Tolerant
Persimmon	P	Moderately Tolerant
Swamp Chestnut Oak	SC	Weakly Tolerant
Southern Red Oak	SR	Intolerant
Swamp Tupelo	ST	Tolerant
Water Hickory	WH	Moderately Tolerant
White Oak	WO	Moderately Tolerant to Intolerant

Table 2. Non-Sheltered Percent Survival Species Performance Across All Terraces and Drainage Classes.

Terrace	*WD*	*MWD*	*SPD*	*PD*
SRS	bg,co,oc,p,sc,sr,wo	co,oc,sc,wo	bg,co,oc,sc,sr,wo	oc,st
WWC	co,d,h,oc,p,sc,sr,wo	oc	co,oc,sc	co
SEF	NA	bg,co,oc,p,sc,sr,wo	bg,co,oc,p,sc,st,wh	oc
Overall	co,oc,p,sc,sr,wo	co,oc,sc,wo	bg,co,oc,sc	oc

Table 3. Sheltered Percent Survival Species Performance Across All Terraces and Drainage Classes.

Terrace	*WD*	*MWD*	*SPD*	*PD*
SRS	bg,co,d,mh,oc,p,sc, sr,wh,wo	bg,co,oc,p,sc,sr, st,wo	bg,co,h,oc,p,sc,sr,st, wo	bg,co,oc,p,sc,st
WWC	bg,co,d,h,mh,oc,p, sc,sr,wh,wo	bg,co,oc,p,sc,sr,wo	bg,co,oc,p,sc,srwo	bg,co,oc,p,sc, wo
SEF	NA	bg,co,oc,p,sc,sr, st,wo	bg,co,mh,oc,p,sc,st, wh,wo	bg,co,oc,p,sc, stwh
Overall	bg,co,d,mh,oc,p,sc, sr,wh,wo	bg,co,oc,p,sc,sr, stwo	bg,co,oc,p,sc,st,wo	bg,co,oc,p,sc,st

suite of species was chosen based on their ability to meet the minimum cutoff requirements. Once a suite was determined for every drainage class in each terrace, the results were compared and any overlapping species were put into an overall group which consisted of one particular drainage class on all three sites. This process was repeated for each of the four drainage classes. The final step was to compare the overall groups and check for frequency of species overlap among the four drainage classes. If a species was in either three or four of the groups it was included in the final suite of species. In some cases, a species was included if it fell into two of the groups if the groups were not in the same category (i.e., if a species turned up in both sheltered and non-sheltered percent survival but not in any of the total height classes it would not be a valid choice for the final suite determination). The result was a performance index which gauged the species most appropriate for a drainage class within the coastal plain of South Carolina.

Statistical Methodology

Analysis of Variance (ANOVA) was used to compare the means of the variables being considered. All differences in sample means were judged statistically significant (or not) by comparing them to the variation within samples. Additionally, Duncan's Multiple Range Test ($\alpha = .05$) was used to rank and compare the sample means.

Results

Savannah River Site (SRS)

Percent survival of non-sheltered trees at SRS varied greatly by drainage class. Overcup oak and swamp tupelo had 50 percent or higher survival rates in the poorly drained class. White oak, southern red oak, swamp chestnut oak, overcup oak, cherrybark oak, and blackgum had 50 percent or higher survival rates in the somewhat poorly drained class. White oak, swamp chestnut oak, cherrybark oak, and overcup oak had 50 percent or higher survival rates on the moderately well drained class while white oak, southern red oak, swamp chestnut oak, persimmon, overcup oak, cherrybark oak, and blackgum displayed the highest survival rates in the well drained class. The sheltered trees at SRS generally averaged much higher survival rates. Swamp tupelo, overcup oak, swamp chestnut oak and blackgum all had survival rates of at least 60 percent in the poorly drained class with swamp tupelo displaying 100 percent rate of survival. In the somewhat poorly drained class, white oak, swamp tupelo, southern red oak, swamp chestnut oak, persimmon, overcup oak, cherrybark oak and blackgum all had survival rates of at least 60 percent. White oak, southern red oak, swamp chestnut oak, overcup oak, cherrybark oak and blackgum had survival rates of at least 45 percent in the moderately well drained class while every species except swamp tupelo and hackberry had at least 70 percent survival rates in the well drained class. The overall percent survival means in the well drained and somewhat poorly drained classes were not significantly different from each other but were significantly different from the moderately well drained and poorly drained classes. The sheltered trees in every drainage class had significantly greater survival rates than did the non-sheltered trees.

Total height for the trees without shelters varied greatly by drainage class. White oak, swamp chestnut oak, overcup oak and blackgum all attained heights of 90 cm or better in the poorly drained class while southern red oak, overcup oak and cherrybark oak were at least 60 cm tall at the end of three years growth in the somewhat poorly drained class. In the moderately well drained class, white oak, swamp tupelo and overcup oak were at least 70 cm in height while swamp chestnut oak, hackberry, dogwood and blackgum were at least 45 cm in height in the well drained class. Generally, greater heights were found in the poorly drained class among all twelve species. For the sheltered trees in the poorly drained class, swamp tupelo, overcup oak and blackgum grew to heights of at least 140 cm. Only blackgum and cherrybark oak attained heights of at least 95 cm in the somewhat poorly drained class while white oak, overcup oak, cherrybark oak and blackgum grew to at least 110 cm in the moderately well drained class. Only persimmon reached a height greater than 90 cm in the well drained class. Again, the greatest total height was generally found in the poorly drained class. The poorly drained class produced significantly taller trees than the other drainage classes. There was no significant difference between the total height means on the moderately well drained and somewhat poorly drained classes, but the moderately well drained class produced significantly taller trees than the well drained class. In all drainage classes, sheltered trees were significantly taller than the non-sheltered trees.

Webb Wildlife Center (WWC)

At WWC only one species, cherrybark oak, displayed a survival rate greater than 40 percent in the poorly drained class for the non-sheltered species. Swamp chestnut oak, overcup oak and cherrybark oak all had survival rates greater than 50 percent in the somewhat poorly drained class, while only overcup oak had a survival rate greater than 50 percent in the moderately well drained class. In the well drained class every species except water hickory, swamp tupelo, mockernut hickory and blackgum had a greater than 50 percent survival rate. For the sheltered trees, white oak, swamp chestnut oak, persimmon, overcup oak, cherrybark oak and blackgum all had survival rates greater than 50 percent in the poorly drained class. In the somewhat poorly drained class white oak, southern red oak, swamp chestnut oak, persimmon, overcup oak and cherrybark oak all had greater than 50 percent survival rates while white oak, southern red oak, swamp chestnut oak, persimmon, overcup oak, cherrybark oak and blackgum displayed at least 60 percent survival rates in the moderately well drained class. Every species with the exception of swamp tupelo had 60 percent or greater survival rates in the well drained class. The well drained class percent survival mean was significantly larger than the means in any of the other three drainage classes. In every drainage class, the sheltered trees displayed significantly greater percent survival than the non-sheltered trees.

The non-sheltered trees displayed much variation in total height. In the poorly drained class, only overcup oak displayed extreme total height (over 130 cm). Swamp chestnut oak and overcup oak were the only species which were taller than 70 cm in the somewhat poorly drained class while swamp chestnut oak, overcup oak and cherrybark oak were the tallest in the moderately well drained class. White oak, swamp chestnut oak, overcup oak and cherrybark oak each were at least 80 cm in height in the well drained class. For the sheltered trees the total height was, on the average, much greater. Using a baseline of 100 cm in height, swamp tupelo, swamp chestnut oak, overcup oak, dogwood, cherrybark oak and blackgum attained heights of at least 100 cm in the poorly drained class. Cherrybark oak, overcup oak, swamp chestnut oak, swamp tupelo and white oak were at least 110 cm in height in the somewhat poorly drained class while white oak, swamp chestnut oak, overcup oak, cherrybark oak and blackgum were at least 100 cm in the moderately well drained class. The greatest height was measured in the well drained class with white oak, southern red oak, swamp chestnut oak, overcup oak, dogwood and cherrybark oak attaining heights of at least 150 cm. Overcup oak and swamp chestnut oak each averaged over 200 cm in total height in this same class. The well drained class produced significantly taller trees than the other drainage classes. The sheltered trees in all drainage classes were significantly taller than the non-sheltered trees.

Santee Experimental Forest (SEF)

At SEF, non-sheltered trees had varying survival rates among drainage classes. Only overcup oak had a greater than 50 percent survival rate in the poorly drained class. Water hickory, swamp tupelo, swamp chestnut oak, persimmon, overcup oak, cherrybark oak and blackgum had at least 50 percent survival in the somewhat poorly drained class while white oak, swamp chestnut oak, overcup oak, cherrybark oak and blackgum had at least 50 percent survival in the moderately well drained class. For the sheltered trees, water hickory, swamp tupelo, swamp chestnut oak, persimmon, overcup oak, cherrybark oak and blackgum had 50 percent or greater survival in the poorly drained class while every species except southern red oak, mockernut hickory, hackberry, and dogwood had greater than 50 percent survival in the somewhat poorly drained class. With the exception of dogwood, hackberry, mockernut hickory and water hickory, all species in the moderately well drained class had at least 50 percent survival rates. Generally, blackgum, cherrybark oak, overcup oak, persimmon, and swamp chestnut oak did extremely well in survival across all drainage classes. There were no significant differences among the overall percent survival rates in any of the three drainage classes. However, in every drainage class, sheltered trees exhibited significantly greater survival rates than did the non-sheltered trees.

The greatest total height generally occurred in the driest drainage class, and in general the heights attained at SEF were lower as compared to the other two terraces. Swamp tupelo, persimmon, overcup oak, cherrybark oak, and blackgum all attained heights of at least 40 cm in the poorly drained class. Swamp tupelo, southern red oak, swamp chestnut oak, persimmon, overcup oak, and cherrybark oak were at least 60 cm in total height in the moderately well drained class. The sheltered trees in the poorly drained class were generally shorter than the sheltered trees in the other two drainage classes with

blackgum, overcup oak and swamp tupelo attaining heights of at least 70 cm. In the somewhat poorly drained class, swamp tupelo, cherrybark oak and blackgum were at least 80 cm in height while white oak, swamp chestnut oak, overcup oak, cherrybark oak and blackgum grew to at least 120 cm in height. The moderately well drained class produced total heights which were significantly greater than the total heights found on the somewhat poorly drained and poorly drained classes. Again, in all drainage classes sheltered trees were significantly taller than the non-sheltered trees.

Species Suites

The results of the performance index used to segregate the species most appropriate for a particular drainage class within the South Carolina coastal plain are seen in Tables 2-6. These tables show the progression for the final suite determination in each drainage class. Table 2 shows the non-sheltered trees which met the 45 percent survival rate cutoff point in each terrace for every drainage class. Table 3 is the same as Table 2 except that the trees are sheltered. Table 4 shows the nonsheltered trees which met the 40 cm cutoff point for minimum height for every terrace and every drainage class. Table 5 shows the same data for sheltered trees. Table 6 shows the species which performed best in each of the four classifications (nonsheltered percent survival, sheltered percent survival, non-sheltered total height, and sheltered total height) in each drainage class for each terrace. Table 7 shows the species best suited for each particular drainage class within the coastal plain of South Carolina.

The above performance index indicates that cherrybark oak, overcup oak, persimmon, swamp chestnut oak, southern red oak and white oak will perform well in the well drained class. Cherrybark oak, overcup oak, swamp chestnut oak, white oak, persimmon, southern red oak, and swamp tupelo are the desired species for survival and growth in the moderately well drained class while blackgum, cherrybark oak, overcup oak, swamp chestnut oak, persimmon, and swamp tupelo will perform well in the somewhat poorly drained class. Finally, cherrybark oak, overcup oak, blackgum, persimmon, swamp chestnut oak, and swamp tupelo are the most likely species to exhibit the greatest survival and growth in the poorly drained class.

Cherrybark oak, overcup oak, persimmon, and swamp chestnut oak are all present in each drainage class suite and swamp tupelo is present in every drainage class except for the well drained class. Swamp tupelo is the only species in the study which is truly tolerant to flooding; thus, it is not unusual that it performed as well as it did in the wetter drainage classes. The unusual result is that it ended up in the final suite of species in the moderately well drained class. The fact that it was planted and not naturally regenerated partially explains this since flooded conditions are necessary for successful germination of this species. Both overcup oak and persimmon are moderately tolerant to flooding and both were in the final suite of species for the somewhat poorly drained and poorly drained drainage classes. Likewise, they were also in the moderately well drained and well drained drainage class suites. In contrast, cherrybark oak is weakly tolerant to intolerant of flooding, but it did well not only on the well drained and moderately well drained soils but also on the somewhat poorly drained and poorly drained soils. Williams et al. (1992) reported in a study on water oak *(Quercus nigra* L.), nuttall oak *(Quercus nuttallii* L.) and cherrybark oak that first year survival and growth performance of cherrybark oak were poor on the hydric soils. This contradicts the findings in this study. Likewise, swamp chestnut oak is weakly tolerant to flooding but it showed excellent survival and growth on the poorly drained soils. It is not unusual to find swamp chestnut oak on the fringes of swamps or in bottoms, but for it to behave as it did in a flooded situation is unusual. Blackgum responded as expected. It is a moderately flood tolerant species and it fell into the final suite of species in both the somewhat poorly drained and poorly drained drainage classes.

Several species including dogwood, mockernut hickory, and water hickory did poorly on the wetter classes on all terraces. Both dogwood and mockernut hickory are intolerant to flooding and consequently did not fare well in the wetter drainage classes. In the drier soils these two species did fairly well in terms of survival and total height only when a shelter was present. The best survival and growth of water hickory was found in the well drained class across all terraces.

Discussion

Although the three terraces involved in the study cannot be statistically compared, some differences

Table 4. Non-Sheltered Total Height Species Performance Across All Terraces and Drainage Classes.

Terrace	*WD*	*MWD*	*SPD*	*PD*
SRS	bg,co,d,h,p,sc,sr, wo	bg,co,oc,p,sc,sr,st, wo	bg,co,h,oc,p,sc,sr, st,wo	bg,co,h,oc,p,sc,sr,st, wo
WWC	bg,co,d,h,oc,p,sc, sr,st,wh,so	bg,co,mh,oc,p,sc,sr, wh,wo	co,h,oc,p,sc,sr, st,wo	bg,co,mh,oc,p,sc,w o
SEF	NA	bg,co,oc,sc,sr,st,wh, wo	co,h,oc,p,sc,sr, st,wo	bg,co,oc,p,st
Overall	bg,co,d,h,oc,p,sc, sr,wo	bg,co,oc,p,sc,sr,st, wh,wo	co,h,oc,p,sc,sr,st, wo	bg,co,oc,p,sc,st, wo

Table 5. Sheltered Total Height Species Performance Across All Terraces and Drainage Classes.

Terrace	*WD*	*MWD*	*SPD*	*PD*
SRS	bg,co,d,h,oc,p,sc, sr,st,wh,wo	bg,co,d,oc,p,sc,sr,st, wh,wo	bg,co,h,oc,p,sc,sr, st,wh,wo	bg,co,h,oc,p,sc,sr,st, wh,wo
WWC	bg,co,d,h,mh,oc,p, sc,sr,st,wh,wo	bg,co,d,h,mh,oc,p, sc,sr,wh,wo	bg,co,d,h,mh,oc,p, sc,sr,wh,wo	bg,co,d,h,mh,oc,p,sc, sr,st,wh,wo
SEF	NA	bg,co,d,h,mh,oc,p, sc,sr,st,wh,wo	bg,co,d,mh,oc,p, sc,sr,st,wh	bg,co,mh,oc,p,sc,sr, st,wo
Overall	bg,co,d,h,oc,p,sc, sr,st,wh,wo	bg,co,d,h,mh,oc,p, sc,sr,st,wh,wo	bg,co,d,h,mh,oc,p, sc,sr,st,wh	bg,co,oc,p,sc,sr,st, wh,wo

Table 6. Best Species Performance in Each Drainage Class from the Four Classification Categories.

Class	*WD*	*MWD*	*SPD*	*PD*
Non-Sheltered % Survival	co,oc,p,sc,sr,wo	co,oc,sc,wo	bg,co,oc,sc	oc
Sheltered % Survival	bg,co,d,mh,oc, p,sc,sr,wh,wo	bg,co,oc,p,sc,sr,st, wo	bg,co,oc,p,sc,st, wo	bg,co,oc,p,sc,st
Non-Sheltered Total Height	bg,co,d,h,oc,p, sc,sr,wo	bg,co,oc,p,sc,sr,st, wh,wo	co,h,oc,p,sc,sr, st,wo	bg,co,oc,p,sc,st,wo
Sheltered Total Height	bg,co,d,h,oc,p, sc,sr,st,wh,wo	bg,co,d,h,mh,oc, p,sc,sr,st,wh,wo	bg,co,d,h,mh, oc,p,sc,sr,st,wh	bg,co,oc,p,sc, sr,st,wh,wo

Table 7. Suite of Species Exhibiting the Best Performance Across All Study Drainage Classes in the South Carolina Coastal Plain.

Class	*WD*	*MWD*	*SPD*	*PD*
Overall	co,oc,p,sc,sr,wo	co,oc,sc,wo,p,sr,st	bg,co,oc,sc,p,st	co,oc,bg,p,sc,st

among them are apparent. These differences are important in explaining the varying performances among the species within each site. The seedlings experienced rainfall extremes and constant temperatures during the three growing seasons. The first growing season averaged 71 degrees Fahrenheit during the eight month period and had 39 inches of rainfall. The second growing season averaged 73 degrees Fahrenheit and had 35 inches of rainfall. And finally the third growing season averaged 73 degrees Fahrenheit and had an abundance of rainfall (52 inches).

At WWC and SEF, the general trend was that the trees attaining the greatest heights were found in the better drained classes. At SRS, the general trend was the opposite, with the poorly drained class displaying the greatest total height and incremental height growth while the well drained class had the lowest total height and incremental height growth. These results are consistent with what is known about flooding and plant growth. There is extensive documentation to support the idea that flooding is generally injurious to a plant's ability to grow. Kozlowski (1984) also indicated that flooding adversely impacts shoot growth by inhibiting internode elongation of numerous species including *Alnus rugosa, Platanus occidentalis, Betula nigra, Ulmus americana, Ulmus* alata and Acer *rubrum.* Further he found that 39 days of flooding decreased the height growth of *Ulmus americana* seedlings, with unflooded plants showing a rate of growth three to five times greater than the growth rate of flooded plants.

The hydrology of each terrace in this study is an important factor to consider. Dramatic differences in hydrologic regimes can be seen among these terraces. At WWC and SEF, the flood waters tended to be stagnant. They lacked topography and slope; hence, drainage was limited. At SRS, however, water had a constant opportunity to flow through the poorly drained site. There was a definite slope from the mostly well-drained to poorly-drained site. Also, the poorly-drained site was drained by an actively flowing stream. Kozlowski (1984) found that standing water is much more injurious to trees than moving water. Hunt (1951) found that both height and root growth were reduced more in stagnant water than in moving water. In a study on swamp and water tupelo, Hook et al. (1970) found that the height growth in moving water was about twice the height growth found in stagnant water treatments. WWC and SEF both had poorly drained soil classes in which the water was stagnant while SRS had moving water in its poorly drained class. The height differences (greater average heights at SRS than at the other two terraces) of species such as blackgum (moderately flood tolerant) and swamp tupelo (flood tolerant) among these poorly drained classes was apparent and the hydrology seems to be one viable explanation.

There was also a difference in history of site use. Sites at SEF and WWC had been cleared and treated for several years prior to planting the seedlings. In contrast, the SRS was logged in the fall and planted the following spring. Also overcup oak (moderately flood tolerant) and swamp chestnut oak (weakly flood tolerant) attained much greater heights at SRS than at SEF, but attained similar heights at WWC. Each of the three terraces contain drainage classes with varying degrees of slopes which effect water movement in the poor drainage classes among the terraces. At SEF and WWC the terrain on all drainage classes was generally flat, void of any significant slopes. SRS, however, has three of its drainage classes located on a fairly significant slope, with the poorly drained class located at the bottom of the slope. This slope affected both the surface and subsurface water movement on this terrace, by increasing the movement of both.

Summary and Conclusions

1. Survival of all species at SEF and WWC was highest on the well drained and moderately well drained drainage classes. At SRS, the highest survival rates were found on the well drained and somewhat poorly drained classes.
2. On all terraces, the greatest total heights of most species were found in the well drained and moderately well drained drainage classes.
3. Among the poorly drained classes across all terraces, SRS displayed greater total heights of most species. This is probably due to the effect of running water vs stagnant water at the site.
4. Drainage class had a significant effect on tree diameters, with generally larger diameters found in the wetter drainage classes.
5. Survival rate of all species on all terraces increased significantly with tree shelters.
6. Total height and height growth increment of all species was significantly larger with tree shelters.

Although there was overlap in species performance (both survival and height) across the soil drainage classes, patterns of species suitability were evident.

Species know to be upland species did better on the WD and MWD sites. Also, the most flood-tolerant species (OC and ST) did best on the PD sites. Except for extreme site conditions, it appears that it takes longer than 3 years for site traits to truly weed out those species that are least suited to the site. Use of tree shelters appeared to increase the ecological tolerance of most species on dry and wet sites. The long-term implications of these responses need further observation and documentation.

Literature Cited

Bardon, R. E. and D. W. Countryman. 1992. First year survival and growth of underplanted northern red oak seedlings in southcentral Iowa. Fifth Workshop on Seedling Physiology and Growth Problems in Oak Plantings (Abstracts). USDA Forest Service, General Technical Report NC-158.

Dahl, T. E. 1990. Wetland losses in the United States, 1780s to 1980s. U.S. Dept. of Interior, Fish and Wildlife Service, Washington, DC. 21 pp.

Hook, D. D., O. G. Langdon, J. Stubbs, and C. L. Brown. 1970. Effect of water regimes on the survival, growth, and morphology of tupelo seedings. Forest Science 16:304-311.

Hook, D. D. 1984a. Adaptations to flooding with fresh water. pp. 265-294 In Flooding and Plant Growth. Ed. T. T. Kozlowski. Academic Press, NY.

Hook, D. D. 1984b. Waterlogging tolerance of lowland tree species of the South. Southern Journal of Applied Forestry 8(3):136-149.

Hunt, F. M. 1951. Effects of flooded soil on growth of pine seedlings. Plant Physiology 25:363-368.

Kozlowski, T. T. 1984. Flooding and plant growth. Press, NY. 356 pp.

Lantagne, D. O., C. W. Ramm, and D. I. Dickmann. 1990. Tree shelters increase heights of planted oaks in a Michigan clearcut. Northern Journal of Applied Forestry 7:24-26.

Manchester, E. H., F. G. Roland, and D. H. Sims. 1988. Tree shelters show promise for oak regeneration. USDA Forest Service, Southern Region, Management Bulletin R8-MB25.

McConnel, J. L. 1992. The use of tree shelters and underplantings for oak regeneration. Oak Regeneration Symposium Proceedings, Sept. 1992. Southern Region, USDA Forest Service, Atlanta, GA.

McKnight, J. S., D. D. Hook, O. G. Langdon, and R. L. Johnson. 1980. Flood tolerance and related characteristics of trees of the bottomland forests of the southern United States. Reprinted from Wetlands of Bottomland Hardwood Forests. Proceedings of a Workshop on Bottomland Hardwood Forest Wetlands of the Southeast. USDA Forest Service, Proceedings Reprint, pp. 29-69.

Minter, W. F., R. K. Myers, and B. Fischer. 1992. Effects of tree shelters on northern red oak seedlings planted in harvested forest openings. Northern Journal of Applied Forestry 9:58-63.

Ponder, F., Jr. 1992. Microclimate and growth of oak seedlings grown in treeshelters inside and outside of a shadehouse. Fifth Workshop on Seedling Physiology and Growth Problems in Oak Plantings (Abstracts). USDA Forest Service, General Technical Report NC-158.

Potter, M. J. 1989. Treeshelters: Their effects on microclimate and tree establishment. Paper presented at International Conference on Fast Growing and N-Fixing Trees. Phillips University, Marburg, FRG.

Teskey, R. D. and T. M. Hinckley. 1977. Impact of water level changes on woody riparian and wetland communities. Vol. II. Southern Forest Region. U.S. Fish and Wildl. Serv., Biological Serv. Program, Washington, DC. FWS/OBS-77/59. 46 p.

Tiner, R. G. 1991. The concept of a hydrophyte for wetland identification: Individual plants adapt to wet environments. BioScience 41(4):236-247.

Tuley, G. 1983. Shelters improve the growth of young trees in the forest. Quarterly Journal of Forestry 77:77-87.

Tuley, G. 1985. The growth of young oak trees in shelters. Journal of Forestry 58(2):181-195.

Walbridge, M. R. 1993. Functions and values of forested wetlands in the southern United States. Journal of Forestry 92:15-19.

Williams, H. M., B. A. Kleiss, M. N. Humphrey, and C. V. Klimas. 1992. First-year field performance of oak species with varying flood tolerance planted on hydric and non-hydric soils. Paper presented at the Seventh Biennial Southern Silvicultural Research Conference, Mobile, AL. pp. 409-414.

Fish Community Changes in the Buffalo River, Buffalo, New York, Following Water Quality Improvements

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Abstract: **Changes in the fish community of the Buffalo River, Buffalo, New York, were assessed using three indices; the Shannon-Weiner diversity index (H'), percentage similarity index (PSC), and an index of biotic integrity (IBI). Published fish catch data from four reaches in the lower 9.7 km of the river for the years 1981 and 1992 were used. An IBI was developed based on indicator fishes particular to the geographic area investigated. Index-specific analyses among survey sites show that diversity (H') generally increased, as did biotic integrity (IBI scores) indicating a moderate degree of recovery in the fish community over the period evaluated.**

Introduction

Prior to the early 1800's, the lower Buffalo River provided plentiful habitat for reproduction, feeding and growth of a myriad of resident and migratory fish, birds, and wildlife (Poole et al. 1994). Since the mid-1800's, engineering (e.g., dredging, filling, locks, canals) and compensating works (e.g., bulkheads and berms) have been constructed to improve navigation for upstream industries. Cumulatively, these activities have resulted in changes of channel substrate, flow regime, water quality, and riparian soils and vegetation, thereby degrading aquatic habitat. By the early 1900's, municipal and industrial waste discharges to the Buffalo River degraded water quality such that the lower river became essentially devoid of any fish life (Greeley 1929).

In the late 1960's, water quality began to improve as a result of increased sewage treatment, pollution abatement programs, and flow augmentation by industrial cooling water pumped from Lake Erie and discharged into the river (Sweeney and Merckel 1972, Sauer 1979). By 1972, fish started to re-inhabit the lower river (Sweeney 1972). Plant closures and process shutdowns associated with economic changes eliminated still more pollution sources through the 1980's; however, municipal and industrial waste assimilation continues to be an important use of the lower river (Bode et al. 1993).

Though water quality has improved over the past twenty years and fish again inhabit the lower river, this section of the river still does not meet its designated best use (Class C *fishing and fish propogation);* it is considered by the New York State Department of Environmental Conservation (DEC) to be "impaired" (NYSDEC 1993a). Impairment is largely due to contaminated sediments, groundwater leaching from inactive hazardous waste sites (NYSDEC 1993a), combined sewer overflows, and nonpoint sources (Irvine and Pettibone 1996). Use impairments directly related to fish are: restrictions on consumption; tainting; degradation of populations; tumors and other deformities; and, loss of habitat (NYSDEC 1989). The fish community, therefore, should be useful in terms of assessing progress toward restoring the aquatic ecosystem of the Buffalo River.

Although water quality improvements have been documented in the Buffalo River, there have been no assessments of improvements in the fish community. Six fishery surveys of the river conducted between 1928 and 1975 were limited to annotated species lists (NYSDEC 1993b). Although these studies did a great deal to describe patterns of distribution and relative abundances of fishes throughout the drainage basin, detailed analyses of fish community structure were beyond the scope of the research.

Structurally and functionally diverse fish communities can provide evidence of water quality in that they are particularly responsive to environmental stress (Hocutt 1981, Fausch et al. 1984). Response to

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chemical stress often involves rapid changes in species composition that can translate into changes in various aspects of community structure. Direct and indirect responses can include: increased frequency of disease; disrupted reproductive processes; reduced diversity; reduced size (biomass); retrogressed species composition (i.e., opportunistic/tolerant species); or, debased food prey items (Ford 1989, Gray 1989).

The objectives of our study were to:

- Document long-term changes in Buffalo River fish communities;
- Develop and apply an index of biotic integrity (IBI) for the river; and,
- Evaluate the degree to which the Shannon-Weiner diversity index (H'), percentage similarity (PSC), and an index of biotic integrity (IBI) correspond relative to one another.

We used H', PSC, and an IBI because, when used together, they may permit more reliable assessment of natural groupings than any one method alone. We believe this approach could be especially advantageous in the analysis of disturbed ecosystems and observed changes in biological structure in response to stress or rehabilitation.

Study Area

The Buffalo River drainage basin is located in western New York and extends nearly 65 km along a north-northwest axis from the Allegheny foothills to eastern Lake Erie (Figure 1). The 1,155 km^2 drainage basin is roughly crescent-shaped and lies in two major physiographic provinces, the Allegheny Plateau and Erie-Ontario Lake Plain. The drainage basin is divided into three tributary sub-basins (Cayuga, Buffalo, and Cazenovia creeks) and has a dendritic or branching pattern. The Buffalo River is formed by the junction of the Cayuga and Buffalo creeks, 13 km upstream of Lake Erie; Cazenovia Creek enters the river 9.7 km upstream of the lake (Figure 2). Mean annual discharge (total) is about 17 m^3/sec (Irvine et al, 1992).

Land use within the drainage basin varies. Much of the upper portion of the drainage basin is characterized by woods and farmland, but prior to joining the river, the tributary creeks also pass through several small communities and receive both industrial and municipal discharges. The lower portion of the river (9 km) is densely urbanized and industrialized. Active industries (e.g., cereal and flour mills, dye plants, gas compressors, chemical refineries) and abandoned grain elevators dominate the shoreline.

The seasonal flow of the river varies widely and corresponds to seasonal differences in precipitation. In general, the region experiences cold and snowy winters and moderately warm summers, the moderating effect becoming more pronounced nearer the lake. Current velocities vary considerably in the river; Lake Erie and the Buffalo River are subject to wind-driven waves and seiches, which can result in rapid and large changes in water levels (more than 2 m over a 9 hr period, Irvine et al. 1992) that are greater than the average annual variation (1.7 m, International Joint Commission 1981). Locally, the passage of large vessels can also cause similar short-term changes in water level as well as current velocities (McNabb et al. 1986).

The average annual depth of the river varies from about 2 m at the confluence of Cayuga and Buffalo creeks to over 8.5 m at the mouth. To improve navigation and develop industry, the lower 9.4 km of the river have been dredged to a depth of at least 6.7 m below low water datum, with widths ranging from 46 m to 245 m to allow passage of commercial deep-draft vessels (Poole et al. 1994).

Methods

Published fish catch data from four reaches in the lower 9.7 km of the river for 1981 (Makarawicz et al. 1982) and 1992 (Kozuchowski et al. 1993) were used for this study. Reach 1 extends from the mouth at Lake Erie upstream for approximately 1.5 km; reach 2 extends from reach 1 upstream for approximately 2.5 km; reach 3 extends from reach 2 upstream for approximately 4.0 km; and, reach 4 extends from reach 3 upstream for approximately 1.7 km to Cazenovia Creek (Figure 2).

In both Markarwicz et al. 1982 and Kozuchowski et al. 1993, sampling for adult fishes was conducted at each reach with an electrofishing boat following a path parallel to the shoreline in depths less than 3 m. In 1981, electrofishing was conducted for approximately four minutes at each reach on one or two days per month from April through December (Markarawicz et al. 1982). In 1992, electrofishing was conducted for approximately 30 minutes at each reach once a week from April through July (Kozuchowski et al. 1993). At the end of the electrofishing period, the fish were identified to species

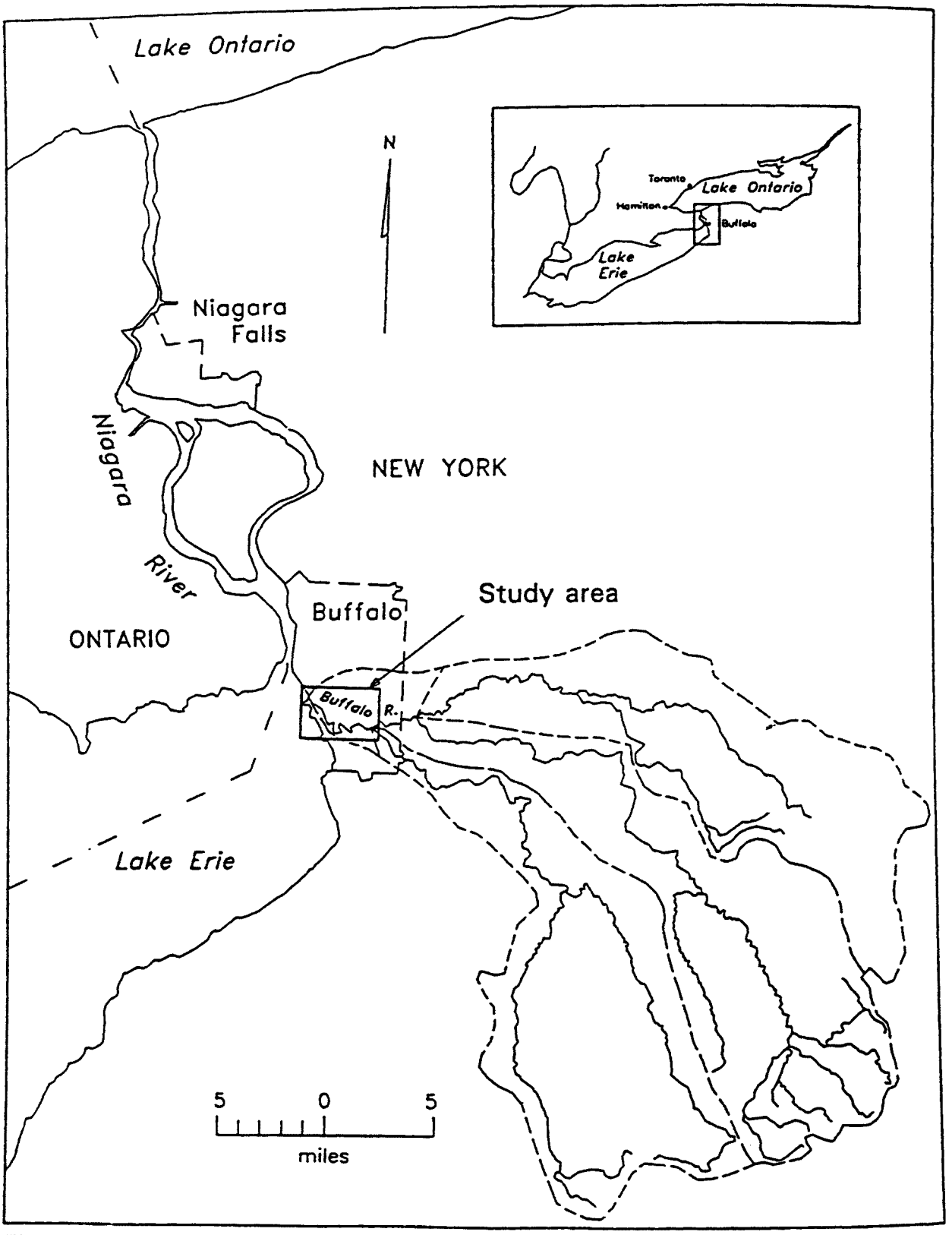

Figure 1. Buffalo River drainage basin.

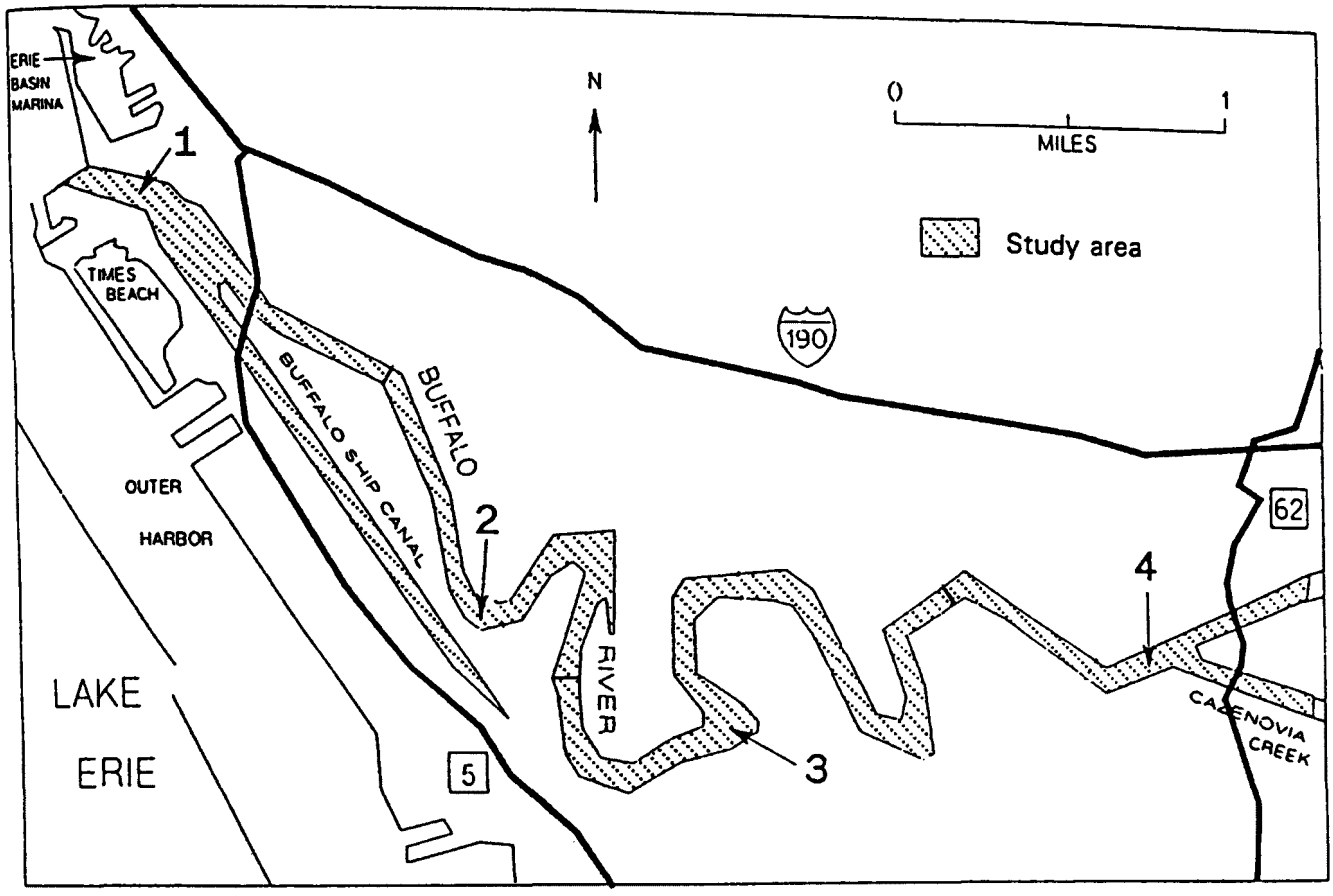

Figure 2. The lower Buffalo River.

(Eddy 1969, Werner 1980), measured, and released in the same reach (Markarwicz et al. 1982, Kozuchowski et al. 1993).

Only data for the months of April through July were used in this analysis. The data were converted to a standard catch-per-unit-effort measurement for analysis.

Analyses

Diversity, percent similarity and an index of biotic integrity were calculated for each reach, for 1981 and 1992. Diversity is a measure of the number of species (i.e., richness) in a community and their relative abundances (i.e., evenness). Low diversity refers to few species or unequal abundances; high diversity to refers to many species or equal abundances. Diversity was calculated for each sampling date at each reach using the Shannon-Wiener diversity index (Washington 1984):

$$H' = -\sum_{i=1}^{S} (p_i \ln p_i)$$

$p_i = n_i/N$
S = no. of species
n_i = no. of organisms in the i^{th} species
N = total no. of organisms

The resultant values were then averaged by reach and by year for 1981 and 1992. Statistical comparisons of the fish community at each reach were made between years using the student's t-test (Excel, Microsoft Corporation).

Percent similarity (PSC) is an index used to measure community similarity (or dissimilarity). The PSC was used to compare the similarity of the fish community at each reach between 1981 and 1992. The PSC was calculated using the Whittaker and Fairbanks' index (Washington 1984):

$$PSC = 100 - 0.5 \sum_{i=1}^{K} |a-b|$$

K = the total number of species combined from 1981 and 1992
a = relative percentage of species i from 1981
b = relative percentage of species i from 1982

The PSC comparing 1981 and 1992 was calculated for each reach. Percent similarity ranges from 0, when two samples contain no species in common, to 100, when the two samples are identical in both species and individual abundances. The PSC can fail if the relative proportions of taxa remain the same but the overall abundance changes; this fault may not be disqualifying in fisheries studies of polluted rivers as environmental changes tend to alter dominance relationships. If the balance of taxa is nearly identical then the communities may be functionally the same (Washington 1984).

The Index of Biotic Integrity (IBI) is a composite index that integrates characteristics of the community, population, and individual organisms. IBI parameters or metrics include species richness, indicator taxa, trophic guilds, fish abundance, hybridization, and diseases and anomalies. An IBI for the Buffalo River was developed on the basis of methods described in Karr (1981) and Lyons (1992). Eleven metrics similar to those developed by Karr et al. (1986) were used (Table 1). The twelfth metric defined by Karr, proportion of individuals with disease, tumors, fin damage, and skeletal anomalies, was dropped because this information was not available for the years evaluated. The sixth metric was redefined from proportion of individuals as green sunfish to proportion of individuals as carp, goldfish, or carp-goldfish hybrids, because green sunfish do not occur in the Buffalo River watershed.

Each metric consists of fishes that belong to a single structural or functional group. Individual species may belong to more than one group, and hence, contribute to more than one metric. Once the metrics were defined, all the fish species found in the Buffalo River from both sampling periods were characterized according to taxonomic groups, feeding guilds, and tolerance to environmental degradation (Table 2).

Physical characteristics also influence community structure and function. An important physical factor that influences community structure and function is stream size (Lyons 1992). In general, the total number of fish and the number of species increase with stream size. To account for this change, metrics 1, 2, 3, 4, 5, and 10 were modified based on the stream-order of each reach.

Stream-order is a classification system used to indicate progressive increases in stream size (Strahler 1957, as cited in Karr et al. 1986). The smallest streams, highest in the watershed, are first-order. When two first-order streams join, they form a stream of the second-order; when two second-order streams join, they form a stream of the third-order, etc. Applying this classification system to the Buffalo

Table 1. Criteria used to score Index of Biotic Integrity metrics for the Buffalo River, New York (modified from Karr et al. 1986).

Category	Metric	Scoring criteria 5	3	1
Species richness and composition	1. Total number of fish species			
	2. Number of darter species			
	3. Number of centrarchid species (excluding *Micropterus* spp.)	Expectations for metrics 1 through 5 vary with stream order. See Appendix A.		
	4. Number of sucker species			
	5. Number of intolerant species			
	6. Percent carp, goldfish, or carp x goldfish	<5	5 - 20	>20
Trophic function	7. Percent omnivores[1]	<21	22 - 45	>45
	8. Percent macroinvertivorous cyprinids[2]	>45	45 - 20	<20
	9. Percent piscivores[3]	>5	5 - 2	<2
Fish abundance	10. Number of individuals collected, expressed as CPUE[4]	Expectations for metric 10 vary with stream order. See Appendix A.		
	11. Percent hybrids	0	>0 - 1	>1

[1] Omnivores - species with diets composed of at least 25 percent plant material and ≥25 percent animal material.
[2] Macroinvertivorous cyprinids - *Cyprinidae* spp. with diets composed of >75 percent macroinvertebrates (insects, crustaceans, annelids, etc.).
[3] Piscivores - species with diets composed of >75 percent fish.
[4] CPUE = catch-per-unit-effort (total number of individuals / total electrofishing effort, in minutes).

Table 2. Taxonomic group, trophic guild, and identification of intolerant species of adult fishes collected in the Buffalo River watershed since 1928. ***Taxa***: A = Catostomid; D = Darter; E = Centrarchids, excluding Micropterus spp.; Y = Cyprinid. ***Trophic guild***: O = omnivore; P = planktivore; F = piscivore; M = macroinvertivore.

Scientific Name	Common Name	Taxa	Trophic Guild	Intolerant Species
Lepisosteidae				
Lepisosteus osseus	longnose gar	-	F	-
Clupeidae				
Alosa pseudoharengus	alewife	*	P	-
Dorosoma cepedianum	gizzard shad	-	P	-
Cyprinidae				
Carassius auratus	goldfish	Y*	O	-
Cyprinus carpio	carp	Y*	O	-
(Hybrid)	carp x goldfish	Y*	O	-
Notemigonus crysoleucas	golden shiner	Y	P	I
Nocomis micropogon	river chub	Y	M	I
Cyprinella spiloptera	spotfin shiner	Y	M	-
Notropis hudsonius	spottail shiner	Y	O	I
Notropis atherinoides	emerald shiner	Y	P	-
Pimephales notatus	bluntnose minnow	Y	O	-
Notropis cornutus	common shiner	Y	O	I
Semotilus atromaculatus	creek chub	Y	O	-
Catostomidae				
Carpiodes cyprinus	quillback	A	M	-
Catostomus commersoni	white sucker	A	M	-
Hypentelium nigricans	northern hog sucker	A	O	I
Moxostoma sp.	redhorse spp.	A	M	-
Ictaluridae				
Ameiurus nebulosus	brown bullhead	-	O	-
Ameiurus melas	black bullhead	-	O	-
Ictalurus punctatus	channel catfish	-	O	-
Noturus flavus	stonecat	-	M	I
Esocidae				
Esox lucius	northern pike	-	F	-
Esox masquinongy	muskellunge	-	F	I
Osmeridae				
Osmerus mordax	rainbow smelt	*	P	-
Salmonidae				
Oncorhynchus mykiss	rainbow trout	*	M	I
O. kisutch	coho salmon	*	F	I
O. tshawytscha	chinook salmon	*	F	I
Salmo trutta	brown trout	*	F	I
Percopsidae				
Percopsis omiscomaycus	trout-perch	-	M	I

Percichthyidae				
Morone americana	white perch	*	M	-
Morone chrysops	white bass	-	M	-
Centrarchidae				
Ambloplites rupestris	rock bass	E	M	I
Lepomis gibbosus	pumpkinseed	E	M	-
Lepomis macrochirus	bluegill	E	M	-
Micropterus dolomieu	smallmouth bass	-	F	-
Micropterus salmoides	largemouth bass	-	F	-
Pomoxis annularis	white crappie	E	P	-
Pomoxis nigromaculatus	black crappie	E	P	I
Lepomis gulosus	warmouth	E	M	-
Percidae				
Etheostoma nigrum	johnny darter	D	M	-
Perca flavescens	yellow perch	-	M	-
Percina caprodes	logperch	D	M	-
Stizostedion vitreum	walleye		F	-
Sciaenidae				
Aplodinotus grunniens	freshwater drum	-	M	-

* Denotes nonindigenous species

Table 3. Total index of biotic integrity scores, integrity classes, and the attributes of those classes (modified from Karr et al. 1986).

Total IBI score (sum of the 11 metric ratings)	Integrity class	Attributes
47 - 55	Excellent	Comparable to the best situations without human disturbance; all regionally expected indigenous species for the habitat and stream order, including the most intolerant forms, are present with a full array of age (size) classes; balanced trophic structure.
38 - 46	Good	Species richness somewhat less than best situations, especially due to the loss of the most intolerant forms; some species are present with less than optimal abundances or size distributions; trophic structure shows some signs of stress.
29 - 37	Fair	Signs of additional deterioration include loss of intolerant forms, fewer species, highly skewed trophic structure (e.g., increasing frequency of omnivores and carp, goldfish, or carp x goldfish); older age classes of top piscivores may be rare.
20 - 28	Poor	Dominated by omnivores, tolerant forms, and habitat generalists; few piscivores; growth rates and condition factors commonly depressed; hybrids and diseased fish often present.
11 - 19	Very poor	Few fish present, mostly non-indigenous or tolerant forms; carp, goldfish, or carp x goldfish common; disease, parasites, findamage, and other anomalies regular.
	No fish	Repeated sampling finds no fish.

River, reaches 1 and 2 were classified a stream-order 6 and reaches 3 and 4 were classified a stream-order 5.

Fisheries data for the whole basin collected since 1928 were used to develop plots of species versus stream-order, and the scoring criteria for those 6 metrics was estimated from those plots (Appendix A). The scores from each metric for each year and reach were summed to give the overall IBI score for each reach and year. The maximum possible IBI score is 55 and the lowest, 11. The IBI scoring range was divided into 5 sub-ranges, or integrity classes, qualified as "excellent" (high IBI score) to "very poor" (low IBI score) (Table 3). The IBI scores for each reach were compared between years.

Results

In comparing 1981 and 1992, there was virtually no change in diversity (H') at reach 1 (river mouth); however, there was an increase in H' at reaches 2, 3, and 4 (Figure 3). The increase was significant at reach 3 ($P<0.0001$) which went from the lowest H' in 1981 to the highest H' in 1992. In 1981, there was a trend of decreasing diversity from reach 1 (the mouth) going upstream to reach 3. In 1992, H' was fairly similar for all sites.

Percent similarity (PSC) indicated that, when comparing 1981 and 1992, the fish community of reach 2 remained the most similar (Figure 4). The PSC for reaches 1, 3, and 4 were less similar.

The IBI improved from 1981 to 1992 at all reaches (Figure 5). IBI scores reveal significant improvement at reaches 2, 3, and 4, although the scores still reflect some degree of stress. The integrity class remained "fair" for reach 1, with only a slight (6%) increase from 1981 to 1992. The IBI score increased most dramatically (82%) at reach 2 with a change in integrity class rating from "very poor" to "fair." The increase was largely due to the decline in percent hybrids (metric 11); a property not addressed by H'. The IBI score for reach 3 increased by 10 points (37%) and for reach 4 increased by 14 points (61%), with integrity class changes from "poor" to "fair." In 1981, there was a trend from the mouth going upstream of decreasing biotic integrity; in 1992, biotic integrity was similar for all reaches. Reach 1 displayed the least difference in IBI score between 1981 and 1992.

Closer inspection reveals that the improvements in IBI scores at reaches 2, 3, and 4 were due primarily to increases in the number of: total species (metric 1); centrarchid species (metric 3); sucker species (metric 4)(reaches 3 and 4); and, intolerant species (metric 5)(reaches 2 and 3), and decreases in the percentage of omnivores (metric 7)(reaches 2 and 4)(Figure 6). Overall, IBI metric 3 (centrarchids, excluding Micropterus) showed the greatest net increase in score from 1981 to 1992.

Discussion

Indices of diversity (H') and biotic integrity (IBI) reveal discernable improvement in the fish community of the lower Buffalo River between 1981 and 1992. Diversity generally increased (significant at reach 3), as did biotic integrity (significant at reaches 2, 3, and 4) during the period evaluated. Total number of species (richness) increased and abundance (CPUE) remained the same, except at reach 1 where both declined slightly.

Overall, PSC's were not consistently compatible with observed increases in H' and the IBI, especially at reaches 1 and 2. The degree of similarity in fish community composition (PSC) between 1981 and 1992 was very low at reaches 1, 3, and 4, and moderate at reach 2, indicating a high degree of heterogeneity over time. Intuitively, when there is a significant change in IBI score between years, one would expect a low PSC; when there is no significant change in IBI score, a high PSC might be expected. This premise is not substantiated by results for reaches 1 and 2.

At reach 1, in 1981, 61 percent of the total catch was composed of four species, ranging from 10 to 29 percent each. In 1992, 62 percent of the total catch was composed of the same four species; however, one migratory lake species overwhelmingly predominated (emerald shiner, 50 percent of total catch). The prevalence of this species in 1992, weighed heavily in the PSC calculation in which the relative proportions of taxa changed significantly. The PSC, therefore, was relatively low (37.8).

In contrast, values for H' and IBI were relatively unchanged at reach 1; furthermore, the species composition of the community at reach 1 was similar to nearshore-lake communities. This may indicate niche overlap and transition in community composition between the river and Lake Erie, potentially affecting IBI scores within the individual metrics. If so, the succession in habitat structure from riverine to

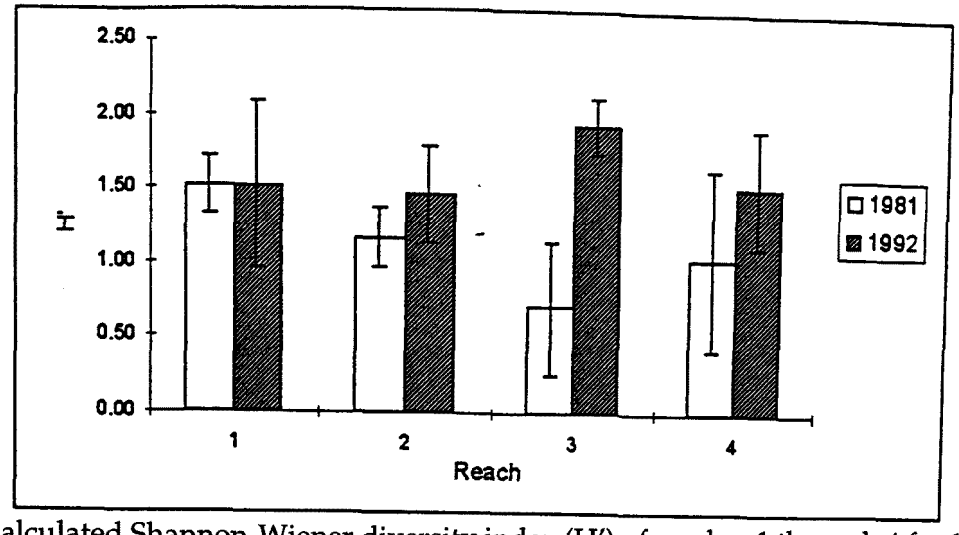

Figure 3. Calculated Shannon-Wiener diversity index (H') of reaches 1 through 4 for 1981 and 1992 (+/- standard deviation).

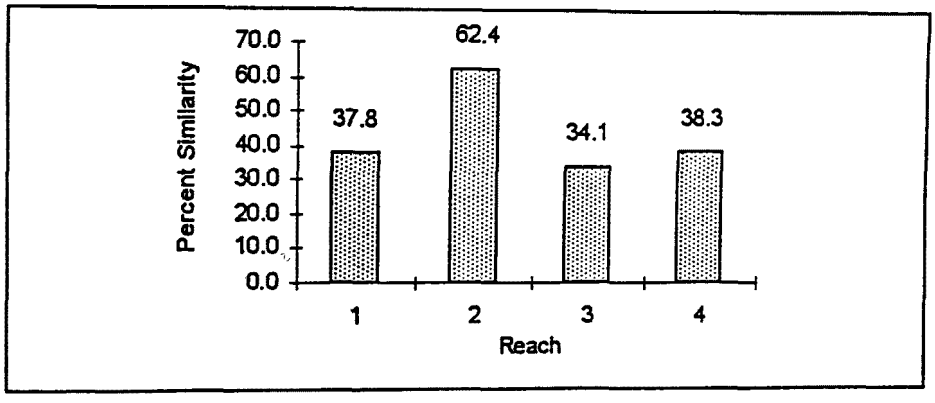

Figure 4. Calculated Percent Similarity Index (PSC) at each reach between 1981 and 1992.

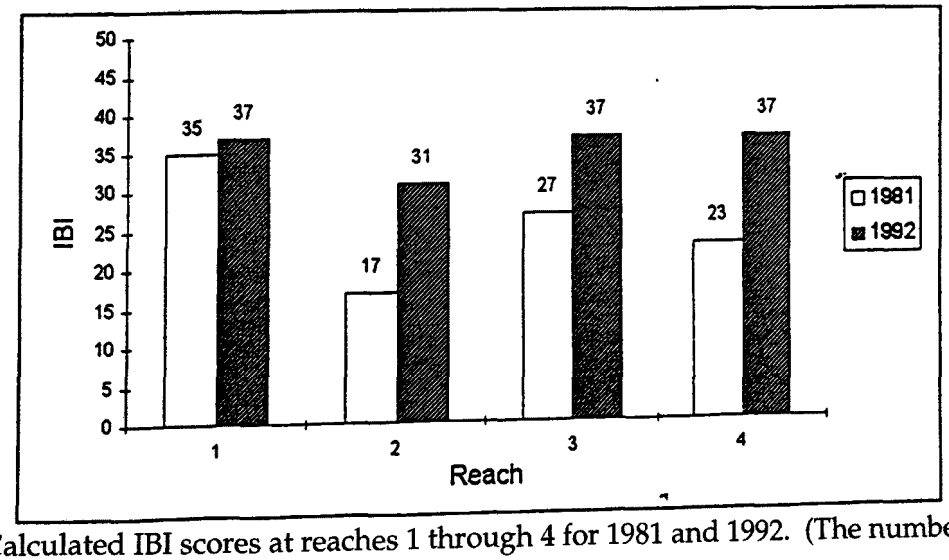

Figure 5. Calculated IBI scores at reaches 1 through 4 for 1981 and 1992. (The number at the top of each bar is the actual IBI score.)

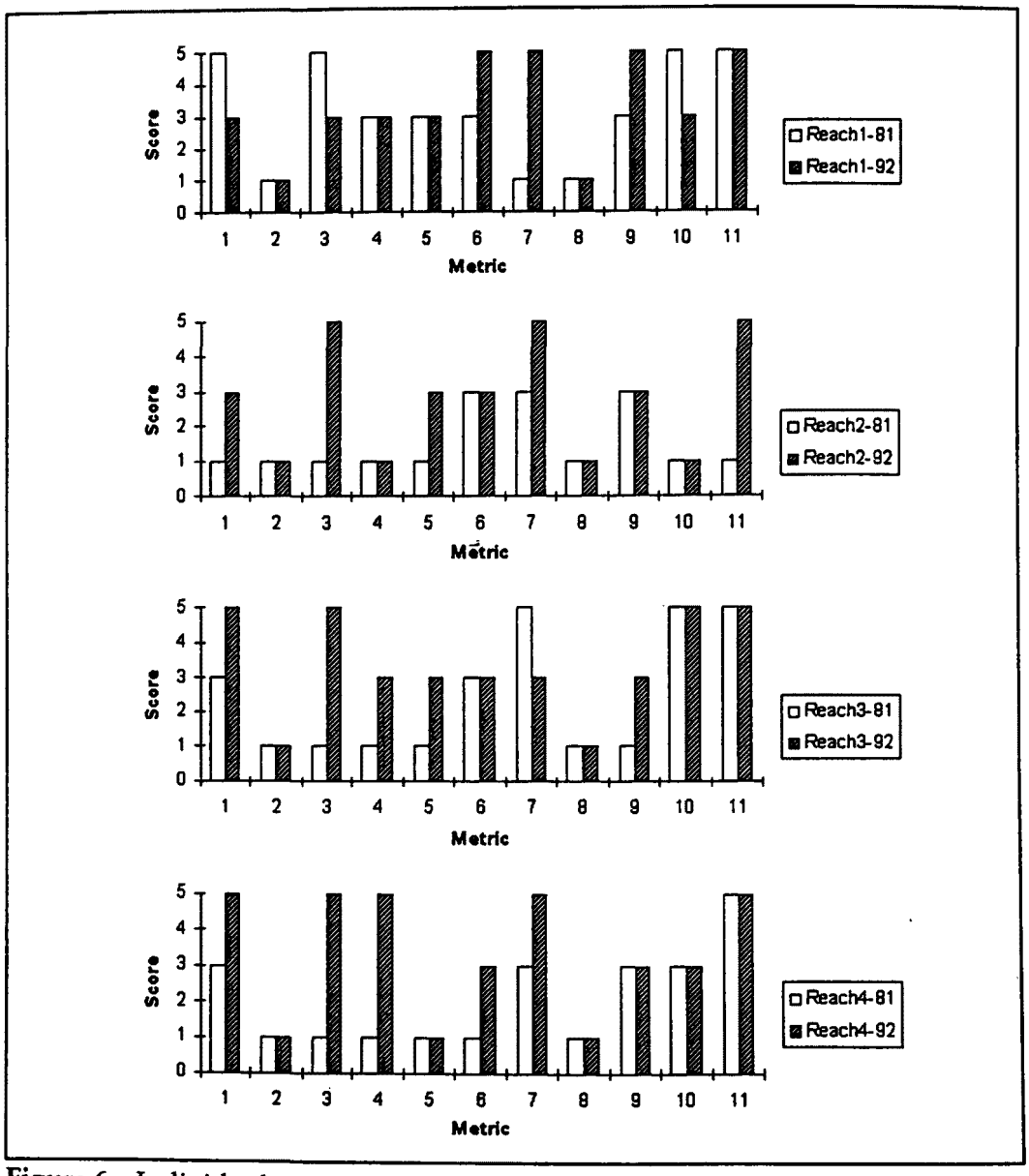

Figure 6. Individual metric scores at each reach for 1981 and 1992.

estuarine would require modification of the IBI metrics used.

At reach 2, in 1981, 76 percent of the total catch was composed of four species; two-thirds of which was due to two migratory lake species (33 percent emerald shiner and 11 percent gizzard shad). In 1992, the same four species accounted for 80 percent of the total catch, of which emerald shiner and gizzard shad accounted for 37 and 23 percent, respectively. Twelve species were found in 1992 (accounting for about 19 percent of the total catch) that were not found in 1981. The presence of these additional, relatively uncommon species in 1992, however, did not weigh heavily in the PSC calculation because the relative proportions of the predominant taxa remained the same. The PSC, therefore, was relatively high (62.4).

In contrast, since species were grouped by structure or function in the IBI, the effect of the uncommon species on IBI scores was pronounced. For example, at reach 2, centrarchids accounted for about 3 percent of the total catch in 1981 and 10 percent of the total catch in 1992. The number of species, however, increased from 1 in 1981 to 6 in 1992 (including 2 intolerant species) resulting in a 4 point increase in IBI score.

In the initial stages of environmental improvement, one can expect increasing diversification of community structure and progressive changes in community function. A secondary response in the community may follow, caused by the re-colonization of an intolerant species or altered interactions (i.e., competition). In the case of the lower Buffalo River, however, quantifiable relationships between recovery of the fish community and water quality improvement cannot be made.

Water chemistry monitoring data (i.e., dissolved oxygen, temperature, and conductivity) indicate dissolved oxygen increased between 1970 and 1980. In the early 1970's, migratory Lake Erie fishes were found in upstream reaches of the river where they had not been seen for at least thirty years (Sweeney and Merkel 1972). Since 1981, there has been no apparent improvement in water quality (Appendix B); therefore, recovery of the fish community appears to lag behind water quality improvements by 10 to 15 years. Improvements in the fish community at reaches 2, 3, and 4 may have occurred as a result of re-population by riverine and migrating nearshore-lake fishes (e.g., crappies and emerald shiner, respectively) and increased reproductive success. Long-term collection and analysis of larval, as well as adult, fish data will be necessary to pursue this line of inquiry.

In this study, we used a new approach to assess fish community changes in the lower Buffalo River based on three common-used indices. Our results show that the combined use of H', PSC, and an IBI minimized individual deficiencies within each index and avoided potential misinterpretations of the data. In order to enhance similar studies of the Buffalo River in the future, we offer the following suggestions:

Analytical methods should be developed prior to conducting sampling in order to gather relevant information. Our study was not designed with the intent of applying an IBI to the results, therefore several fish community attributes were not measured. Data on the incidence of external anomalies (i.e., deformities, fin erosion, lesions, tumors) would be especially useful given existing use impairments. Water chemistry data (e.g., dissolved oxygen, temperature, biochemical oxygen demand, suspended sediments) would also be useful in the absence of intensive water quality monitoring.

The IBI metrics used for this study should be modified before applying them to other large, lake-influenced rivers. Five metrics that we used were species-poor relative to the overall fish community. We believe that this may have caused the metrics to be more insensitive to environmental changes than when applied to smaller river systems. Also, a metric for larval fish (expressed as CPUE) should be added. Larval fish data would be useful to evaluate relative reproductive success.

Metric 2. Number of Percidae species. Four species of percids, including one species of darter, exist in the lower Buffalo River.

Metric 3. Number of centrarchid species. Eight species of centrarchids exist in the lower Buffalo River when two Micropterus species are included.

Metric 4. Number of benthic species. Four species of Ictaluridae, in addition to the four species of Catostomidae (suckers), exist in the lower Buffalo River.

Metric 8. Percent macroinvertivores. Eighteen species of macroinvertivores, including two cyprinid species, exist in the lower Buffalo River.

Metric 11. Percent non-indigenous species. Nine non-indigenous species, including hybrid carp x goldfish, exist in the lower Buffalo River.

Literature Cited

Bode, R. W., M. A. Novak, and L.E. Abele. 1993. Twenty-year trends in water quality of rivers and streams in New York State based on macroinvertebrate data 1972-1992. New York State Department of Environmental Conservation. Division of Water. Stream Biomonitoring Unit. Albany, NY. 196 pp.

Eddy, S. 1969. How to know the freshwater fishes. Wm. C. Brown Company. Dubuque, IA. 286 pp.

Fausch, K. D., J. R. Karr, and P. R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. Transactions of the American Fisheries Society 113:39-55.

Ford, J. 1989. The effects of chemical stress on aquatic species composition and community structure. pp. 99-144, (eds.) In S. A. Levin, M. A. Harwell, J. R. Kelly and K. D. Kimball, Ecotoxicology: problems and approaches. Springer-Verlag. New York, NY.

Greeley, J. R. 1929. Fishes of the Erie-Niagara watershed. pp. 150-180, In A biological survey of the Erie-Niagara System. Suppl. 18th Annual Report (1928), New York Conservation Department. Albany, NY. 244 pp.

Gray, J. S. 1989. Effects of environmental stress on species rich assemblages. Biological Journal of the Linnean Society 37:19-32.

Hocutt, C. H. 1981. Fish as indicators of biological integrity. Fisheries 6(6):28-31.

International Joint Commission. 1981. Lake Erie water level study. Appendix A, Vol. 1, *Lake Regulation.* International Joint Commission. Lake Erie Regulation Study Board. Winsor, Ontario.

Irvine, K. N., G. P. Stein, and J. K. Singer. 1992. An environmental guidebook to the Buffalo River. In Proceedings of the International Symposium on Environmental Dredging, September 30 to October 2, 1992, Buffalo, NY. 30 pp.

Irvine, K. N. and G. W. Pettibone. 1996. Planning level evaluation of densities and sources of indicator bacteria in a mixed land use watershed. Environmental Technology 17:1-12.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6(6):21-27.

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

Kozuchowski, E. S., E. A. Poole, and C. E. Lowie. 1993. Buffalo River fisheries assessment: report on the results of the 1992 larval and adult fish survey. U.S. Department of the Interior, Fish and Wildlife Service, Lower Great Lakes Fishery Resources Office, Amherst, NY. Administrative Report 93-03. 26 pp.

Lee, C. R., D. L. Brandon, J. W. Simmers, H.E. Tatem, and J. G. Skogerboe. 1991. Information summary, area of concern: Buffalo River, New York. Misc. paper EL-91-9, Department of the Army, Corps of Engineers, Vicksburg, MS. 29 pp.

Lyons, J. 1992. Using the index of biotic integrity (IBI) to measure environmental quality in warmwater streams of Wisconsin. General Technical Report NC-149. United States Department of Agriculture. Forest Service, North Central Forest Experiment Station, St. Paul, MN. 51 pp.

Makarewicz, J.C., R.C. Dilcher, J.M. Haynes, and K. Shump. 1982. Biological survey: Buffalo River and outer harbor of Buffalo, NY. Final Report, Volumes 1 and 2. State University College at Brockport, Biology Department, Brockport, NY.

McNabb, C. D., T. Batterson, J. Craig, P. Roettger, and M. Siami. 1986. Effects of commercial ship passage on emergent wetlands in Lake Nicolet, St. Marys River, 1984 (Part II) In Environmental baseline studies during 1984 of Lake Nicolet and Munuscong Bay, St. Marys River, MI, in relation to proposed extension of the navigation season. C.R. Liston and C.D. McNabb, principal investigators. U.S. Department of the Interior, Fish and Wildlife Service, Twin Cities, MN and U.S. Army Corps of Engineers, Detroit District Office, MI. Biological Report 86(3). 78 pp.

Mikol, G., K. Roblee, and M. Wilkinson. 1993. Fish and wildlife habitat inventory and assessment of the lower Buffalo River Watershed. New York State Department of Environmental Conservation, Albany, NY. 86 pp.

New York State Conservation Department (NYSCD). 1929. A biological survey of the Erie-Niagara system. Supplemental to the Eighteenth Annual Report, 1928.

New York State Department of Environmental Conservation (NYSDEC). 1989. Buffalo River remedial action plan. New York State Department of

Environmental Conservation, Division of Water, Buffalo, NY.

New York State Department of Environmental Conservation (NYSDEC). 1990. Rotating intensive basin studies, Appendix A, water column. New York State Department of Environmental Conservation, Division of Water, Albany, NY. 114 pp.

New York State Department of Environmental Conservation (NYSDEC). 1993a. Priority water problem list - Region 9. New York State Department of Environmental Conservation, Division of Water, Buffalo, NY.

New York State Department of Environmental Conservation (NYSDEC). 1993b. Reprint of the NYS rivers inventory resource report for the Buffalo River Watershed. New York State Department of Environmental Conservation, Division of Water, Raybrook, NY.

Poole, E. A., E. S. Kozuchowski, and C. E. Lowie. 1994. Fish and wildlife habitat restoration on the Buffalo River, Buffalo, New York. Prepared by U.S. Department of the Interior, Fish and Wildlife Service, Lower Great Lakes Fishery Resources Office, Amherst, New York, for Erie County Department of Environment and Planning, Buffalo, NY. 26 pp.

Sauer, D. E. 1979. An environmental history of the Buffalo River. Text of presentation, 50th meeting, New York Water Pollution Control Association, New York City, January 1979. Buffalo Color Corporation, Buffalo, NY. 88 pp.

Sweeney, R. A. 1971. Survey of benthic macroinvertebrates and analysis of water and sediments from the Buffalo River 1970. State University College at Buffalo, Great Lakes Laboratory, Buffalo, NY. 24 pp.

Sweeney, R. A. 1972. River on the mend. Limnos 5(2).

Sweeney, R. A. and C. Merkel. 1972. Survey of benthic macroinvertebrates and analysis of water and sediments from the Buffalo River, 1972. State University College at Buffalo, Great Lakes Laboratory, Buffalo, NY. 26 pp.

Ward, C. A .B. 1980. A survey of the crustacean zooplankton in the Buffalo River, 1979, Buffalo River Study X. State University College at Buffalo, Great Lakes Laboratory, Buffalo, NY. 103 pp.

Washington, H. G. 1984. Diversity, biotic, and similarity indices, a review with special relevance to aquatic ecosystems. Water Resources 18(6):653-694.

Werner, R. G. 1980. Freshwater fishes of New York State: a field guide. Syracuse University Press. Syracuse, NY. 186 pp.

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APPENDIX A

Scoring criteria for metrics 1, 2, 3, 4, 5, and 10
based on stream order.

Figure A1. Total number of fish species vs. stream order in the Buffalo River Watershed (Metric 1).

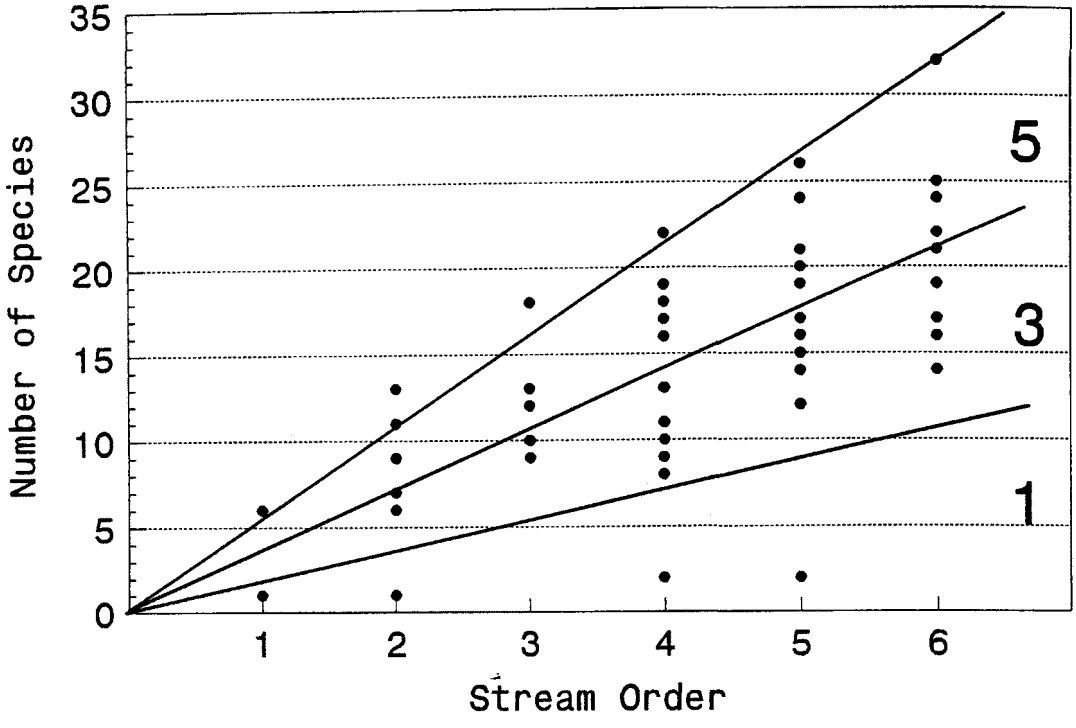


Figure A2. Total number of darter species vs. stream order in the Buffalo River Watershed (Metric 2).

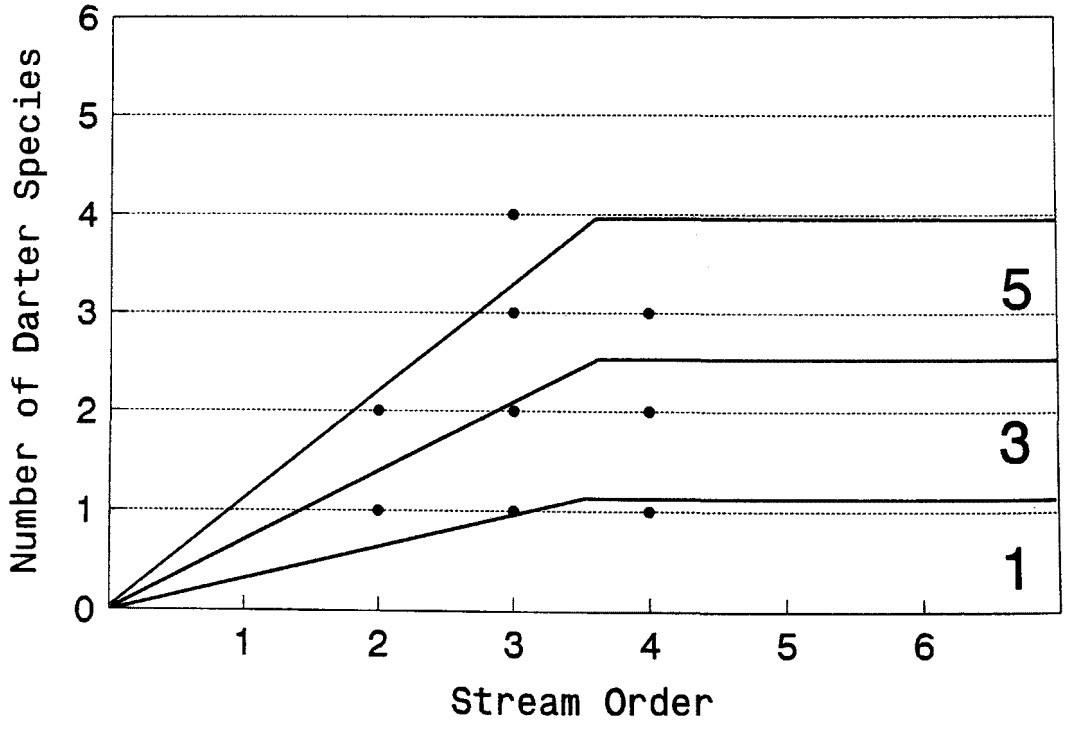


Figure A3. Total number of centrarchid species (excluding Micropterus) vs. stream order in the Buffalo River Watershed (Metric 3).

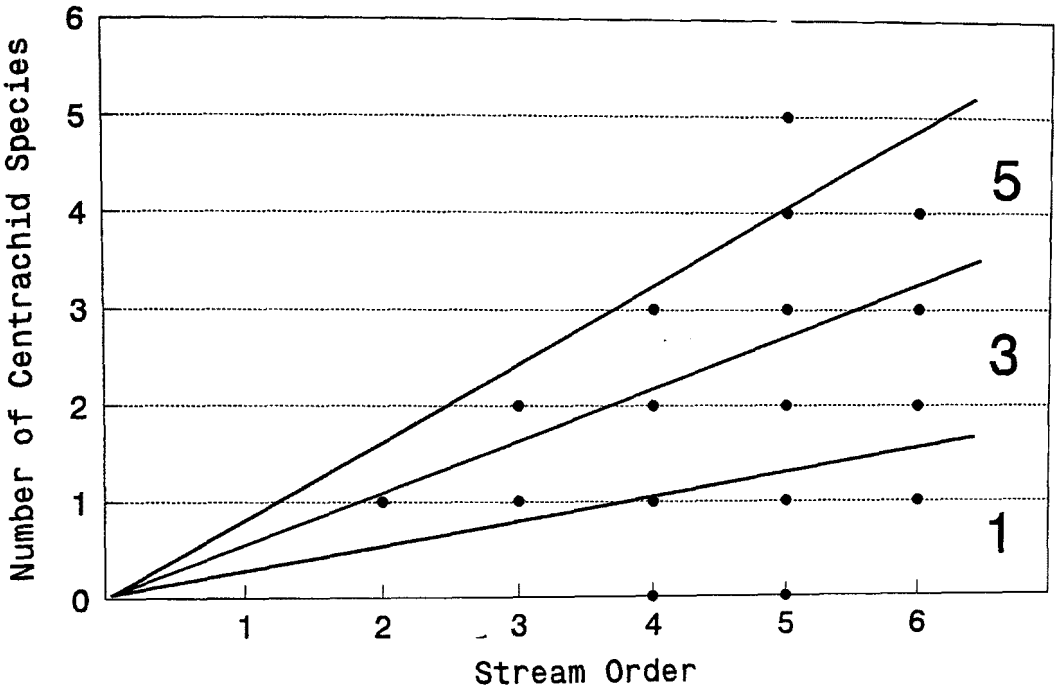


Figure A4. Total number of sucker species versus stream order in the Buffalo River watershed (metric 4).

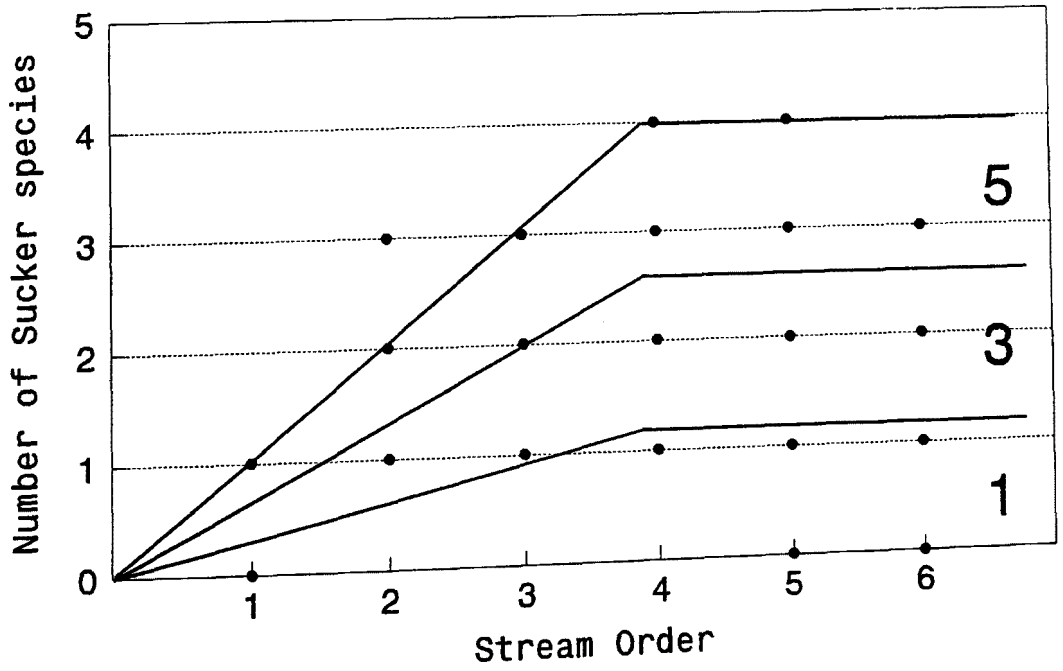


Figure A5. Total number of intolerant species vs. stream order in the Buffalo River Watershed (Metric 5).

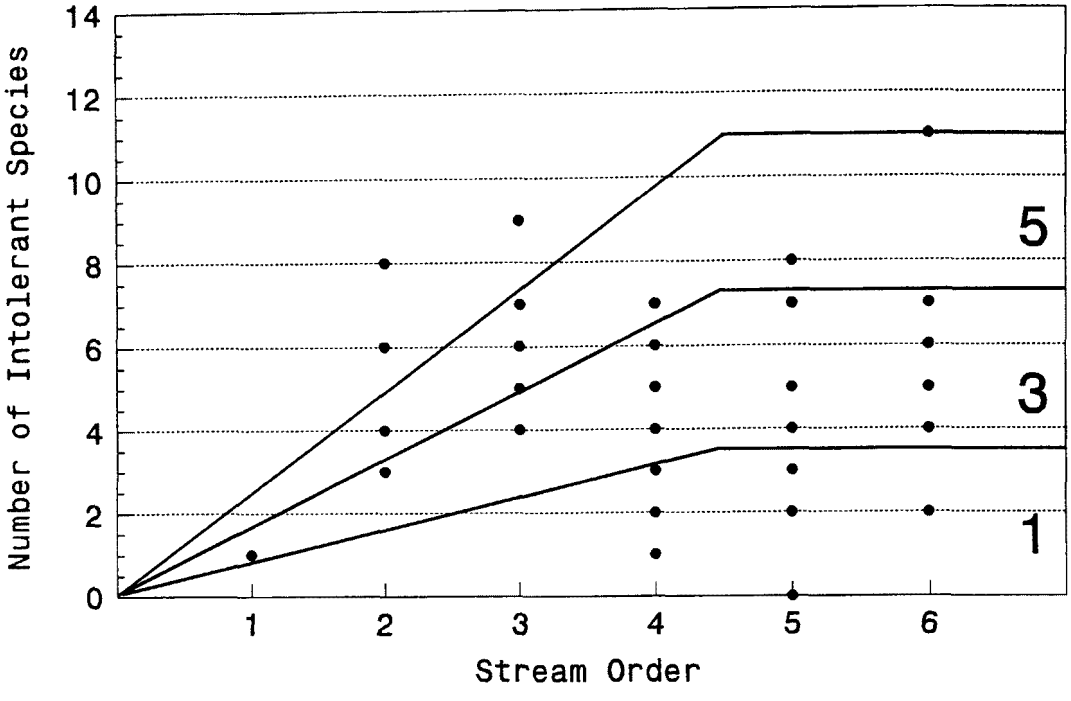
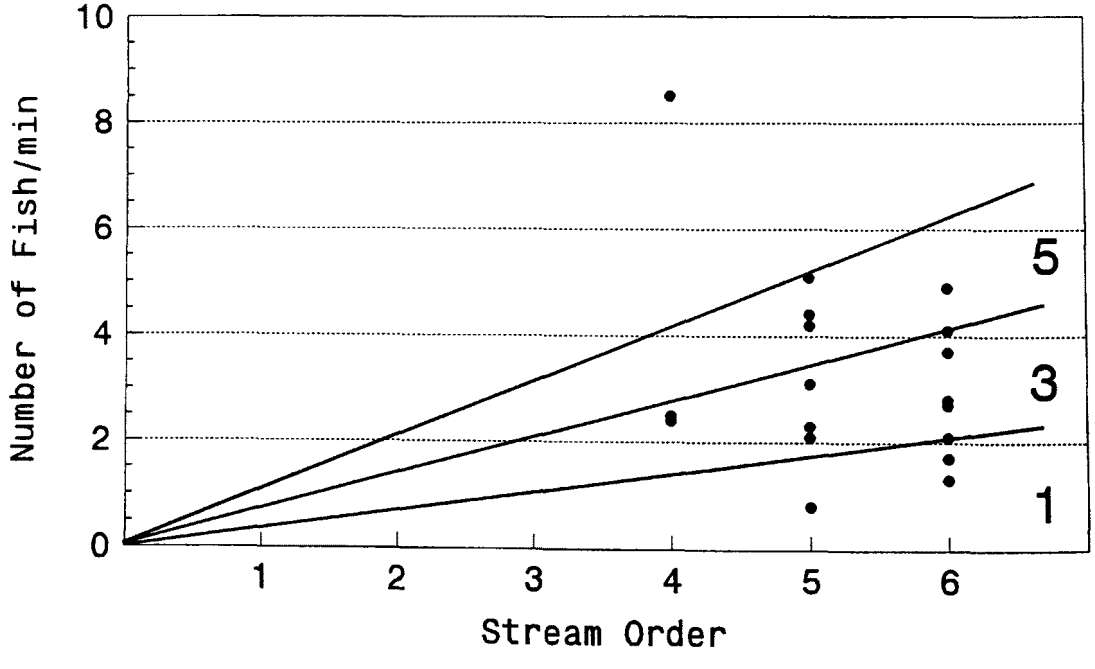


Figure A6. Number of individuals in sample (total fish collected for year / total electrofishing minutes for year) vs. stream order in the Buffalo River Watershed (Metric 10).



APPENDIX B

Water chemistry monitoring data, 1970 to 1993.

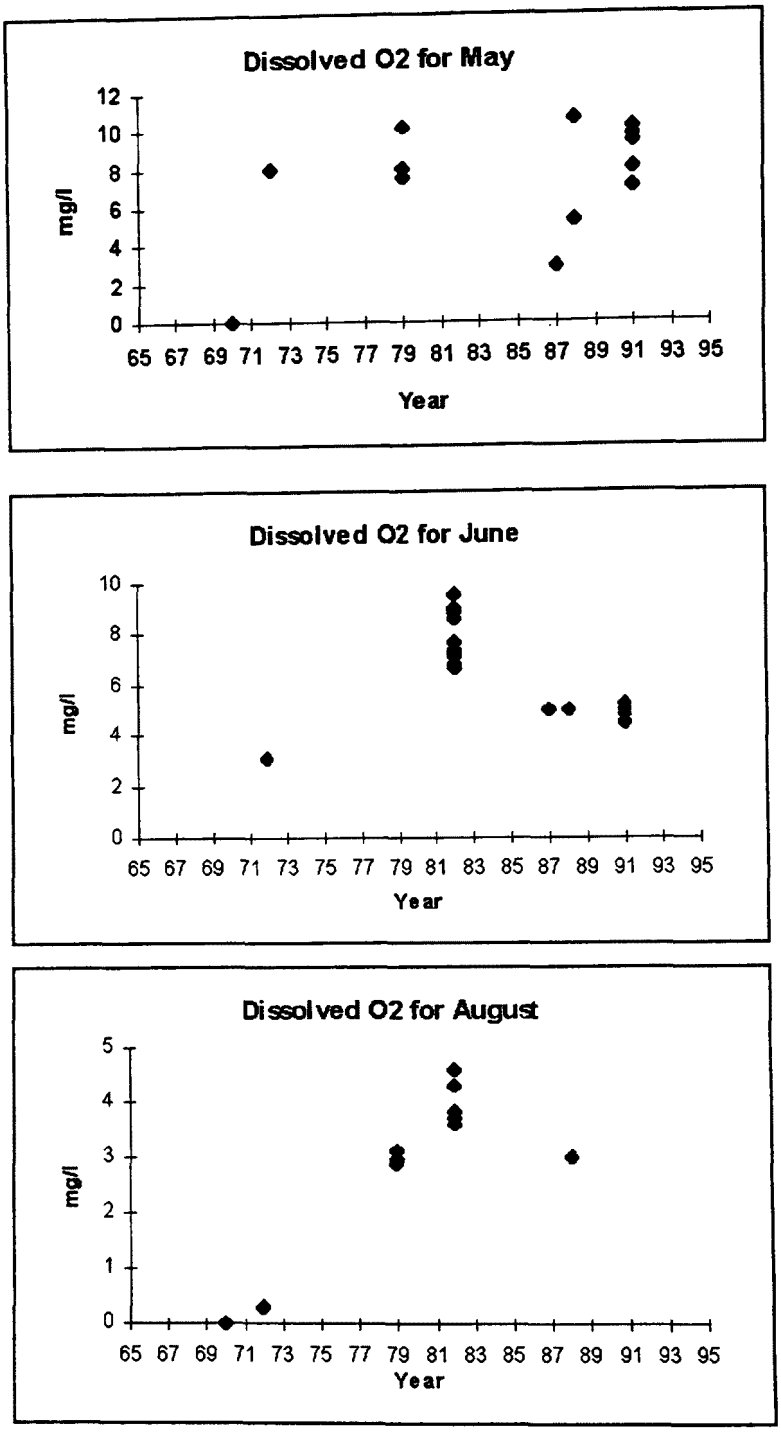
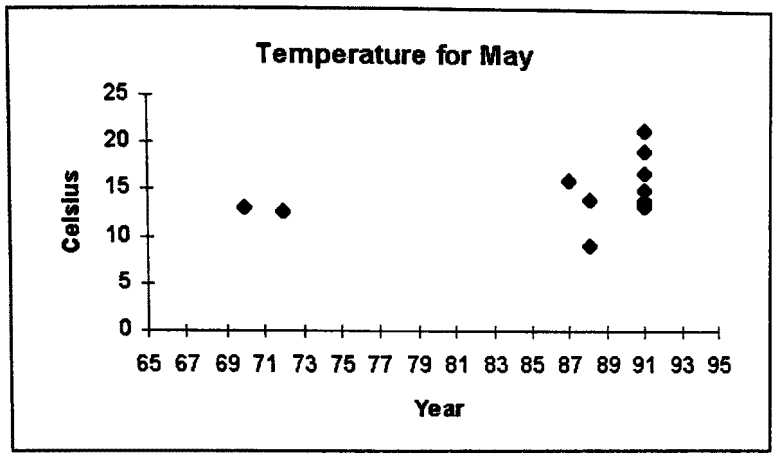
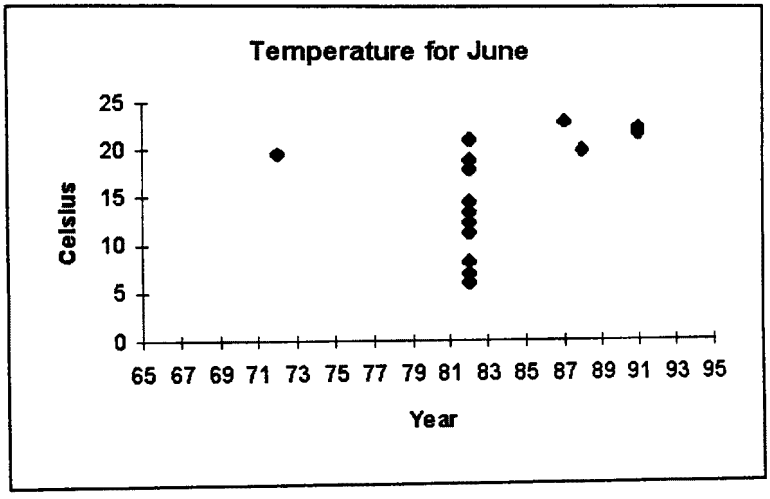


Figure B1. Dissolved oxygen (mg/l) in the Buffalo River measured near the Ohio Street Bridge (reach 2) during the months of May, June and August (Lee et al. 1991, Mikol et al. 1993, NYSDEC 1990, Sweeney 1970, Sweeney and Merkel 1972, Ward 1980).





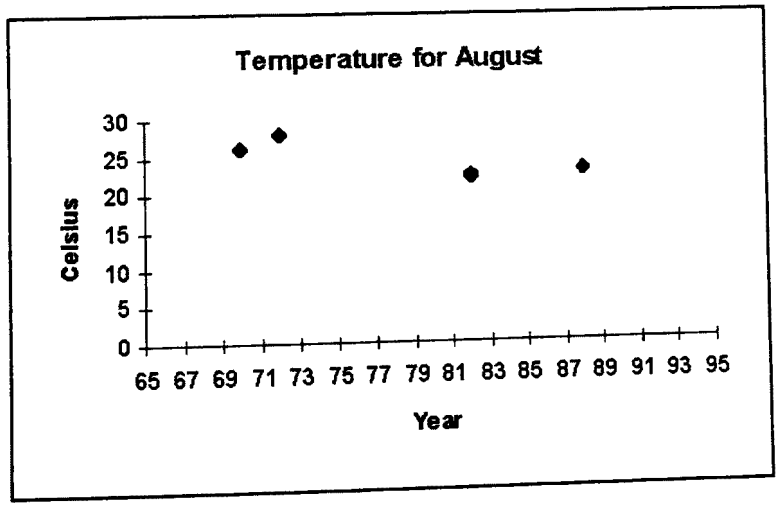


Figure B2. Temperature in the Buffalo River measured near the Ohio Street Bridge (reach 2) during the months of May, June and August (Lee et al. 1991, Mikol et al. 1993, NYSDEC 1990, Sweeney 1970, Sweeney and Merkel 1972, Ward 1980).

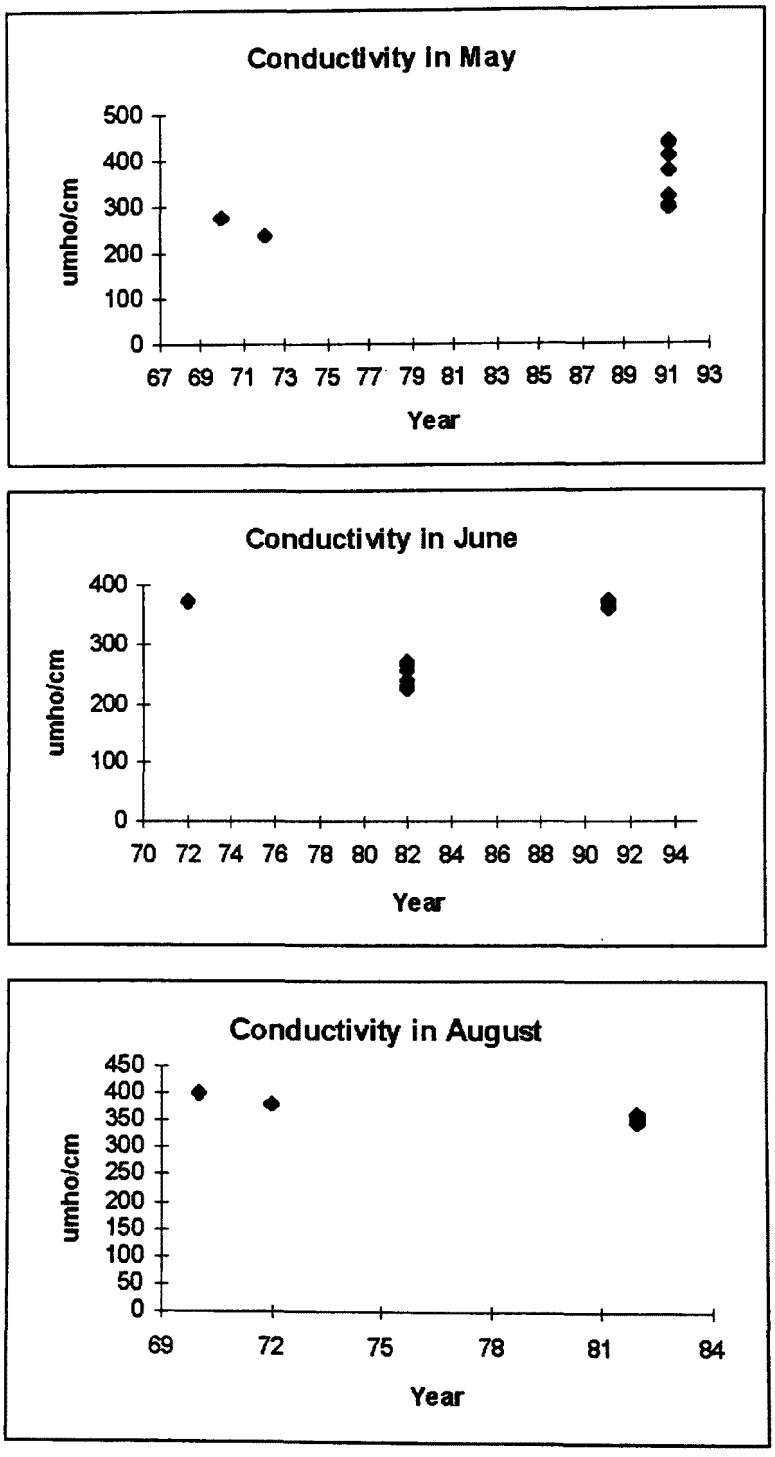
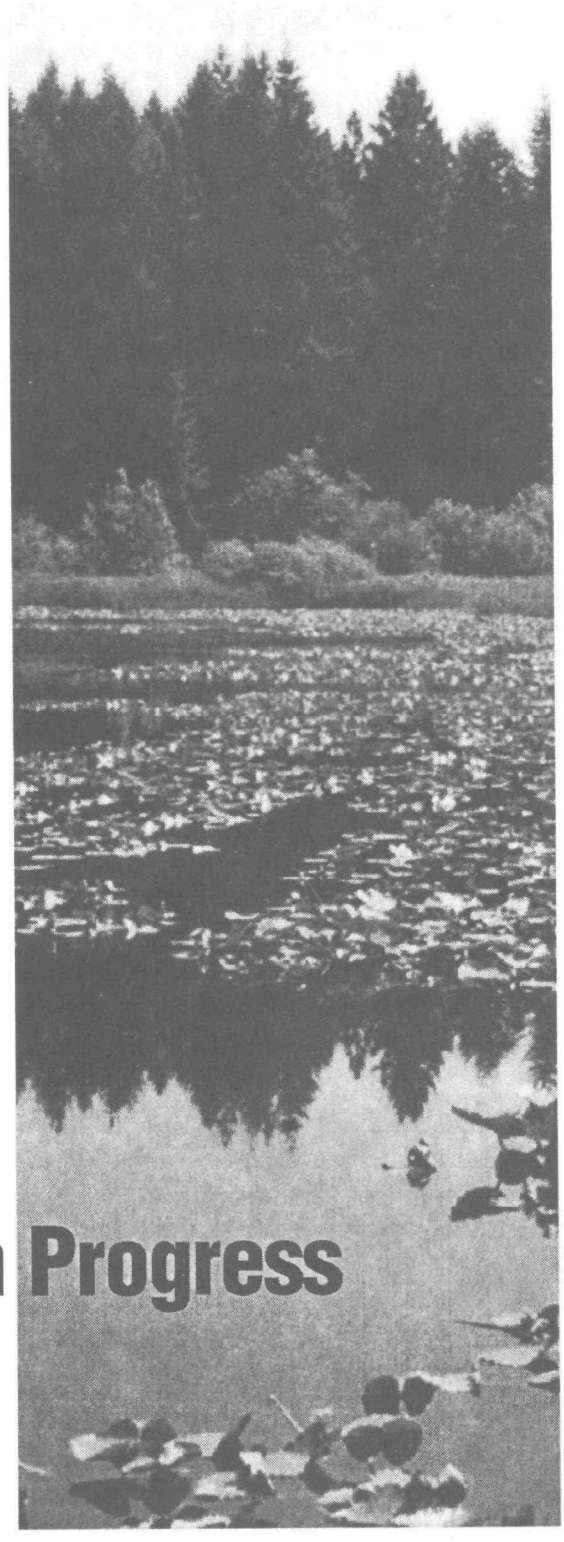


Figure B3. Conductivity in the Buffalo River measured near the Ohio Street Bridge (reach 2) during the months of May, June and August (Lee et al. 1991, Mikol et al. 1993, NYSDEC 1990, Sweeney 1970, Sweeney and Merkel 1972, Ward 1980).



Works in Progress

Selected Abstracts

"Seek out diverse friends. By just hanging out with them, cross-pollination of ideas occurs."

Mary Ann Norris
Mattel Strategic Planner
El Segundo–1997

Complexities of Species Conservation Under the Endangered Species Act: Ecosystem Management in the Cedar River Watershed

W. James Erckmann[1]

Ecological restoration and ecosystem management are essential parts of a Habitat Conservation Plan for the Cedar River Municipal Watershed, near Seattle, Washington, which is being prepared by Seattle Water under Section 10 of the Endangered Species Act. This paper will discuss the major elements of the plan (which may change), the procedural challenges to gaining approval, and lessons learned from the experience.

The HCP is intended to provide greater certainty for the region's water supply in the face of potential listing of a number of fishes affected by Seattle Water operations. The plan attempts at once to sustain ecological values, including protection of many at-risk species and their habitats, and human uses of the environment, including the drinking water for nearly one-fourth of the state population.

Like many other regional planning efforts, however, this plan requires approval of local elected officials, and federal government must be able to give assurances that implementation of the plan will provide a reasonable amount of certainty in water supply operations for Seattle Water, in turn requiring that the plan meet its ecological objectives.

While the commitments inherent in this HCP have substantial environmental benefits, the changing, unpredictable, and often conflicting political and legal contexts under which the plan is being developed make these commitments difficult to achieve. Uncertainties in state water resource planning, ambiguities inherent in the ESA, legal challenges to ESA regulations, and potential Congressional changes to the act combine to work against effective ecosystem management for the many at-risk species addressed by the plan.

Species of particular concern in the HCP include several anadromous salmonids, resident bull trout, northern spotted owl, and marbled murrelet. Restoration efforts focus primarily on protecting and restoring natural processes that create or maintain aquatic, riparian, and old-growth forest habitats in the 90,000-acre montane watershed, and recovery of depressed populations of anadromous salmonids affected by diversion of water from the Cedar River.

An ecological reserve comprising nearly 65 percent of the watershed, and all existing late-successional forest, will provide habitat protection and connectivity. Silvicultural and timber harvest guidelines outside the reserve, as well as limited experimental restoration of second growth inside the reserve, should result in long-term improvements in forest structure and habitat complexity in second-growth forests over time, and restoration of important riparian functions.

Watershed analysis forms the basis for planning forest road improvements and abandonment to reduce sediment loads to streams, and replacement of stream crossing structures that block fish passage. Forest practices are also constrained to minimize the likelihood of landslides and erosion resulting from cumulative impacts in specific basins.

The HCP will also include commitment to a specific instream flow regime to protect fish habitat, and other measures to protect and recover anadromous salmonids. Because good information on many of these and other key species is sorely lacking, the plan will also include elements of adaptive management, which presents additional challenges in the current regulatory arena.

Lessons in this process are still being learned, but it is already clear that the level of coordination among resource agencies needed for such a plan to be successful is not typical. The need to accommodate varying interests and overcome institutional barriers highlights the value of regular, open

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communications with all organizations, at both staff and management levels. This experience suggests that ecosystem planning will be most effective when a problem-solving, partnership mode can be adopted in the process.

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Restoration of Biodiversity from the Genetic to Landscape Levels: The Elwha River Restoration Project

Brian Winter[1]
Sarah Bransom[1]
Jeff Bohman[2]

The Elwha River Restoration Project offers an opportunity for the restoration of biodiversity at all levels of the biological hierarchy - from the genetic level to the landscape level (Winter and Hughes 1995). Most (83 percent) of the Elwha River Basin lies within Olympic National Park and is essentially pristine. Since the 1920's, two hydroelectric dams have prevented anadromous salmon and trout from reaching over 90 percent of their historic spawning and rearing habitat; the result is diminished and extinct fish stocks and a dearth of nutrients once provided by decomposing salmon carcasses to the largest drainage within Olympic National park (NPS 1995). In addition, the dams have inundated important riverine and terrestrial habitat and trapped sediment within the reservoirs, resulting in the armoring of the river bed below the dams, reduction in the size and complexity of the estuary, and steepening of nearshore marine areas. Studies for the Elwha Report (DOI et al. 1994) and Environmental Impact Statements for dam removal indicate dam removal and sediment management are feasible and would provide good to excellent chances for full restoration of the anadromous fishery and the riverine ecosystem (NPS 1996). Restoration on smaller scales may not prevent the fragmentation and ultimate loss of native ecosystems; by addressing the entire system, the Elwha project offers a comprehensive restoration opportunity. In addition, most rivers suffer continuing habitat perturbations, whereas the one-time effort of removing the Elwha and Glines Canyon dams and management of the accumulated sediments will allow full restoration to occur.

Literature Cited

Department of the Interior (DOI), Department of Commerce, and Lower Elwha Klallam Tribe. 1994. The Elwha Report, Restoration of the Elwha River Ecosystem and Native Anadromous Fisheries.

National Park Service (NPS). 1995. Final Environmental Impact Statement, Elwha River Ecosystem Restoration. Olympic National Park, Port Angeles, WA.

National Park Service (NPS). 1996. Draft Environmental Impact Statement, Elwha River Ecosystem Restoration Implementation. Olympic National Park, Port Angeles, WA.

Winter, B. D., and R. M. Hughes. 1995. American Fisheries Society draft position statement on biodiversity. Fisheries 20(4):20-26.

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Estuarine Restoration: A Landscape Ecology Perspective

David K. Shreffler[1]
Ronald M. Thom[1]

Our central thesis is that landscape ecology principles provide the critical link between restoration ecology theory and effective, practical restoration in estuaries. We contend that recent efforts in Puget Sound to identify damaged natural resources through the Natural Resource Damage Assessment (NRDA) process could provide a unique opportunity to use landscape ecology principles to restore lost or degraded wetlands, as well as other habitats, in ecologically optimal locations within the estuary and watershed. In particular, we will demonstrate the need for greater emphasis in site selection and design on habitat size, shape, structure, buffers, accessibility, connectance, and self-maintenance to help ensure that restored sites are functionally and structurally integrated into the landscape. Using our landscape-based restoration planning procedure, the highest priority sites: (1) have potential for restoration to historic conditions, (2) are larger than 2 acres, (3) have adequate buffers, (4) are easily accessed by target species, (5) are connected to existing viable natural habitats, and (6) have a high probability of long-term self-maintenance.

Literature Cited

Shreffler, D. K. and R. M. Thom. 1993. Restoration of urban estuaries; new approaches for site location and design. Battelle Pacific Northwest Laboratories, Richland, WA.

Shreffler, D. K. and R. M. Thom. 1995. Estuarine Restoration: A Landscape Perspective. pp. 702-712 In Puget Sound Research. 1995 Proceedings. Puget Sound Water Quality Authority. Olympic, WA.

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Suitability Analysis for Enhancing Wildlife Habitat in the Yolo Basin

Joseph J. Donaldson[1]
Marcus Rawlings
Steve Chainey
Jay Stallman

Historically, California's Central Valley was a mosaic of wetlands, woodlands, grasslands, and riparian corridors that provided diverse and important habitat for wildlife, especially migratory birds. During the past 150 years, much of this valuable habitat has been lost or degraded by activities that converted the land to agricultural, residential, and other uses. Fueled by an intent to reverse a long-term decline in populations of water birds that winter in or migrate through the Central Valley, recent efforts have focused on restoring large portions of the formerly vast marshlands and other habitats in the southern Sacramento Valley. This presentation will describe these large-scale restoration efforts and focus on the development of a study to analyze the suitability for enhancing wildlife habitat in the 170-square-mile Yolo Basin near Sacramento, California. The study was requested in 1991 by a broad coalition of local landowners, conservation groups, state and federal agencies, and local elected officials. Its purpose is to support the creation of a diverse complex of historically important habitats in the Yolo Basin. Inherent in the study's purpose is the necessity for preserving and balancing agriculture and other land uses in the basin with the needs of wildlife and identifying implementation and incentive mechanisms (e.g., easement and cost-share programs) for private landowners. This presentation will describe the study's context and illustrate the approach and methods used to develop the suitability analysis. The study investigated historical and existing conditions, including soils, landforms, hydrology, vegetation, and agricultural activities and employed a geographic information system in the analysis of restoration suitability. The study may serve as a model for restoring and managing ecosystems in the California Central Valley and elsewhere.

Literature Cited

Central Valley Habitat Joint Venture. 1990. Central Valley Habitat Joint Venture Implementation Plan. Sacramento, CA. February, 1990.

Jones & Stokes Associates, Inc. 1990. Inventory of wetland and riparian habitats of Yolo County, California. Final Report. Prepared for Yolo County Community Development Agency, Woodland, CA.

Jones & Stokes Associates, Inc. 1994. Suitability analysis for enhancing wildlife habitat in the Yolo Basin, Sacramento, CA. Prepared for Central Valley Habitat Joint Venture, Sacramento, CA.

Yolo Basin Working Group. 1990. Yolo Basin wildlife area concept plan. Prepared for California Department of Fish and Game, Region II, Rancho Cordova, CA.

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Historical Analysis: A First Step in Stream Restoration

Katharine R. Grant[1]
Eddie Huckins

In 1990, the Nature Conservancy created the 1,200-acre Middle Fork John Day Preserve (MFJD) in eastern Oregon "to restore the cold water aquatic and riverine ecosystem to its natural conditions and functions." Federal and state restoration activities have highlighted the John Day River Basin because it lacks dams and hatcheries, making it one of the most important remaining wild salmon resources in the Columbia Basin. Historically, the MFJD provided some of the best spawning beds in the watershed, and today still supports small populations of spawning wild chinook, despite extensive channelization and habitat degradation. This project addressed the question, how has this 4-mile stretch of the MFJD river, and the river as a whole, changed since the turn of the century, and why? Analysis addressed four data sources: aerial photos, flow records, climate records, and oral history. While the complexity of watersheds and the inherent patchiness of historic data makes it difficult to draw firm conclusions about the past, this historic analysis provides information important in designing the long-term restoration of the riverine system.

Literature Cited

Larkin, Geri. 1990. Plant Community Survey at the Middle Fork of the John Day River Preserve. Prepared for The Nature Conservancy. Available through Oregon Field Office, Portland, OR.

Oregon Water Resources Department. 1991. Stream Restoration Program for the Middle Fork Subbasin of the John Day River. Prepared in cooperation with U. S. Bureau of Reclamation. Available from ORWRD, Canyon City, OR.

Ottersberg, R. J. 1992. Detailed Soil Survey of the Valley Floor of Dunstan Ranch, Middle Fork John Day Preserve. Prepared for The Nature Conservancy. Available through Oregon Field Office, Portland, OR.

Welcher, Karin E., 1993. Channel Restoration Plan and Geomorphology of the Middle Fork John Day Preserve. Prepared for The Nature Conservancy. Available through Oregon Field Offices, Portland, OR.

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Multi-Scale Planning for Riparian Woodland Restoration on Streams of the Western United States

Richard R. Harris[1]
Craig Olsen
Rebecca Johnson

Prioritization and planning of riparian woodland restoration requires data collection and analysis at three levels: the watershed as a whole, the stream reach, and the site. Landscape measurements obtained from topographic and land cover maps can be used to classify stream reaches within a watershed that have the greatest potential for restoration. Field sampling, within the priority reaches focuses on assessment of geomorphic and vegetation conditions relevant to restoration. Standardized cluster analysis is applied to the field sample data to distinguish groups of samples with similar structure and composition occurring on floodplain landforms. These groups are then further evaluated in relation to desired riparian ecological functions to establish targets for restoration management.

Literature Cited

Harris, R. R. and C. Olson. In press. Two-stage inventory and analysis for prioritizing riparian restoration at the stream reach and community scales. Restoration Ecology.

Harris, R. R., P. Hopkinson, S. McCaffrey and L. Huntsinger. In press. Use of geographical information systems versus manual techniques for map analysis in riparian restoration projects: a comparison. Journal of Soil and Water Conservation.

Olson, C. and R. R. Harris. In press. Applying two-stage inventory and analysis to prioritize riparian restoration at the San Luis Rey River, San Diego County, California. Restoration Ecology.

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Plant Species Diversity in a Semi-Arid Riparian Ecosystem in Southeastern Arizona

Keirith A. Snyder[1]
D. Phillip Guertin[1]
Peter F. Ffolliott[1]
Roy L. Jemison[2]

Knowledge of community structure and associated hydrologic characteristics is fundamental to the successful restoration and conservation of riparian ecosystems. High plant species diversity is often cited as one of the foremost attributes of riparian areas. Baseline data were collected from a small mid-elevation watershed in southeastern Arizona to evaluate differences in community structure and diversity between riparian areas and adjacent uplands, and to determine relevant environmental factors that influence plant community characteristics. Herbaceous and woody plant distribution and abundance was sampled in 192 quadrats located on transects that radiated away from a mixed perennial and intermittent stream. Principal Components Analysis (PCA) was used to display community patterns of species composition across environmental gradients. PCA indicated that patterns of woody plant composition varied with height above the primary stream channel and hydrologic regime, but herbaceous plant composition appeared unaffected by proximity to a riparian area. Woody plants comprised three distinct communities associated with differences in height above the primary stream channel: an upland, shrub-dominated *Mimosa biuncifera* community; a mesquite (*Prosopis velutina*) woodland community found primarily on secondary floodplain terraces; and a low-lying riparian community dominated by *Platanus wrightii*. A fixed effects analysis of variance model was used to determine whether calculated diversity indices varied between riparian areas and upland areas. Overall species richness, based on the combination of herbaceous and woody species sampled, was not consistently higher in low-lying riparian areas compared to upland areas. Alpha diversity indices (species richness (S), Shannon's Index (H'), and Simpson's Index (D)) calculated for woody plants demonstrated no consistent pattern; however, vertical structural diversity, measured by the relative density of woody species within various height classes, was highest near the stream. Similarly, herbaceous plant diversity was not consistently higher in the riparian areas. Herbaceous plants appeared to be more sensitive to overall topographical differences than to fluvial system linkages. Diversity was not higher in riparian areas than in the surrounding uplands for this semi-arid system. However, PCA demonstrated that the composition of these low-lying areas was distinct from that of the uplands, and vertical structural diversity was highest near the stream. Therefore, riparian areas make a unique and important contribution to overall landscape diversity.

Literature Cited

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

Jemison, R. L. and Ffolliott, P. F. (in press). Conditions that define a riparian zone in southeastern Arizona. Water Resources Bulletin.

Snyder, K. A. 1995. Patterns of plant species diversity and composition in a semi-arid riparian ecosystem. Unpublished M.S. thesis, University of Arizona, Tucson, AZ.

Snyder, K. A., Guertin, D. P., Jemison, R. L., and Ffolliott, P. F. (in review). Plant species composition in a semi-arid riparian ecosystem. Vegetatio.

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Manipulating the Disturbance Regime in a Southeast Missouri Bottomland Hardwood Forest Fragment to Promote Overstory Species Recruitment and Regeneration

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James W. Hormann[1]
Brad Larson
Michael Williams
Alan R. P. Journet

Big Oak Tree State Park occupies less than 450 hectares in Mississippi County, Southeast Missouri. Following extensive ditching, draining, and agricultural development, Southeast Missouri, once a vast expanse of bottomland hardwood forest containing a few scattered prairies, now contains virtually no representatives of these original ecological systems. Close to 8 percent of the park is an old growth forest fragment with a canopy dominated by oaks, hickories, pecan, and sweetgum. An adjacent area, subject to more extensive flooding, supports a canopy dominated by cypress. In addition, the park contains over a dozen state or national champion specimens of bottomland woody species. Studies of the structure and composition of these forest fragments consistently reveal a pattern through overstory and understory that indicates transition to a box-elder/sugarberry forest. It is hypothesized that maintenance of the current canopy composition, particularly the cypress and oak components, may require a management regime incorporating modified managed hydrology and disturbance.

As a result, a study is currently being conducted, in collaboration with Missouri's Department of Natural Resources, to explore the role of understory removal, and understory removal in combination with fire, in promoting recruitment of oak seedlings, and to determine the growth rates of nursery grown transplanted seedlings derived from local seed sources.

Initial results indicate that understory removal, in conjunction with fire, selectively removed "late successional" herbaceous and woody species, significantly increasing the amount of photosynthetically active radiation available to the forest floor. The tie required to measure regenerative rate in response to disturbance mandates further monitoring. Initial winter survival of two species of nursery grown transplanted seedlings in the park indicate no significant seedling predation after six months.

This presentation described the forest structure and composition, predictions regarding future unmanaged transitions, and the consequences of the initial phase of imposed fire disturbance.

Literature Cited

Abrams, M. D. 1992. Fire and the development of oak forests. Bioscience 42:346-353.

Hormann, J. W. 1995. Fire and restoration in the old-growth bottomland hardwood forest fragment of Big Oak Tree State Park. Master's Thesis. Southeast Missouri State University.

Lorimer, C. G. 1994. Tall understory vegetation as a factor in the poor development of oak seedlings beneath mature stands. Journal of Ecology 82:227-237.

Taylor, S. 1995. Spatial and temporal patterns in the bottomland hardwood forest of Big Oak Tree State Park. Master's Thesis. Southeast Missouri State University.

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Invasive Plant Species Distribution and Abundance in Freshwater Wetlands of Puget Sound Lowlands, King County, Washington

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Invasive plant species often hinder restoration efforts in freshwater wetlands. Selecting wetlands with a low potential for invasion and creating environmental conditions and community structures that discourage invasive species are critical components of successful wetland restoration projects. This study characterizes wetland environmental conditions and community structures that are dominated by invasive plant species. The distribution and abundance of invasive plant species were studied in 19 freshwater wetlands within the lowlands of King County, Washington. Patterns of invasive plant species distribution, dominance, and abundance were compared among and within wetlands and community types. Relationships between the degree of invasion and watershed condition and seasonal water levels were explored for invasive species as a group and individually.

Invasive plant species occur within all wetlands and community types in King County, Washington, but vary considerably in the number, frequency, and abundance among and within these habitats. Some of this variability was related to watershed condition and seasonal water levels. The frequency of invasives (as a group) was positively associated with the proportion of impervious area within a watershed; invasives were more common in urbanized watersheds. This relationship does not hold true for all invasive species, however. Some invasives (e.g., *Phalaris arundinaceae, Rubus discolor,* and *Solanum dulcamara)* were most abundant within wetlands in urbanized watersheds, while others (e.g., *Typha latifolia* and *Juncus effusus)* were generally in greater abundance within wetlands in less urbanized watersheds. Wetland hydrology, specifically the length of flooding, also was associated with the abundance of some invasives. *Typha latifolia* and *Juncus effusus* were more abundant under permanently flooded conditions, while *Rubus discolor* tended to inhabit sites with a limited to nonexistent flooding period. *Phalaris arundinacea* was a generalist, thriving in a range of hydrologic conditions.

These results suggest that invasion is influenced by life history attributes of the invader as well as specific environmental conditions of a particular site. Predicting or preventing invasion in wetland habitats should consider attributes of each potential invasive plant species as well as site characteristics.

Literature Cited

Ehrenfeld, J. G. and Schneider, J. P. 1991. *Chamaecyparisthyoides* wetlands and suburbanization: effects on hydrology, water quality, and plant community composition. Journal of Applied Ecology 28:467-490.

Houck, Catherine A. 1996. The distribution and abundance of invasive plant species in freshwater wetlands of the Puget Sound lowlands, King County, Washington. M. S. Thesis, University of Washington, Seattle, WA.

van der Valk A. G. 1981. Succession in wetlands: a Gleasonian approach. Ecology 62:688-696.

van der Valk, A. G., L. Squires, and C. H. Welling. 1994. Assessing the impacts of an increase in water level on wetland vegetation. Ecological Applications 4:525-534.

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Considerations for Restoring Structure, Function, and Diversity to an Ecosystem Colonized by Invasive Plants: A Phragmites Case Study

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The objective of our research is to identify ecosystem processes altered by the colonization of *Phragmites australis* in a disturbed fresh water marsh. Phragmites is an invasive rhizomatous reed grass that frequently forms a monoculture and inhibits the establishment of other plant species. Degraded ecosystems often result from multiple disturbance factors occurring at different temporal scales. When an invasive plant alters ecosystem-level processes such as nutrient cycling, then the structure and function of that system is impacted by that plant (Vitousek and Walker 1989). These factors may create legacy effects that hinder rehabilitation (Vogt et al. 1995, see table).

Cutting and burning can encourage vigorous regrowth of phragmites due to increased light availability and thus are not always effective restoration methods (Marks et al. 1994). Herbicide treatments can require repeated applications and affect non-target plant species as well (Ahearn et al. 1996). We hypothesize that the clonal stage of stand development needs consideration when attempting to restore a site dominated by *P. australis*. For example, a recently established stand can be eradicated by treatment with herbicide followed by cutting. In contrast, increased stand age and detrital accumulation may modify nutrient cycles reducing the availability of limiting nutrients to other plant species in the area. Phragmites invasion may leave legacy effects that make simple eradication an inadequate restoration technique. These legacy effects necessitate restoration of pre-colonization ecosystem cycles by accelerating rates of decomposition and regenerating desired vegetation.

We are investigating natural and anthropogenic factors that control ecosystem function and increase the competitive advantage of phragmites, including land use history, light and temperature regimes, changes in nutrient cycles, hydrology, alteration of the substrate, and legacy effects of invasive species. Our hypotheses are 1) it is insufficient to simply eradicate *P. australis*. Legacy effects created by colonization will hinder the reestablishment of desired species and the return to the pre-disturbance successional pathway, 2) *Phragmites australis* is modifying the site and causing it to become nutrient poor. While *P. australis* may tolerate lower nutrient availability, nutrients held in the phragmites detritus are unavailable to other plants due to slow rates of decomposition, 3) the effects of colonization by *P. australis* on structure and function change over time resulting in increased degradation as the site is occupied by phragmites.

We have established experimental plots and are applying treatments to identify the competitive mechanisms of *P. australis*, the effects of stand age, and the impacts of *P. australis* on nutrient pools and cycles. By using information collected at the ecosystem level to modify plant competition and restore ecosystem structure and function, we plan to develop methods to combat the general phenomenon of invasive clonal species.

Factors Contributing to Plant Invasions and Potential Direct and Indirect Ecosystem Effects

This table summarizes some of the factors that make ecosystems vulnerable to plant invasions. The direct and indirect effects outlined in the table below can all contribute to habitat degradation. However, some factors will have stronger impacts on habitat quality than others. In a phragmites dominated ecosystem, the importance of each effect

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Factors in plant invasions	Direct ecosystem effects	Indirect ecosystem effects
Anthropogenic disturbances: • Roads • Reservoirs • Pollution • Agriculture • Grazing • Forestry **Changes in intensity/frequency of natural disturbances:** • Suppression of fire/flooding • Dams **Changes in faunal populations:** • Introduction of exotic fauna • Reintroduction of extirpated species • Predator loss	Colonization by invasive plants Changes in community structure and faunal populations Decreased plant diversity Alteration of light and temperature regimes Detrital accumulation Modification of nutrient cycles Retention of heavy metals Modification of hydrology Alteration of soil structure Decreased regeneration of native plants/reduced food resources	Modified nutrient availability Carbon fixation, nutrient availability and nutrient pulses Changes in the composition and diversity of micro/macro flora, fauna, decomposers and plants Disrupt symbiotic relationships Loss of pollination/seed dispersal vectors Soil water relations Catastrophic fire/flooding

should vary depending on phragmites stand age. Identifying how the impacts of these variables change with time is central to our research and needs to be considered in any attempt to restore sites dominated by *P. australis*.

Literature Cited

Ahearn, L., Triol, J., Berlyn, G. 1996. The persistence of the herbicide glyphosate in soil. Unpublished paper presented at the annual conference for the Society of Ecological Restoration, Rutgers University, New Brunswick, NJ.

Marks, M., B. Lapin, and J. Randall. 1994. *Phragmites australis* (*P. communis*): Threats, Management, and Monitoring. Natural Areas Journal 14:285-294.

Vitousek, P. M. and L. R. Walker. 1989. Biological Invasion by *Myrica faya* in Hawaii: Plant Demography, Nitrogen Fixation, Ecosystem Effects. Ecological Monographs 59(3):247-265.

Vogt, K. A., D. J. Vogt, P. Boon, A. Covich, F. N. Scatena, H. Asbjornsen, J. L. O'Hara, J. Perez, T. G. Siccama, J. Bloomfield, J. F. Ranciato. 1996. Litter dynamics along stream, riparian and upslope areas following Hurricane Hugo, Luquillo Experimental Forest, Puerto Rico. Biotropica (In press).

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Biological Control of Purple Loosestrife

Bernd Blossey[1]

Purple loosestrife (*Lythrum salicaria*) is a wetland perennial introduced to North America from Europe in the early 19th century now occurring in large, monotypic stands throughout much of the temperate regions. Because conventional control techniques have proven ineffective, the development of a biological weed control program began with surveys and host specificity screening tests in Europe in 1986. Five-insect species demonstrated the host specificity required for introduction to North America. Four species are now successfully established. These are a root-mining weevil, *Hylobius transversovittatus,* attacking the main storage tissue of purple loosestrife; two leaf-beetles, *Galerucella calmariensis* and *G. pusilla* capable of completely defoliating individual plants and entire *L. salicaria* populations; and the flower feeding weevil *Nanophyes marmoratus.* Since 1992, about 400,000 leaf-beetles and 80,000 *Hylobius* eggs were shipped to 28 different states and Canada to collaborators in a wide range of organizations - many have started their own mass rearing programs. Increased attention is being given to follow-up studies to monitor target plant and control agent populations. A standardized monitoring guideline has been developed. This should allow the comparison of results across the entire distribution of *L. salicaria.*

Literature Cited

Blossey, B. 1993. Herbivory below ground and biological weed control: life history of a root-boring weevil on purple loosestrife. Oecologia 94:380-387.

Blossey, B., Schroeder, D, Hight, S. D. &. Malecki, R. A. 1994a. Host specificity and environmental impact of the weevil *Hylobius transversovittatus,* a biological control agent of purple loosestrife *(Lythrum salicaria).* Weed Science 42: 128-133.

Blossey B., Schroeder, D., Hight, S. D. & Malecki, R. A. 1994b. Host specificity and environmental impact of two leaf beetles *(Galerucella calmariensis* and *G. pusilla)* for the biological control of purple loosestrife *(Lythrum salicaria).* Weed Science 42:134-140.

Blossey, B. 1995. Coexistence of two competitors in the same fundamental niche. Distribution, adult phenology and oviposition. Oikos 74:225-234.

Blossey, B. & Schroeder, D. 1995. Host specificity of three potential biological weed control agents attacking flowers and seeds of *Lythrum salicaria* (purple loosestrife). Biological Control 5:47-53.

Hight, S. D, Blossey, B., Laing, J., & DeClerck-Floate, R. 1995 Establishment of insect biological control agents from Europe against *Lythrum salicaria* in North America. Environmental Entomology 24:967-977.

Malecki, R. A., Blossey, B., Hight, S. D., Schroeder, D., Kok, L. T. & Coulson, J. R. 1993. Biological control of purple loosestrife. Bioscience 43:480-486.

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Rodeo® Herbicide Use to Control Smooth Cordgrass (*Spartina alterniflora* L.) in Pacific Northwest Estuaries

Ron P. Crockett[1]

Smooth cordgrass (cordgrass; *Spartina alterniflora* L.) is an introduced perennial weed in the estuarine ecosystems of Washington State. Cordgrass has invaded bare mudflats and competes with desirable intertidal species. The presence of cordgrass in Willapa Bay and other coastal bays impacts recreation, commercial oyster culture, juvenile salmonid, and migratory waterfowl habitats. It also alters the hydrology and affects the accretion of fine sediment, raising the elevation of the mudflats. *Spartina* control efforts include mechanical treatments and the use of Rodeo® herbicide. Mowing and covering plants with plastic represent the mechanical options evaluated. Rodeo® herbicide applications using a variety of application techniques have been shown to be effective in controlling both the cordgrass leaves and dense fibrous rhizomes. Application rates of 3.75 quarts/acre broadcast as 2 to 5 percent v/v spray-to-wet, and 33 percent v/v solution through wiper equipment have all been shown to be effective in controlling cordgrass. Complete spray coverage and maximizing rain-free or tidal inundation intervals following treatment are important factors in obtaining consistent control. The environmental and toxicological characteristics of glyphosate the active ingredient in Rodeo®, are well suited to its use in these sensitive sites.

Rodeo® herbicide is a registered trademark of the Monsanto Company, St. Louis, MO 63167.

Literature Cited

Kilbride, K. M., F. L. Paveglio, and C. E. Grue. 1995. Control of smooth cordgrass with Rodeo® in a southwestern Washington estuary. Wildlife Society Bulletin 23:520-524.

Paveglio, F. L., K. M. Kilbridge, C. E. Grue, C. A. Simenstad, and K. L. Fresh. 1996. Use of Rodeo® and X-77® Spreader to control smooth cordgrass (*Spartina alternifiora*) in a southwestern Washington estuary: I. Environmental Fate. Environmental Toxicology and Chemistry 15:961-968.

Simenstad, C. A., J. R. Cordell, L. Tear, L. A. Weitkamp, F. L. Paveglio, K. M. Kilbride, K. L. Fresh, and C. E. Grue. 1996. Use of Rodeo® and X-77® Spreader to control smooth cordgrass (*Spartina alternifiora*) in a southwestern Washington estuary: II. Effects on benthic microflora and invertebrates. Environmental Toxicology and Chemistry 15:969-978.

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Integrated Pest Management Strategies for Reed Canarygrass (*Phalaris arundinacea* L.) in Seasonal Wetlands

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R. Bruce Wiseman[3]

Reed canarygrass (canarygrass; *Phalaris arundinacea* L.) is responsible for the reduction of biodiversity in wetlands nationwide. Dense monocultures of canarygrass displace open mudflats and out-complete native wetland plant species, reducing habitat value for aquatic migratory birds and other wetland-dependent wildlife. Although methods for control of canarygrass have been studied in the mid-West, little information is available regarding the effectiveness of intergraded pest management techniques in the Pacific Northwest. The objective of this study was to determine the effectiveness of Rodeo® herbicide (2.25 quarts/acre), disking, mowing, and herbicide-disking for the control of canarygrass at Ridgefield National Wildlife Refuge in southwestern Washington State. Preliminary findings from the 1994 treatments indicate that spring herbicide application combined with fall disking resulted in the greatest control of canarygrass. Other treatments, in order of declining effectiveness, were spring and fall application of Rodeo®; fall disking; fall Rodeo® application, and mowing. Vegetation response during the 1995 growing season will be discussed. Vegetation response to treatments will also be determined during the 1996 growing season. Evaluation of Rodeo® application rates, timing, and method of application, as well as surfactant type, for replicated small plot studies at the Ridgefield National Wildlife Refuge will also be presented.

Rodeo® herbicide is a registered trademark of the Monsanto Company, St. Louis, MO 63167.

Literature Cited

Paveglio, F. L., K. M. Kilbride, R. B. Wiseman, and R. P. Crockett. 1996. Integrated management techniques show promise for control of reed canarygrass (*Phalaris arundinacea)* in seasonal wetlands (Washington). Restoration and Management Notes 14:79-80.

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Bioengineering in Stream and River Restoration: Success of Enhancing Specific Functions Under Multiple Constraints

Marit Larson[1]
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Bioengineering in stream and river restoration is the use of living plants in combination with organic and inorganic material for re-constructing, stabilizing, and establishing new morphological and vegetative features. The proliferation of modern bioengineering techniques began about two decades ago in Western Europe, with the increasing concern over the loss of the ecological and hydrological functions of riparian systems. Initially, bioengineering was embraced as a technically acceptable alternative to concrete and riprap. Bioengineering also augmented natural riparian functions, and reestablished the appeal of a more naturalistic landscape. In Europe, in particular, channel revegetation techniques began to offer a controlled approach to enhancing elements of urban stream systems, which have been entirely disrupted.

Today, as our understanding of the full scope of challenges in actually restoring and sustaining the viability of stream ecosystem functions increases, the role of bioengineering bears more careful scrutiny. Stream enhancement projects using bioengineering should be examined for their contribution to restoration efforts. Bioengineering projects are often insufficiently monitored and lack baseline data, however, making their influence on habitat and the aquatic and wildlife community difficult to quantify. Consequently, the importance of bioengineering in meeting restoration goals often has to be evaluated using indicators that are available. The bioengineering projects we are discussing at this conference have been in place for various lengths of time in various locations. Their success and limitations are evaluated according to the goals identified for each project.

The projects or case studies explored here have three common themes. First, most of these streams are in urbanized watersheds, have undergone extreme modification, and have problems that would normally be addressed with conventional engineering structures. Second, these projects are implemented on a limited section of the river system and are not accompanied by comprehensive watershed restoration. Third, these projects are constrained by space, time, and funding. For these reasons, restoration goals are confined to specific features of the stream system. The projects discussed include: Wedel Brook near Hamburg and the Enz River near Pforzhelm in Germany, and the North River in Massachusetts. Each project had different goals and succeeded in achieving these goals to varying degrees.

Several conclusions can be drawn from all three of these projects, however. First, the projects demonstrated that erosion control through vegetation and vegetable materials can be successful. Second, each project underscores the need to clearly identify the problems, and to incorporate an understanding of channel geomorphology into the applications of bioengineering principles. Third, the projects successfully focused attention on rivers or sections of rivers that are frequently marginalized as resources. By elevating the importance of waterways in even urbanized or constrained settings, we have a greater chance of modifying our actions that impact these resources. Since an increasing number of streams and rivers are subject to the constraints facing the streams in these projects, we must continue to evaluate the effectiveness of bioengineering approaches to stream and river restoration.

Literature Cited

Allen, H. H. and C. V. Klimas. 1986. Reservoir Shoreline Guidelines. Waterways Experiment

[1] The Bioengineering Group, Inc., 53 Mason Street, Salem, Massachusetts 01970.
[2] Bestmann Green Systems. Inc., Salem, Massachusetts 01970.

Station, U.S. Army Corps of Engineers. Technical Report E-86-13, Vicksburg, MI.

Bestmann Green Systems, Inc. 1993. North River Streambank Stabilization Bioengineering Demonstration Project, First Year Monitoring Report. U. S. Army Corps of Engineers. Waterways Experiment station, Vicksberg, MS.

Carey, M. 1996. Effective low cost streambank bioengineering techniques for the landowner. International Erosion Control Association Conference Proceeding, Seattle, WA. pp. 169-175.

Coppin, N. J. and I. G. Richards. (eds.) 1990. Use of vegetation in civil engineering. Butterworths, London.

Gore, J. A. and F. D. Shields. 1995. Can Large Rivers be Restored? Bioscience 45(3):142-152.

Gray, D. H. and A. T. Leiser. 1989. Biotechnical Slope Protection and Erosion Control. Robert E. Krieger Publishing Co. Malabar, FL.

Groeneveld, D. P. and R. H. French. 1995. Hydrodynamic control of an emergent aquatic plant (*Scirpus acutus*) in open channels. Water Resources Bulletin 31(3):505-514.

Humborg, G. 1995. Typologische und morphologische Untersuchungen an Bergbaechen im Buntsandstein-Odenwald. (Classification and morphological research of mountain streams in the Sandstone-Oden Forest,) Institute for Hydraulic and Civil Engineering, University of Karlsruhe, Germany. Vol. 192.

Larson, M. G. 1995. Developments in River and Stream Restoration in Germany. Restoration Management Notes. Journal of the Society for Ecological Restoration 13(1):77-84.

Miller, D. E. 1996. Design guidelines for bioengineering river bank stabilization. International Erosion Control Association Conference Proceedings, Seattle, WA. pp. 157-168.

Nadolny, I. 1994. Morphologie und Hydrologic naturnahe Flachlandbaeche unter gewaessertypologischeen Gesichtspunkten. (Morphology and hydrology in the classification of natural lowland streams,) Institute for Hydraulic and Civil Engineering, University of Karlsruhe, Germany. Vol. 189.

Peirce, G. J. 1994. Adaptive Modes in Wetland Plants, A preliminary Review. Southern Tier Consulting, Inc., P.O. Box 30, West Clarksville, NY 14786.

Rotar, M. A. and J. T. Windell. 1996. Innovative bioengineering techniques used to restore Boulder Creek, CO. International Erosion Control Association Conference Proceedings, Seattle, WA. pp. 55-64.

Schiechlt, H. M. and R. Stern. 1994. Handbuch fuer naturmahen Wasserbau. Oesterreichischer Agrarverlag. Vienne, Austria.

Schiechtl, H. M. 1980. Bioengineering for land reclamation and conservation. The University of Alberta Press.

Thomas, M. R. and R. E. Smiley. 1995. Stream channel stabilization using soil bioengineering - a case history. International Erosion Control Association. Conference Proceedings. Atlanta, GA. pp. 63-73.

Thomas, M. R. and R. E. Smiley. 1996. Stream channel stabilization using soil bioengineering. International Erosion Control Association Conference Proceedings, Seattle, WA. pp. 65-71.

U. S. Army Corps of Engineers (ACOE). 1987. Water Resources Development in Massachusetts. New England Division.

Wraith, J. G. 1996. Tannery Creek - Bioengineering stream realignment. International Erosion Control Association Conference Proceedings, Seattle, WA.

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Planting Deep Willow Cuttings via Hydraulic Jetting, for Riparian Erosion Control and Habitat Enhancement

Lon Drake[1]
Rick Langel[1]

Planting willow cuttings for streambank erosion control is an ancient technology, revived in recent decades as part of "bioengineering." Problems include difficulties with emplacement, while slumping can easily undermine the short cuttings normally used. We have implemented a new hydraulic jetting technique, which can rapidly (20 to 30 seconds) open planting holes in streambank alluvium, which are 6 feet deep by 1 to 2 inches diameter. Slender dormant willow cuttings pushed into these holes extend to damp soil and can sprout roots their entire below-ground length. The technology consists of a specially machined jetting tip welded to a 6-foot length of ordinary ½-inch steel pipe. Two of these jetting pipes are connected to a portable high-pressure pump, drawing water from the stream and delivering it to the jets via sturdy garden hoses. A planting team of 4 is most efficient, with 2 persons jetting holes and 2 planting willows. At favorable locations, a team of 4 can harvest 1,000 to 1,500 slender willow cuttings per day and plant 1,000 cuttings per day. All the equipment is portable and can be carried on muddy stream banks. The jetting equipment plus 1,000 bundled willow cuttings fit into a pickup truck.

The hydraulic jetting works well in most poorly sorted alluvial sands, silts, and fine gravels, including those saturated with groundwater. Even under quicksand conditions, the added water pumps up the hydraulic head so that the jetted holes remain open long enough to push a cutting down 6 feet. We have also jetted through rotten logs and can flush stones 1 inch in diameter up out of the hole. The technique does not work well in dense clayey glacial till, stoney gravel or coarse clean sands.

On the sandy eroding cutbanks of Clear Creek, Iowa, in 1994, we planted approximately 4,000 sandbar willow cuttings. At the end of the first growing season, we had a 98 percent survival rate for those planted below the terrace top. However, the banks were left in a vertical position and during the floods of 1995 from this flashy 88-square-mile watershed, the cutbanks continued to collapse and damaged portions of this planting. In spring 1995, we planted 2,000 willows along a broad curving stretch of Hare Creek, Iowa, in silty colluvium. The Hare Creek watershed is very flashy but only 4 square miles in area. We excavated the vertical banks back to a 1:1 slope before planting and broadcast seeded them with oats and timothy grass. At least 7 floods passed through within 2 months after planting but willow survival was about 96 percent for the season and erosion was hardly measurable. Test plantings in 1996 include red osier dogwood cuttings for enhanced habitat value. Caveat: before they become established, willow cuttings can be drowned by prolonged flooding. They do not root well below the permanent water table and should not be planted in the flowing water of streams at baseflow level.

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Revegetation at La Franchi Demonstration Wetland, Santa Rosa, California

Marco Waaland[1]

Abstract: **Tule (*Scirpus californicus*) plants at La Franchi demonstration wetland have been monitored for growth parameters (number of stems per tule clump, area of marsh zone occupied by tule clump) after transplantation into three separate marsh zones in winter of 1990. Twenty-nine of the 30 plants in the sample survived. Initial stem numbers and clump area were not measured. However, it is unlikely stem number exceeded seven and clump area exceeded 0.5 ft^2. By fall of 1991, mean stem density was 121 stems/plant and the mean area occupied by individual tule clumps was 7.6 ft^2. As expected, a significant correlation exists between the number of tule stems and the area occupied per tule clump. By projecting the relationship between increase in tule clump area over time, it was possible to estimate that the vegetation at the demonstration wetland would occupy the entire marsh zone approximately 1.5 years after transplantation. At that time, the full animal waste treatment ability of the wetland would be attained because the surface area of the tule stems and oxygen transfer to the soil via the root system would be maximized.**

Introduction

The dairy industry is the largest user of the Santa Rosa Subregional Water Reclamation System's reclaimed water. However, its viability may be threatened by the costs associated with animal waste control water quality compliance. Animal waste control is an important element of the existing reclamation system and was selected by the City of Santa Rosa for a long-term reclamation project.

The potential loss of any dairies in the Laguna has a negative effect on the ability of the industry to survive, and has potentially negative implications for the Subregional System's irrigation reuse program. Constructed treatment wetlands (i.e., freshwater marsh systems) have been shown to successfully treat animal waste to acceptable levels (Maddox and Kingsley, 1989). Wetland wildlife also benefit from increased marsh habitat acreage. The Subregional System has undertaken this animal waste control demonstration project to provide a local example of management practices that can be used on dairies to meet water quality objectives. This paper presents preliminary findings on the development of the planted vegetation of the wetland.

Physical Layout

The wetland consists of three vegetated zones, separated by deeper open water sections. Operating water depth for the marsh zones was designed to be 1.5 feet. Topsoil (Huichica loam) was stockpiled during construction and subsequently spread in a layer one-foot thick in the marsh zones to provide a planting substrate. The dimensions of each planting zone were approximately 45 feet by 50 feet, for a total planting area of 6,750 ft^2.

Revegetation Scheme

A total of 420 California tule (*Scirpus californicus*) plugs were transplanted into the three marsh zones in rows. The tule source area was a freshwater marsh along Roseland Creek, bordering Sonoma County Water Agency mitigation area at Alpha Farm. Tule plugs were dug from the soil during the fall and winter of 1990. Plugs consisted of rootwads and several live stems which were kept intact and transported by truck to La Franchi's and transplanted the same day. Individual rootwads were planted on staggered four foot centers along each row. Tules were transplanted into wet or flooded soils at the demonstration wetland on the following dates: Zone 1, 12/6/90; Zone 2, 11/18/90; and Zone 3, 11/12/90.

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Data Analysis and Monitoring Methods

A sampling scheme involving 30 randomly selected tule plant individuals was developed to reflect performance of the La Franchi tule population. Water depth, tallest stem, total number of live stems per clump and individual plant (clump) circumference were measured. Individual tule plant circumference was measured at the water surface and converted to a cross-sectional[2] area. This parameter is referred to as the "area of marsh zone occupied by tule clumps." In the course of the study, one of the selected sampling plants died and was removed from regression analysis. The statistical package "Statgraphics" was utilized to perform multifactor ANOVA and simple regressions. The regressions lumped together all repeated observations of stem number and clump area to determine the degree of correlation.

The data were transformed by natural log for the regression analysis. The ANOVA sorted the data by season to estimate stand growth trend, and by zone to determine sources of any variation that may be explained by spatial differences. Monitoring dates were: 6/6, 8/7, and 11/11, 1991.

Results

The growth response of the 30 sample plants is shown in Figure 1. The three discrete distributions represent the spring, summer and fall monitoring events. The number of stems per tule clump is arrayed along the Y-axis and the area occupied per tule clump is arrayed along the Z-axis. The X-axis represents the number of days elapsed since the plants were initially transplanted. The increase in stem number and clump area are discussed further below.

Of the 30 plants subjected to monitoring, only one died; a 97 percent survival rate. This result demonstrates the resiliency of the California tule to uprooting and transplantation shock.

There was a significant increase in number of stems per tule clump over time. The original number of live stems is not known, but it is unlikely that any clump exceeded seven live stems, based on the clump size that could be handled by one person with a shovel. Figure 1 shows that mean number of stems increased from 14.7 in spring, to 62.1 in summer and 121 by fall. By fall, there were significantly fewer live stems in Cell 3 than either Cells 1 or 2, despite the fact that Cell 3 was planted first and had three more weeks of growth than the other cells.

The mean area of marsh zone occupied by individual tule clumps increased over time from 0.9 ft^2 in spring, to 4.9 ft^2 in summer, and 7.6 ft^2 by fall. Unlike stem growth, there was no significant difference in area occupied by tule clumps between the three separate marsh zones.

Discussion

Tule Growth Dynamics

The 97 percent survival rate exhibited by the California tule transplanted at the La Franchi Demonstration Wetland demonstrates a high degree of success. Corollary studies underway at the Kelly Farm Demonstration Wetland also report high tule survival rates of 78 percent and 94 percent (CH2M HILL et al, in prep). It is apparent that the California tule is a hardy species able to withstand the injury and shock of transplantation.

The transplanted tules spread across the revegetated marsh zones at a rapid rate. The constructed planting area set aside for tule marsh totaled approximately 6,750 ft^2. In fall 1991, the area occupied by all the tule clumps was estimated at 3,990 ft^2 (7.6 ft^2 x 420 plants). Therefore, by fall, over 50 percent of the area of all three marsh zones was occupied by tule plants.

Another means of assessing the development of marsh at the La Franchi Demonstration Wetland is to compare the overall stem density within marshes at different stages of development. Overall density of tule stems within a defined area of marsh can vary from 0.5 stems/ft^2 in sparsely vegetated, recently created marsh, such as the Subregional System's Kelly Farm Demonstration Wetland (CH2M HILL et al, in prep.), or exceed 15 stems/ft^2, as found in mature natural marsh growing in the Central Valley where tules occupy the entire marsh area (Fox 1989). At the time of the most recent La Franchi wetland monitoring (11/11/91), the average density of tule stems in

[2]This estimate assumes stem growth for each plant expands in a geometrically round form, fitting the simple relationship: area = πr^2.

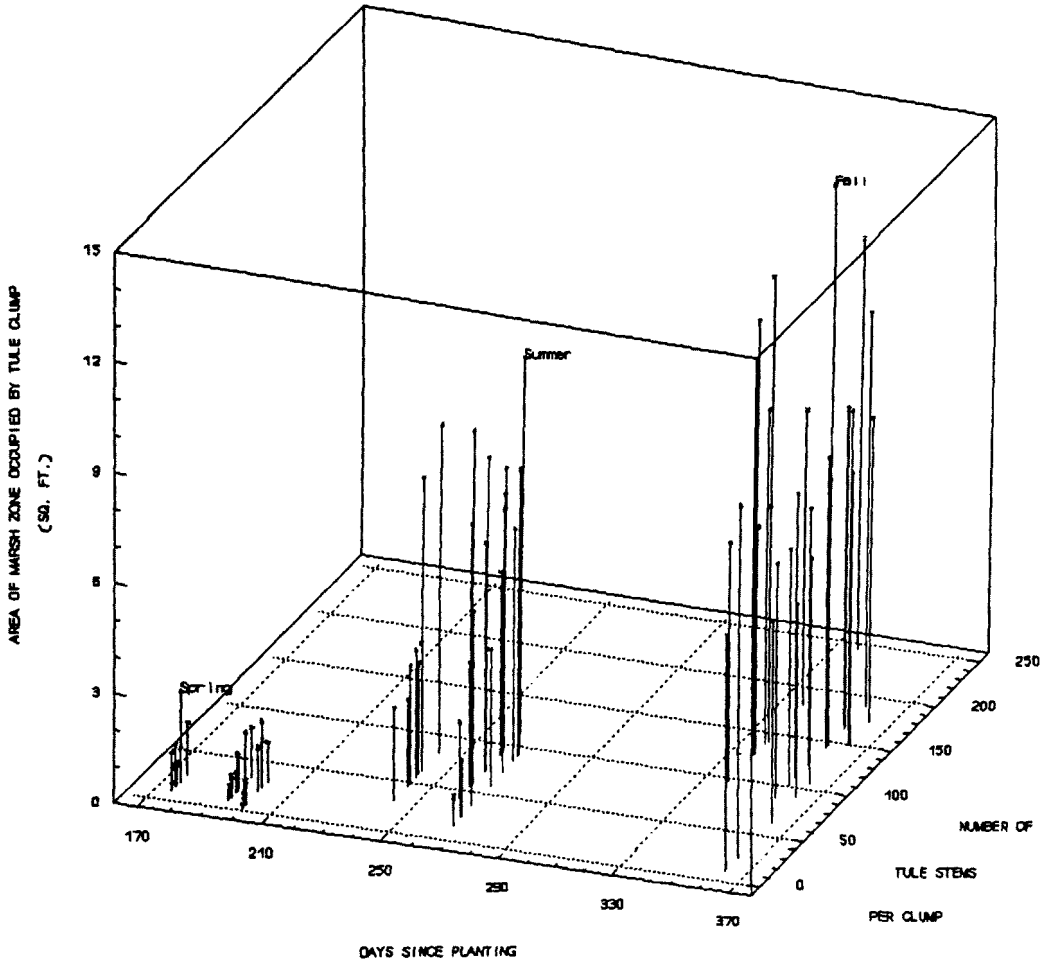

Figure 1. Increases in Area of Marsh Zone Occupied by Tule Clumps and Number of Stems at La Franchi Demonstration Wetland.

the marsh zone was: 7.5 stems/ft^2 [(121 stems/plant x 420 plants) ÷ 6,750 ft^2]. If tule stem density in mature marsh can reach up to 15 stems/ft^2, then the most recent estimate of 7.5 stems/ft^2 is 50 percent that of mature marsh.

Both the area of marsh zone occupied by tule clumps (50 percent) and percent stem density compared to mature marsh (50 percent) indicate a marsh in a youthful stage of development. However, it is also a marsh in a phase of rapid vegetation growth. As would be expected, the number of stems per tule clump and the area of marsh zone occupied by tule clumps are highly correlated. This log-linear relationship shows a rapid rate of tule clump expansion as the transplanted stems take root and begin sending up new sprouts. As more stems are added the rate of expansion increases dramatically.

For the existing tule clumps to expand to the extent that they fully occupied the entire area of the three marsh zones, they would have to increase in mean area occupied, from the current 7.5 ft^2 to a mean of 16 ft^2. A regression of the data for clump area against time elapsed since planting indicates a significant correlation. By using this limited data set it is possible to extrapolate the regression to determine approximately when the mean clump area would reach 16 ft^2. The regression model predicts that the mean clump area would reach the maximum of 16 ft^2 after approximately 560 days, or 1.5 years after planting. Within 95 percent confidence intervals, the actual time could range from 1.0 to 3 years. At this time, the full animal waste treatment ability of the wetland will be attained because the surface area of the tule stems and oxygen transfer to the soil via the root system will be maximized.

It must be stressed that this is only an estimate based on a very limited data set. This model assumes that the growth response will continue in a linear fashion. It is possible that the initial growth rate is exponential but tapers off before the marsh zones were entirely occupied. This decline in growth rate would be a natural result of competitive interactions between the individual tule clumps for increasingly limited resources such as space, light, and nutrients.

Future studies should look at comparative rates of spread amongst differently spaced planting centers. Also, studies of individual tule clumps would allow observation of whether there is a self limiting maximum to the numbers of new shoots individual plants can develop.

Related Benefits

Monitoring at a second created wetland showed a rapid development of wildlife habitat. Seventy-eight species of birds have been observed. The total number of birds increased by 38 percent within one year after revegetation, along with an increase in nesting species. A large aquatic prey base (crayfish, frogs, and mosquitofish) has resulted in colonization by river otters and mink. The development of dense marsh vegetation has enhanced the nutrient removal capability of the marsh. Orthophosphate reduction averages 33 percent, despite large numbers of waterfowl. Total inorganic nitrogen was essentially removed (94-99 percent), and TKN was reduced by 80 percent.

Acknowledgements

Carolyn Dixon, natural resources manager for the Subregional System, has been closely involved with the La Franchi project. She assisted in the planting and conducts the routine marsh monitoring. Kristina Sloop also deserves credit for her assistance in all phases of this project.

Literature Cited

CH2M HILL, D. W. Smith, R. Williamson, and M. E. Waaland. In Preparation. Kelly Farm Demonstration Wetland: Management Plan and Monitoring Results. Prepared for Santa Rosa Subregional Water Reclamation System.

Fox, J. P. 1989 Evapotranspiration of macrophytes in California's wetlands. Unpublished report prepared for San Francisco Bay/ Delta Hearings.

Maddox, J. J. and J. B. Kingsley. 1989. Waste treatment for confined swine with an integrated artificial wetland and aquaculture system. In: D.A. Hammer (Ed.) Constructed Wetlands for Wastewater Treatment. Lewis Publishers, Inc. Chelsea, MI. 831 pp.

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Lessons Learned in Habitat Restoration, Comparison of Two Wetland Mitigation Projects in San Diego, California

John L. Minchin[1]

Abstract: **This paper addresses design and installation issues associated with large scale riparian mitigation projects in San Diego County, California.**

Introduction

Over the past 6 years Ogden's habitat restoration team has been involved in well over 100 wetland mitigation projects within San Diego County. Two of the major projects accomplished using this period include, the Twin Oaks Valley Ranch project, a 308-acre master planned recreational community in San Marcos, in northern San Diego County; and the Allred-Collins Business Park project, a 140-acre planned industrial development in San Diego. Both projects involved complex hydrological constraints, diverse mitigation requirements, and multidisciplinary coordination with consultant team members to bring about ultimate success. Both projects are similar in that Ogden staff were involved from start of finish including initial biological evaluation, design development and construction document preparation, installation monitoring, and long-term biological monitoring. As a result, much has been learned in the process.

The purpose of this presentation is to compare the two projects and outline some of the lessons learned in implementing their mitigation programs. This should help point out key issues to consider when planning similar habitat restoration projects, for the benefit of other professionals involved in this type of work.

Twin Oaks Valley Ranch

Twin Oaks Valley Ranch project included biological analysis of existing vegetation; wetland delineations and impact evaluation; conceptual wetland mitigation plan preparation; permit processing with regulatory agencies; coordination with civil engineers for alternative flood channel design; assistance to landscape architects and golf course architects for construction document preparation; construction implementation monitoring; and long-term mitigation monitoring.

The Major Project Challenges at Twin Oaks Included:

- Interfacing with project team consultants to develop an integrated mitigation scheme which would to work around the development constraints of the 644 home residential community and 18-hole golf course.
- Interfacing with recreational amenities including edge transitions from the native habitats to the golf course, and incorporating equestrian and pedestrian trail access.
- Analysis of development impacts and developing compensatory mitigation measures acceptable to the regulatory agencies.
- Evaluation of flood control and hydrologic concerns, developing flood channel treatment alternatives, and developing strategies for check dam construction.
- Coordination of wetland mitigation requirements with permitting and reviewing agencies, and convincing them to accept mitigation strategies with the incorporation of recreational amenities.
- Developing appropriate vegetation associations for the revegetation effort to meet the mitigation goals.

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Allred-Collins Business Park

Allred-Collins Business Park project included biological surveys of existing vegetation; evaluation of canyon hydrology, both before and after construction, to verify suitable conditions for plant material establishment; conceptual wetland mitigation plan preparation; regulatory permit processing; coordination with project engineers and contractors on canyon grading and check dam construction; assistance to the landscape architects on conceptual plan preparation; construction implementation monitoring; and long-term mitigation monitoring.

The Major Project Challenges at Allred-Collins Included:

- Evaluation of flood control/hydrologic concerns to verify suitability for vegetation.
- Incorporating strategies for check dam construction to slow down water and alter the canyon bottom gradient to accommodate an expanded riparian corridor.
- Coordination of wetland mitigation requirements with permitting/reviewing agencies.
- Developing appropriate wetland vegetation associations for the revegetation effort.
- Incorporating recreational park facilities and pedestrian trail systems.
- Working around the industrial park development constraints and evaluating drainage outfall.
- Addressing long-term brush management issues along perimeter of the open space preserve.
- Coordination of grading and check dam construction.
- Intensive field spotting of plant materials and hands-on coordination with contractors during construction.
- Coordination with maintenance contractors during long-term maintenance/monitoring period.

Key Issues Summary and Conclusions:

Based on experience gained during these two large mitigation projects, we believe the issues listed below should be given key consideration in wetland habitat restoration design.

A. Analyze your project site thoroughly including: soils, hydrology (before and after construction), climatic considerations, sun exposure, slopes, and gradients.

B. Inventory all biological resources including :

- plant communities
- sensitive plants and wildlife
- endangered species.

C. Evaluate project development plans and meet with other project team members to thoroughly understand development strategy and scheduling.

D. Evaluate impacts including:

- Direct impacts
- Indirect impacts
- Additional construction access issues and staging areas
- Buffer zones
- Future fire/brush management zones and limits of vegetation clearing
- Possible preservation issues
- Designation of open space parcels
- Future utilities easements.

E. Establish resource agency and local jurisdiction context:

- Appropriate agency review for type of impacts/mitigation required
- Evaluate permitting/processing timeline
- Sequence of review with regulatory agencies.

F. Evaluate mitigation requirements, strategies, and options. Anticipate acceptable mitigation ratios for agency concurrence and provide justifications for ratios being proposed.

G. Evaluate site potentials for type of mitigation required. (Are adequate factors available to support intended mitigation?):

- Soils
- Hydrology and depth to water table
- Sufficient acreage available
- Adjacency of native habitat and potential wildlife/habitat linkages.

H. Explore off-site mitigation alternatives.

I. Develop conceptual mitigation plan program.

J. Develop detailed working drawing/construction documents for bidding; make sure specifications are adequate (i.e., present the installation

goals, delineate maintenance requirements, and establish monitoring responsibilities).

K. Order plants well ahead of anticipated installation schedule, at least 9 months to 1 year in advance. Anticipate special collections, local sources, and realistic implementation schedules. Build in contingencies should delays be incurred.

L. Coordinate with general contractor and landscape contractor for installation:

- Explain environmental constraints
- Help develop implementation strategy/ schedule
- Set up site observation/monitoring schedule at key project milestones
- Monitory actual installations.

M. Coordinate final construction acceptance/ maintenance turnover.

N. Coordinate with maintenance contractor for long-term maintenance/monitoring.

O. Conduct actual monitoring:

- Horticultural
- Botanical.

P. Coordinate final turnover at end of maintenance/monitoring period:

- Client
- Local jurisdictions
- Resource agencies.

Q. Follow up with evaluation of success and lessons learned for future projects.

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Wetland Functional Relationships: A Landscape Ecology Perspective

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The functional attributes of wetlands are not solely the result of internal wetland characteristics. Many of the functional values we assign to wetlands are the outcome of processes originating some distance from the wetland edge. Failure to consider these outside influences has often resulted in ineffective protection and restoration programs. In order to understand wetland functions more clearly, it is necessary to examine the relationship of wetlands to the surrounding landscape. Landscape ecology focuses on three characteristics of landscapes: structure–the spatial relationship among elements; function–interactions among the spatial elements; and process–the alteration of structure and function over time. It therefore provides a useful framework for examining wetland functions as they are linked to the greater landscape. Employing the principles of landscape ecology, we may ask how wetland functions are coupled to landscapes and influenced by the structure, function and processes that occur outside and often distant from wetland boundaries.

Assessing Wetland Condition and Restoration Potential Within Watersheds Using a Performance Criteria Matrix

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Assessing the condition and the restoration potential for wetlands and their associated stream reaches within watersheds is an appealing concept, both ecologically and politically. Yet, issues of scale continue to plague attempts at measuring how individual wetlands functionally contribute to watershed health, how the wetlands themselves are impacted cumulatively, and where to effectively target locations for restoration and creation projects. We are recommending use of a tailored HGM and disturbance classification system coupled with a performance matrix to: 1) describe the variability across wetland types, 2) improve site selection procedures for created and restored wetlands, and 3) predict the successional trajectories of natural reference wetlands, and creation and restoration projects. We recommend that the sources of information used to develop regional classification schemes and performance matrix be obtained from detailed studies of reference wetlands within a region. Intensive studies of the site characteristics of reference wetlands and created projects, conducted by the Penn State Cooperative Wetlands Center (CWC), have greatly increased our understanding of wetlands in their respective landscape positions. We have shown that natural wetlands classified into five categories (receptor, conveyor, donor, fringe, and beaver-impounded) differ in their respective soil characteristics, plant communities, hydrologic and water quality signatures, sedimentation rates, and habitat potential. In addition, comparisons between reference wetlands and creation projects show distinct differences in these attributes. These findings can be used to define expectations regarding how natural wetlands will respond to impacts and to determine whether creation projects are performing their expected functions. We are extrapolating these site characteristics to a larger scale to establish synoptic criteria to screen portions of watersheds regarding their condition and the need for restoration. We propose that these same criteria can be used to build a performance matrix, arranged by HGM and disturbances classes; based on the best information available. The strength of this approach is that the reliability and predictability of assessments of wetland condition are expected to increase as we move from best professional judgment to testable hypotheses.

Wetland Restoration Success Following Pipeline Construction

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Construction of natural gas pipelines has affected hundreds of acres of wetlands throughout the Pacific Northwest. Federal regulations require pipeline companies to restore wetland habitats to their original contours and vegetated condition following construction. However, criteria for evaluating

restoration success are usually poorly defined or focused on a single parameter such as plant coverage. In a study that examined conditions in 120 wetlands one year following pipeline construction, investigators assessed the degree of restoration success relative to three main criteria: total vegetative cover, species diversity, and average species indicator status (according to Reed 1994, 1988). Data concerning the percent of native and introduced species and the survival of planted woody stems were also compiled. Minimum success thresholds were established for each criterion and a data indexing system combined data for each criterion into a single value for each wetland. Pre- and post-construction species lists were compared and the effects of reseeding were examined. Cover in the majority of the wetlands studied exceeded 80 percent and most wetlands supported a dominance of native hydrophytes. Species diversity was typically higher following construction. Reseeding efforts appeared to have been unsuccessful as the species used in restoration seed mixtures were poorly represented in post-construction plant communities. Establishing vegetation appeared to have derived from soil seed banks and/or adjacent seed sources.

Hydrologic and Ecological Modeling and Geographic Information Systems for Predicting Floodplain Forest Restoration Following Dam Removal

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Significant areas of floodplain forests are frequently lost when rivers are impounded by dams. Rodman Dam, a 2,073 m long and 6.7 m high earthen-fill dam on the Ocklawaha River in Florida, is proposed for removal. This dam inundated or severely stressed approximately 2,924 ha of floodplain forest on this canopied, blackwater river. An important component of the environmental assessment research designed to assist in the reservoir retention or river restoration decision-making process was the prediction of floodplain forest restoration if the dam were removed. Floodplain forest restoration following dam removal was predicted using the U.S. Fish and Wildlife Service's FORFLO forest succession simulation model, designed to predict the impacts of altered hydrologic regimes on floodplain forests. FORFLO is a stochastic, individual-based computer model simulating the growth, reproduction and death of each tree in a test plot. The primary variables in FORFLO affecting forest succession are soil type, growing degree days, and river flow regime. Water stage statistics in the river and floodplain with the dam removed were predicted using HEC-1 and HEC-2 simulation models. Bathymetry of the river and floodplain were also measured, digitized, and entered into GIS. FORFLO was used to predict the community dominants within elevational plots across five floodplain transects along the 25.7 km longitudinal axis of the reservoir. The FORFLO predictions were entered into GIS as another data layer over bathymetry, with maps of predicted floodplain habitats produced at intervals through 200 yr of forest succession. Modeling results demonstrated that the floodplain forest canopy would return in about 40 yr after dam removal, occupying about 3,007 ha of the floodplain, only slightly greater than the 2,924 ha of floodplain forest occupying the study area prior to dam construction. The use of GIS maps was an effective method to simplify and present these floodplain forest succession predictions to non-technical decisionmakers.

Restoring Hydrologic, Geomorphic, and Ecologic Functions in the Big Quilcene River Delta

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The Big Quilcene River is a steep, gravel bed stream, which provides spawning and rearing habitat for a Washington State listed species of early run chum salmon. This habitat has been degraded by alterations in the hydrologic functions of the fluvial system that result from disturbances to its physical and ecological structures, including logging in the upper watershed, water diversions, urbanization, levee construction, and periodic removal of gravel from the channel bed. Impacts to spawning habitat include burial of redds by excess sediment, washout of reeds at high flows, dewatering reeds by channel migration, and removal of gravel from the channel in an effort to reduce flood risks. This paper

describes a project designed to enhance chum habitat on the Big Quilcene River delta by restoring the structures of the fluvial system that create and sustain anadromous fish habitat. The project includes removal of levees in the lower 0.5 river mile to reconnect the river channel with the adjacent floodplain and marshes. Improvements in hydrologic functions include reduced flow depths, and more appropriate distribution of conveyance of water and sediment between the channel and the floodplain. Benefits to fish habitat include lower shear stresses and reduced gravel bed mobility, less sediment deposition in the channel and more on the floodplain, and increased in-channel structure and habitat diversity. Additional benefits include reduced flood hazards and rejuvenation of floodplain wetland channels.

Geographic Information Systems Applied to Selection of Forested Wetland Restoration Sites

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Restoration practitioners have typically depended upon pre-existing wetland inventories, such the National Wetlands Inventory (NWI), to locate opportunities for off-site restoration. However, mapping techniques employed to develop this inventory are limited in their ability to differentiate forested wetlands from other forest communities. As a result, identifying sites for forested wetland restoration often involves extensive ground exploration or renewed inventory efforts, either of which can become costly and time-consuming undertakings. The rapidly growing volume of available digital data and increasing access to Geographic Information System (GIS) software offer opportunities for streamlining forested wetland restoration site selection. It is now not only possible, but cost-effective, to focus field exploration by generating digital maps to indicate probable areas of degraded forested wetland.

This paper describes a pilot study carried out to identify suitable sites for restoration to compensate wetland impacts from a planned mining operation in the Okanogan Highlands of eastern Washington. Analysis of the 100-square-mile study area was based entirely upon existing, publicly available datasets, including vegetation maps, soil maps, surface hydrology maps, and Digital Elevation Models. Mapped environmental variables were analyzed using raster GIS software to identify statistically significant spatial correlation with the wetland impact sites and nearby known wetlands. The result was a set of environmental profiles for various wetland types. These environmental profiles were then applied to the digital database as sorting criteria to generate a map indicating sites with the potential to support forested wetlands. This map was in turn overlaid with an existing forest cover data layer to pinpoint likely locations for degraded forested wetland systems.

In addition to describing the methods employed, this paper addresses advantages and limitations of this approach to site selection, including issues of scale and accuracy, costs, and data reliability.

An Evaluation of the Hydrogeomorphic Approach for Assessing Restored Forested Wetland Functions in the Puget Sound Lowlands

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The hydrogeomorphic approach (HGM) to assessing wetland functions was utilized in permitting, design, and implementation of restoration and monitoring protocols for a forested wetland ecosystem on March Point in Padilla Bay, Washington. HGM was used to assess depression and slope wetland functions in four categories: hydrology, biogeochemistry, plant community maintenance, and habitat/faunal support. Fifteen reference sites, classified on the basis of hydrologic and geomorphic features, were used to develop reference standards for impact assessment, design, and monitoring protocols. When used in these contexts, HGM provided a mechanism whereby (1) impacts could be assessed and explicit restoration goals could be negotiated, (2) reference standards could be used as restoration design templates, and (3) the restoration of wetland function could be monitored and compared to regional reference sites, and, if necessary, appropriate contingency measures could be identified and implemented. After one growing season, monitoring results indicate that all classes of wetland functions increased relative to pre-project conditions, yet most functions remained lower than reference standards. All hydrologic functions increased relative to pre-project conditions, and surface and subsurface water storage functions were

comparable to reference standards. Biogeochemical functions also increased relative to pre-project conditions, but water quality conditions did not differ significantly from pre-project conditions. Plant community maintenance functions increased due to the introduction of native wetland trees, shrubs, and herbs. However, percent cover, density, and basal area were lower than reference standards. Habitat/faunal support functions such as habitat structure, interspersion, and connectivity also increased, yet they remained lower than reference standards. The status of the wetland ecosystem restoration was assessed after the first growing season using estimates of change relative to specific wetland functions. Study results show that HGM can be used to assess impacts and to design wetland restoration. Furthermore, HGM can be used to assess the restoration of wetland ecosystems by measuring relatively short-term changes in wetland functions and by comparing results to pre-project conditions and to reference standards.

Use of the Hydrogeomorphic Approach in Design of Riparian Wetland Restoration along the Central California Coast

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The hydrogeomorphic approach (HGM) classifies wetlands based upon their geomorphic setting, water source and transport, and hydrodynamics. In each hydrogeomorphic wetland class, HGM assesses functions in four categories: hydrology, biogeochemistry, plant community maintenance, and habitat/faunal support. The procedure provides a mechanism by which (1) impacts can be assessed and explicit restoration goals can be negotiated, (2) reference standards can be established and used as restoration design templates, and (3) the restoration of wetland functions can be monitored and compared to regional reference sites, and, if necessary, appropriate contingency measures can be identified and implemented. In this study, 32 riverine wetlands and 19 estuarine wetlands were sampled along the Central California Coast. These data have been used to develop reference standards for impact assessment and design criteria fox a restoration of the near-coast reach of Calera Creek in Pacifica, California. With reference standards clearly articulated, it has been possible to ascertain which physical and biological factors need to be addressed in the restoration design procedure. The reference standards show an ability to store surface water in pits, sloughs, channels, and ponds adjacent to a wide, shallow channel located in a broad floodplain. Energy reduction during high flows is enhanced by high scrub-shrub and tree components. Pre-project conditions are characterized by little surface storage capacity. Pits, sloughs, channels, and ponds are absent, the channel is narrow, deep, and highly incised, and there is virtually no floodplain. Furthermore, velocity reduction is low due to low scrub-shrub and tree components. These results suggest that the design of the ecosystem restoration should focus on moving the creek out of its currently incised channel and re-connecting it to its restored floodplain. To complete the analysis, measured and estimated levels of ecosystem functioning were compared for reference standards, pre-project, and post-project conditions. Projected effectiveness of the ecosystem restoration is shown by the changes in the measured and estimated functions. Functional assessment results provided a basis for negotiating permit conditions.

Middle Waterway Pilot Restoration Project

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Restoration of over 3 acres of intertidal, salt marsh, and riparian habitat in the midst of Tacoma's (Washington) heavily industrialized Commencement Bay has recently been accomplished. The project, planned over the past several years and constructed in June of 1995, has three basic components: the excavation of tidal channels similar to those existing in the adjacent relic mudflats; construction of a vegetated intertidal bench; and the removal of upland fill, isolation of a small quantity of metal debris, and resloping the head of the waterway. The twin goals of this pilot project are to create productive and diverse estuarine habitats for fish and wildlife and to provide an outdoor laboratory for studying various planting strategies,

monitoring protocols, and volunteer/stewardship models for future natural resources restoration efforts in the region. This successful joint industry-government-public partnership between Simpson Tacoma Kraft Company, Champion International, and the natural resource trustees [U.S. Depts. of the Interior (F&WS) and Commerce (NOAA), WA Depts. of Ecology, Fish and Wildlife, and Natural Resources, the Puyallup Tribe of Indians, and the Muckleshoot Indian Tribe] resulted in the conversion of 1.5 acres of industrial upland to estuarine high and low salt marsh and intertidal mud/sand fiat habitat. The natural shoreline edge along the +9 to +13 foot contour was increased from 840 to 960 feet. A riparian buffer and transition zone was built to screen, protect, and support the diversity of the remaining Middle Waterway mud flats.

Nearshore Industrialized Areas of Port Gardner, Washington, and their Effects on Early Marine Life History of Anadromous Fishes

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Port Gardner, a natural, deep-water harbor adjacent to the Snohomish River estuary, is critical both to the economy of Everett and to local fish and invertebrate resources. Changes in juvenile salmonid habitat quality and function (e.g., feeding and refuge) were evaluated relative to historic development of the waterfront. For decades, commercial and industrial uses progressively degraded nearshore habitats, which function as migration corridors for juvenile salmonids, into a maze of piling walls, vertical bulkheads, finger piers, and over-water sawmills. Log storage on the mudfiats created obstructions to migratory fish and reduced benthic productivity by shading or compressing the sediments. Chemical and perhaps thermal discharges from a variety of industries also negatively affected water and sediment quality. By the late 1950s, many of the historic sawmills were closed although several pulp mills were active for many more years.

Large-scale filling along the shoreline which continued into the 1970s eliminated large areas of shallow water mud fiats, contributing to the overall reduced biological productivity of nearshore Port Gardner. At the same time, however, these fills often covered in-water pile accumulations of sawdust and debris and simplified the shoreline, eliminating many of the pile walls and floating and over-water structures. Fills were either riprapped or bulkheaded for structural stability.

More recent and improved water quality regulations and log handling practices, as well as regulation of nearshore dredging and filling, are changing conditions found along this corridor. Redevelopment actions, such as replacing vertical bulkheads with sloped riprap, also may improve productivity for salmon food organisms and enhance refuge from predators. Source controls and redevelopment of the shorelines in Port Gardner and restoration of intertidal marsh habitats in the estuary are reversing, to some degree, historically negative environmental effects of urbanization on nearshore habitats encountered by outmigrating juvenile salmon.

Importance of Tidal Channel Geomorphology to Restoring Ecological Functions of Coastal Wetlands

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Divergent ecological functions in constructed versus natural coastal marshes may result from inherent differences in marsh geomorphology, especially morphology of tidal channel systems. We hypothesize that nutrient cycling, species assemblages, primary and secondary production, and species interactions are strongly influenced by such landscape-scale features. Dendritic channel networks form through lengthy processes of sediment erosion and accretion, channel migration and extension, and plant and animal community development, processes that ultimately define complex, ecologically-important features such as fish and wildlife habitat. Although available geotechnical tools can explain influences of tidal prism and range, channel flow and sediment transport on tidal channel systems, we still lack empirical tests of relationships between

tidal channel geomorphology and ecological function. Using examples from Gulf of Mexico, California and Pacific Northwest coastal marshes, we infer how tidal channel planform (e.g., channel density, bifurcation, and sinuosity), cross section, and slope influence fish, macroinvertebrate and avifauna access, refuge and foraging, and describe how development of tidal channel systems and ecological functions both differ among these regions. In restoring and constructing coastal marshes, we are faced with decisions of how to promote natural channel systems, ranging from no intervention to extensive engineering manipulations. A better understanding of basic and empirical geomorphic principles that regulate natural tidal channel formation along pathways toward predictable ecological functions will allow us to make more educated decisions about marsh restoration.

Creating Landscape Restoration Opportunities: A Case Study at Rattlesnake Lake

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Changing patterns in visitor recreational use can serve as catalysts for redesign, creating opportunities for landscape and habitat restoration. In master planning for Rattlesnake Lake in the Cedar River Watershed, Jones & Jones recognized possibilities for improving habitat and water quality while addressing visitor use issues and expanding recreational trails and facilities. To achieve multiple objectives, we re-routed a section of road away from the lake edge and re-sited parking areas to disturbed land on higher ground. This plan created possibilities for restoring riparian habitat and planting new upland forests and meadows, while providing nearly a mile of recreational trails. In developing the design, we proposed the use of bioengineering techniques and native riparian plantings along the lake edge, and plan to employ native forest succession to restore upland areas. As a result of the road re-routing, the client will also daylight and restore over 100 feet of culverted creek. In solving multiple issues, support from the client and user constituency has been maintained. Interpretive signs will highlight natural and cultural history, explain restoration processes, and promote personal stewardship of the Rattlesnake Lake area.

An Assessment of the Mycorrhizal Fungal Status of Wetland Prairie Plant Species: Implications for Ecological Restoration

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Wetland ecosystems are important because of their ecological significance in such areas as natural waste and nutrient recycling. Much has been accomplished in determining the floristics and edaphic characteristics of these areas but little is known about the function of the below ground system. This study was designed to collect descriptive data on the mutualistic mycorrhizal fungal community of a wetland prairie (Zimmerman Prairie, Greene County, Ohio) associated with plant species diversity and soil characteristics. Preliminary analysis of the soil sample from the wetland prairie indicate a positive correlation between mycorrhizal root infection and soil moisture gradient for a 7-month time period. The overall analysis of the wetland prairie will determine the significance of the mycorrhizal fungi and will assist in the restoration of other wetlands.

Watershed Restoration in Deer Creek, Washington–A Ten-Year Review

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Deer Creek is a tributary to the North Fork Stillaguamish River in northwest Washington State. Approximately two thirds of the watershed is in federal ownership. Deer Creek once supported one of the largest populations of native, wild summer run steelhead in the Pacific Northwest, and also provided habitat for coho salmon and native char populations. Concentrated timber management activities on landslide-prone areas throughout the watershed and the occurrence of a number of major storm events over the past 30 to 40 years have been the major reason for the loss or degradation of aquatic habitat and a subsequent decline and near extinction of the native fish populations. Since the early 1980's major landslides and flood events have

dropped these anadromous fish populations to critical levels of viability. Beginning in 1984, Deer Creek has been the focus for comprehensive watershed planning and restoration. All landowners began to coordinate short and long term timber harvesting operations within the watershed; at this time the U.S. Forest Service deferred further timber harvesting in Deer Creek due to the critical decline in fish populations and the loss/decline in fish habitat conditions. Watershed restoration was initiated on federal land in 1984 with the goal of restoring the watershed's physical and biological processes that formed and maintained the favorable aquatic habitat conditions of the past. Initial restoration project emphasis was to reduce the occurrence of mass soil movement (landslides and debris torrents) from logging roads, old harvest units, and naturally unstable areas. Another phase of restoration work occurring at the same time was upgrading and stormproofing the existing road system. Second stage of work was restoring structure to the stream channel network with the goal of controlling the bedload transport and deposition as well as streambank stability. In 1994 more than 460 adult steelhead returned to Deer Creek, a significant increase from the past 6 to 7 years when less than 100 returned to spawn. This documented increase has been due mainly to the increase in the freshwater survival of steelhead juveniles. By protecting and restoring aquatic habitat conditions in Deer Creek anadromous fish populations are rebounding in this watershed. This is a review of watershed management efforts as we pass the 10-year mark and look toward the future.

Stream Restoration in an Alaskan Subalpine Placer-Mined Watershed

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Techniques to promote riparian ecosystem recovery are being tested in a subalpine placer-mined watershed in Alaska's Denali National Park and Preserve. Stream and floodplain geometry were designed to allow the stream to develop floodplains, sinuosity, and pools and riffles similar to premining conditions. Bioengineering techniques, which utilized subalpine plant materials, were tested for stabilization. Earthwork in 1991 and 1992 emphasized recontouring artificially raised floodplains to a shallow slope to restore natural floodplain processes, and constructing a segment of new channel away from the valley wall. Anchored alder and willow brush bars were installed to stabilize the floodplain and collect sediment. A moderate flood occurred at the end of the 1992 construction. The brush bars stabilized the flood-plain surface and trapped sediment, but unconsolidated bank material and a lack of bed armor for the new channel segment led to some bank erosion, slope changes, and an increase in sinuosity in several reaches. To address these problems, we tested new techniques in 1994. These included construction of pools, rock weirs, and meanders with deeper channels. Bioengineering techniques focused on stabilization and revegetation of the newly-created meander cut banks and point bars. On cut banks, root wads and brush bars (redesigned to protect both flood-plain and bank) were tested. On point bars, riparian vegetation and soil from sites away from the present channel were bulldozed over the new point bars, or a brush layer was buried with the ends projecting into the channel. All bioengineering methods used buried willow branches for revegetation.

Restoration of Ashbridge Marsh: Potential Consequences for Water Quality and the Aquatic Community of the Lower Don River

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Investigation of a delta marsh restoration project proposed for the Don River in Toronto, Ontario underlines several concerns of constructed wetland projects designed for water quality improvement and aquatic habitat enhancement. The Don is a highly urbanized river, which has undergone significant physiographic modifications, and which continually receives a complex mixture of conventional, metallic and organic contaminants from multiple point and nonpoint sources. Wetland processes offer a limited capacity for temporary storage of contaminant inputs, and potential reactions may actually produce more toxic and/or bioavailable forms of some chemicals. Concentration of urban contaminants in vegetation and sediments at a site intended to provide spawning and feeding habitat for aquatic fauna may result in toxic responses

affecting the aquatic community. The project is therefore considered inappropriate for the lower Don, as rehabilitative objectives related to water quality and aquatic habitat are not likely to be met in the long term. Recommendations center on increased priority for preventative measures directed further upstream, including source reduction of contaminant inputs. The identification of potential adverse consequences suggests that constructed wetland projects are generally not suited to dual applications for water treatment and habitat enhancement, particularly in urban environments featuring complex mixtures of contaminants.

Native Plant Salvaging: Plant Recycling for Watershed Restoration

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Volunteers in King County, Washington are digging up native plants from sites slated for development for use in a variety of revegetation projects. As part of a unique program initiated by King County Surface Water Management (SWM) in 1992, the "rescued" plants are used to restore stream banks, wetlands, and buffers near a variety of capital improvement and roads projects. Developers, government, and individual citizens collaborate to make the program work. In its first 3 years, the Native Plant Salvage Program has successfully:

- Involved the public in a cooperative effort on public works projects
- "Recycled" valuable locally-adapted plant materials from construction sites
- Become a reliable source of low-cost plant materials for a variety of bioengineering and restoration projects
- Educated the public in native plant propagation and environmental restoration techniques.

The Native Plant Salvage Program saves King County Surface Water Management money in plant costs even after accounting for the staff involved in running the program. It provides a reliable source of native plant material for restoration and enhancement projects. Besides the obvious benefit of providing inexpensive plants for restoration projects, King County Surface Water Management's Native Plant Salvage Program has had more far-reaching effects. It created the opportunity for positive citizen participation in public works projects. It has enhanced relationships with the local development community. It has won several awards for unique recycling opportunities and for promoting sustainability. From its inception, the strategies and techniques outlined above have undergone continuous improvement and refinement. It is with great pride and enthusiasm that the staff at King County are able to share the working "blueprint" for a program that is serving as a model for similar programs across Washington State and the Pacific Northwest region.

Using Citizen Monitoring For Watershed-Level Assessment of Urban Restoration Goals

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Small scale restoration projects involving citizens and schools are springing up all over the country, especially in urban settings and along streams. While these grass roots efforts can have tremendous benefits including promoting stewardship and restoration ethics on a community level, often these projects suffer ecologically from insufficient preliminary study of existing conditions and watershed level needs and constraints. Without good baseline information, sound project design is problematic, and the resulting setbacks or failures can seriously undermine the long-term success of the project. Including citizens in a preliminary assessment phase prior to project design can significantly increase the value of the program in terms of citizen education and involvement, as well the odds of success for the project ecologically. Methodologies for using citizens to collect information on habitat conditions have been developed and tested by the Coyote Creek Riparian Station. Citizens are collecting data on habitat fragmentation, existing native and nonnative vegetation, and extent and location of pollution impacts. These data are ideal for identifying restoration opportunities and constraints, as well as for encouraging planners to design their projects with watershed level goals in mind. CCRS staff will present these methods and demonstrate the GIS-based habitat maps and information, which can help make sound project design possible.

Characterization of Columbia Basin Watersheds: Providing Regional Context for Stream Restoration Goals

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Setting priorities and realistic expectations for basin-wide stream restoration requires extending the principles of geomorphically-based restoration of stream reaches to larger geographic areas, as well as explicitly incorporating linkages to upland terrestrial systems. As part of the Interior Columbia Basin Ecosystem Management Project, we have developed methods to associate the hydrologic function of aquatic systems to the terrestrial systems in which they are nested, and have developed a continuous characterization of watersheds at two-scales across the Interior Columbia Basin (i.e., about 13,000 watersheds of approximately 10 to 25 square miles, nested within 248 watersheds varying from about 50 to 300 sq. mi.). Using principles developed by the U.S. Forest Service for hierarchical stratification of aquatic and terrestrial ecological units, individual watersheds are characterized based on their geoclimatic, vegetative, hydrologic, and morphometric characteristics. This stratification is used to derive comparative indices of inherent soil erosion potential, potential sediment delivery to streams, landslide and debris flow hazards, and sensitivity to road-building. These indices are relativized to permit comparisons across the Interior Columbia Basin or on a province-by-province basis. The watersheds are further described in terms of probable Rosgen stream types to assist in characterizing watershed sensitivity to changes in flow and sediment, and the inherent recovery potential of the characteristic channel types. Watersheds are also classified based on similar geoclimatic characteristics, and on the degree of anthropogenic influence present within the basin.

Restoring Native Communities to the Sandy River Delta

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The 1,400-acre Sandy River Delta, is located at the western end of the Columbia River Gorge National Scenic Area, east of Portland, Oregon. Lewis and Clark and other early surveys reported a mostly forested site, rich in wildlife. Beginning with the original donation land claim in 1851, the site has been subject to severe disturbances. Native vegetation has been cleared and the delta used for agricultural purposes. Attempting to improve smelt runs, the main channel of the Sandy River was dammed in the 1930s, rechanneling the mouth of the river. The site has been diked, ditched, drained, and levied. The extensive hydrological alteration of the Columbia River system has greatly altered the natural disturbance regime of flooding. Reed canarygrass *(Phalaris arundinacea)* and Himalayan blackberry *(Rubus discolor)* currently dominate significant acreage. In 1991 the U.S. Forest Service acquired much of the delta and through an extensive planning process, including an Environmental Impact Statement, has adopted a management plan which will reforest the northern and western parts of the Delta, but maintain open habitats (meadows and seasonal wetlands) in the southern portion. Currently, the Forest Service is establishing a series of innovative experiments designed to control reed canarygrass and Himalayan blackberry, reestablish preimpact plant communities, and create a mosaic of habitats using native plant species. The plan may allow for re-introduction of Columbian white-tailed deer *(Odocoileus viginianus leucurus),* a federally-listed endangered species.

Fundamentals in Restoring Biological Integrity to Regulated, Large Floodplain Rivers–A Case Study

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As North America's largest river, the Mississippi has great geographic, ecologic, and sociologic significance. Free flowing with the excess moisture of its basin for millennia, the Mississippi today is receiving water from a landscape whose hydrologic character has been drastically altered by human activity. Extremes in the hydrograph and in sediment budgets are accentuated in tributary streams. Hydrodynamic processes in the river's corridor have

also been substantially changed. Channel-training structures and dams have been constructed to maintain a shipping canal. Seasonal water-level rhythms have been masked by impoundment. Sediment erosion, transport, and deposition patterns have been profoundly changed by redirecting hydraulic energy in the river. As a result, the upper Mississippi is losing the complex physical template around which its great biological diversity was organized. Reconciling society's need to manipulate these physical processes with their fundamental role in creating the physical architecture, which sustains the river's ecosystem, is essential to successful ecological restoration.

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